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Research and Development

Treatment of Secondary Effluent by Infiltration-Percolation

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August 1979

TREATMENT OF SECONDARY EFFLUENT BY INFILTRATION-PERCOLATION

by

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FOREWORD

The Environmental Protection Agency was established to coordinate administration of the major Federal programs designed to protect the quality of our environment.

An important part of the agency's effort involves the search for information about environmental problems, management techniques and new technologies through which optimum use of the nation's land and water resources can be assured and the threat pollution poses to the welfare of the American people can be minimized.

EPA's Office of Research and Development conducts this search through a nationwide network of research facilities.

As one of these facilities, the Robert S. Kerr Environmental Research Laboratory is responsible for the management of programs to: (a) investigate the nature, transport, fate and management of pollutants in groundwater; (b) develop and demonstrate methods for treating wastewaters with soil and other natural systems; (c) develop and demonstrate pollution control technologies for irrigation return flows; (d) develop and demonstrate pollution control technologies for animal production wastes; (e) develop and demonstrate technologies to prevent, control or abate pollution from the petroleum refining and petrochemical industries; and (f) develop and demonstrate technologies to manage pollution resulting from combinations of industrial wastewaters or industrial/municipal wastewaters.

This report contributes to the knowledge essential if the EPA is to meet the requirements of environmental laws that it establish and enforce pollution control standards which are reasonable, cost effective and provide adequate protection for the American public.

William C Galegar

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Director
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ABSTRACT

The objective of this study was to evaluate the performance of an infiltration-percolation system for improving the quality of secondary effluent from a municipal wastewater treatment system. This was done by constructing three infiltration-percolation basins and monitoring their influent and effluent quality over a two year period.

The facility consisted of three infiltration-percolation basins of sizes ranging between 0.24 hectares (ha) [0.6 acres(ac)] and 0.36 ha (0.9 ac). Unchlorinated secondary wastewater effluent was applied twice a week to each basin at loading rates which varied between 12.2 meters/year (m/yr) [40 feet/year (ft/yr)] and 48.8 m/yr (160 ft/yr). The wastewater percolated through loamy and clay sands covering alluvial sand and gravel. The percolate was collected 2.4-3.0 m (8-10 ft) below the surface by underdrains for discharge.

Analyses of the basin influent and effluent collected from the underdrains indicated that the systems were generally effective in reducing the wastewater concentrations of COD, coliform organisms, and ammonium nitrogen. Phosphorus leakage occurred to some extent in each of the basins, with the most heavily loaded basins yielding the highest phosphorus concentrations in the discharge water. The nitrate concentration of the water increased significantly because of nitrification of the ammonium nitrogen retained within the soil matrix. The concentrations of hardness, alkalinity, and chlorides also showed significant increases in the percolate water.

This report was submitted in fulfillment of Grant No. R803931 by the City of Boulder, Colorado, and the University of Colorado, under the sponsorship of the U. S. Environmental Protection Agency. The report covers the period August 1, 1975, to June 30, 1978, and the work was completed as of October 15, 1978.

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FIGURES (continued)

FIGURES (continued)

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ACKNOWLEDGEMENTS

The authors wish to thank the many people who contributed to the completion of this research effort. The support of the City of Boulder wastewater treatment plant staff is gratefully acknowledged. Mr. Arthur Dike, Mr. Charles Tregay, and Mr. Alton Ragsdale were particularly helpful in the operation and maintenance of the test facility. Similarly, Mr. Chris Rudkin and Mr. Steve Miller provided much information and assistance in the development of sampling procedures at the site.

Sampling, analysis, and operational coordination were provided by the following individuals during their graduate training at the University of Colorado: Mr. Bob White, Mr. Paul Hamilton, Mr. Dan St. John, Mr. Bill DeOreo, Mr. Bill Earley, Mr. Joe Tamburini, and Mr. Paul King.

The authors wish to express appreciation to the Robert S. Kerr Environmental Research Laboratory for its support of this project, and especially to Mr. Lowell Leach, and Mr. Richard Thomas who served as Project Officers during portions of the study. The assistance of Mr. Roger Dean, Region VIII, EPA, in providing coordination is also acknowledged.

SECTION 1

INTRODUCTION

In 1972 the United States Congress passed Public Law 92-500, a bill which committed this country to the task of upgrading polluted waters and preventing degradation of clean waters. Three levels of treatment were identified as the primary goals of this act. By 1977, secondary treatment or its equivalent would be required; by 1983, the best available treatment would be required; and by 1985, the goal was to eliminate the discharge of pollutants to our waterways. In order to comply with the strict standards set forth in this law, communities and industries alike were faced with providing some form of highly efficient wastewater treatment. This treatment would be directed at removing undesirable wastewater constituents such as nitrogen, phosphorus, suspended solids, refractory organics, and microorganisms. Their removal could be accomplished through application of either in-plant physical, chemical, and biological processes, or by land treatment.

In addition to establishing strict discharge requirements, P.L. 92-500 also specified that land treatment be considered as an alternative to other advanced treatment processes. With the implementation of this requirement, it has become apparent that land treatment combines the various physical-chemical, and biological processes into a single process which is very effective in many wastewater treatment situations. Land treatment is especially attractive for smaller communities where the land is readily available and the cost effectiveness of physical-chemical processes is somewhat questionable.

Three major methods of land application have been demonstrated to provide effective wastewater treatment. They include: irrigation, overland flow, and infiltration-percolation. Successful application of one of these land treatment methods depends on several interrelated factors such as the soil type, geology, topography, ground water characteristics, and climate. As a result of these considerations, land treatment processes are very site specific. Selection of the best land use system for a particular location may depend on any one, or all of these considerations.

The study discussed in this report involved the operation of three small infiltration-percolation (i-p) basins in Boulder, Colorado. These infiltration-percolation basins were installed as a demonstration project in the spring of 1976 to treat a portion of the secondary effluent discharged by the City of Boulder wastewater treatment facility. The treatment efficiency of the infiltration-percolation system was monitored throughout the period of operation by regular sampling of the applied secondary effluent and the renovated water. The performance of this system was the subject of this report.

SECTION 2

CONCLUSIONS

The infiltration-percolation system demonstrated excellent capability for polishing secondary effluent in the context of the Boulder, Colorado wastewater treatment situation. Under proper conditions of hydraulic loading and loading cycle control, the system was capable of providing very high levels of removal for virtually all wastewater constituents of major pollution significance.

Infiltration-percolation systems were shown to be capable of operation throughout the year in either flooded basins or ridge and furrow configurations where appropriate surface maintenance was provided. The necessary surface maintenance consisted of scarification to control plant development in the bare basins so that ice which formed in the cold winter months would not be anchored, but could float free with successive wastewater applications.

The infiltration rate in the ridge and furrow system appeared to be less affected by wastewater suspended solids deposition than the flat basins which were completely flooded. This suggests that it would be possible to operate ridge and furrow systems longer between extended drying periods than flooded basins loaded at comparable rates.

When a basin was loaded at a sufficiently high hydraulic rate to stress the infiltration capabilities of the soil system, one of the indications of the stressed condition was a deterioration of the quality of the effluent from the basin. This quality deterioration was evidenced by increased concentrations of both phosphorus and ammonium nitrogen, as well as a reduction in the amount of nitrate nitrogen in the basin discharge.

The basins demonstrated good capability for making major reductions in the concentration of fecal coliforms in the unchlorinated secondary effluent. Nevertheless, significant concentrations of these bacteria were still detected after passage of the wastewater through 2.4-3.0 meters (8-10 feet) of the composite soil mixture found in the Boulder treatment location.

Significant removal of heavy metals was observed in the infiltration-percolation soil system. Most of this removal occurred in the tight clay loam layer at the top surface of the soil profile.

SECTION 3

RECOMMENDATIONS

From the observations developed in conjunction with this investigation, the following recommendations seem appropriate:

The capability of the infiltration-percolation basins for treating a wide range of wastewater pollutants suggest that these systems would be appropriate for treating primary as well as secondary effluent. It is recommended that a comparative study be performed which would assess the acceptable hydraulic loading rates and product water characteristics with these two different levels of pre-application treatment.

From the aesthetic and operational perspectives, it would appear that there might be advantages to the use of a ridge and furrowed application system rather than a flat basin having no vegetation. A study should be undertaken to identify the loading schedule which would provide the best combination of hydraulic loading and treatment performance in this configuration. The aim of this effort should be to minimize the total land area requirements for providing this alternative application method.

In the study discussed in this report, little total nitrogen removal was observed. It was shown in the operation of beds 1 and 2 that if the loading rate was maintained at an appropriate level, the effluent ammonium nitrogen could be controlled at a concentration of less than 1 milligram/liter (mg/l). However, with the loading sequence practiced in this investigation, essentially all of the nitrogen was discharged from these beds as nitrate. Other investigators have demonstrated that substantial denitrification can be achieved in rapid infiltration systems if the loading sequence is appropriately managed. Modification of the loading sequence should be attempted at the Boulder facility for the purpose of maximizing the total nitrogen removal.

Significant phosphorus leakage was observed in the product water from all three of the basins. It is suspected that this leakage occurred in part because of the short hydraulic detention from the basin surface to the underdrains. It would be of interest to determine the impact of varying the underdrain placement to increase the contact time between the wastewater and the soil profile. This could be done by selectively closing off some of the existing drains in the Boulder infiltration-percolation system.

SECTION 4

EXPERIMENTAL SYSTEM AND PROCEDURES

SITE DESCRIPTION AND PHYSICAL FACILITY

The wastewater treated in the demonstration project facility was drawn from the effluent of the City of Boulder, 75th Street Wastewater Treatment Plant. This treatment facility processed approximately 0.53 cubic meter/second (m^3/sec) [12 million gallons per day (MGD)] of wastewater by means of a standard rate trickling filter, as is shown schematically in Figure 1.

The site of the infiltration-percolation system consisted of approximately 1.0 hectares (ha) (2.5 ac) of land located adjacent to the City of Boulder 75th Street Wastewater Treatment Plant, and about 152 meters (500 feet) south of Boulder Creek. The land, which was formerly rangeland, had a slight eastward slope ranging between 0-1%. The soil was classified as the Niwot series soil which has been described by the U. S. Soil Conservation Service as a layer varying from sandy clay loam to light clay loam superimposed over sand and gravel (1). These soils have been described to have moderate permeability and a high seasonal water table ranging from 0.15 meters to 0.46 meters (6 to 8 inches)(1). Underlying this soil at a depth of approximately 3.7 meters (12 feet) was the very impermeable Pierre Shale formation (2). The depth to ground water on the site was found to vary from 0.9 to 1.5 meters (3 to 5 feet) in May, 1975(2).

The City of Boulder Wastewater Utility Department designed and constructed the pilot plant facility. Construction began in December 1975 and was completed in April 1976. Three infiltration-percolation basins were constructed, with each basin separated by a berm approximately 0.76 meters (2.5 feet) high. All three basins were surrounded by an impermeable clay-core dike which was also 0.76 meters (2.5 feet) high. The 1.83 meter (6 feet) wide clay-core dike extended from the ground surface to the impermeable bedrock, and served to completely enclose the system, and minimize the interaction between the beds and the surrounding ground water. The South basin, referred to as Basin 1, was 0.35 ha(0.87 ac); the Middle basin, Basin 2, was 0.24 ha (0.60 ac); and the North basin, Basin 3 was 0.26 ha (0.65 ac). Figure 2 shows a plan view of the three basins and their respective sizes.

After the basin area was completely sealed, an underdrain system was installed to lower the existing water table and to collect the applied water during the operation of the system. The underdrain system for each basin consisted of two 0.18 meter (7 inch) perforated PVC drain pipes located

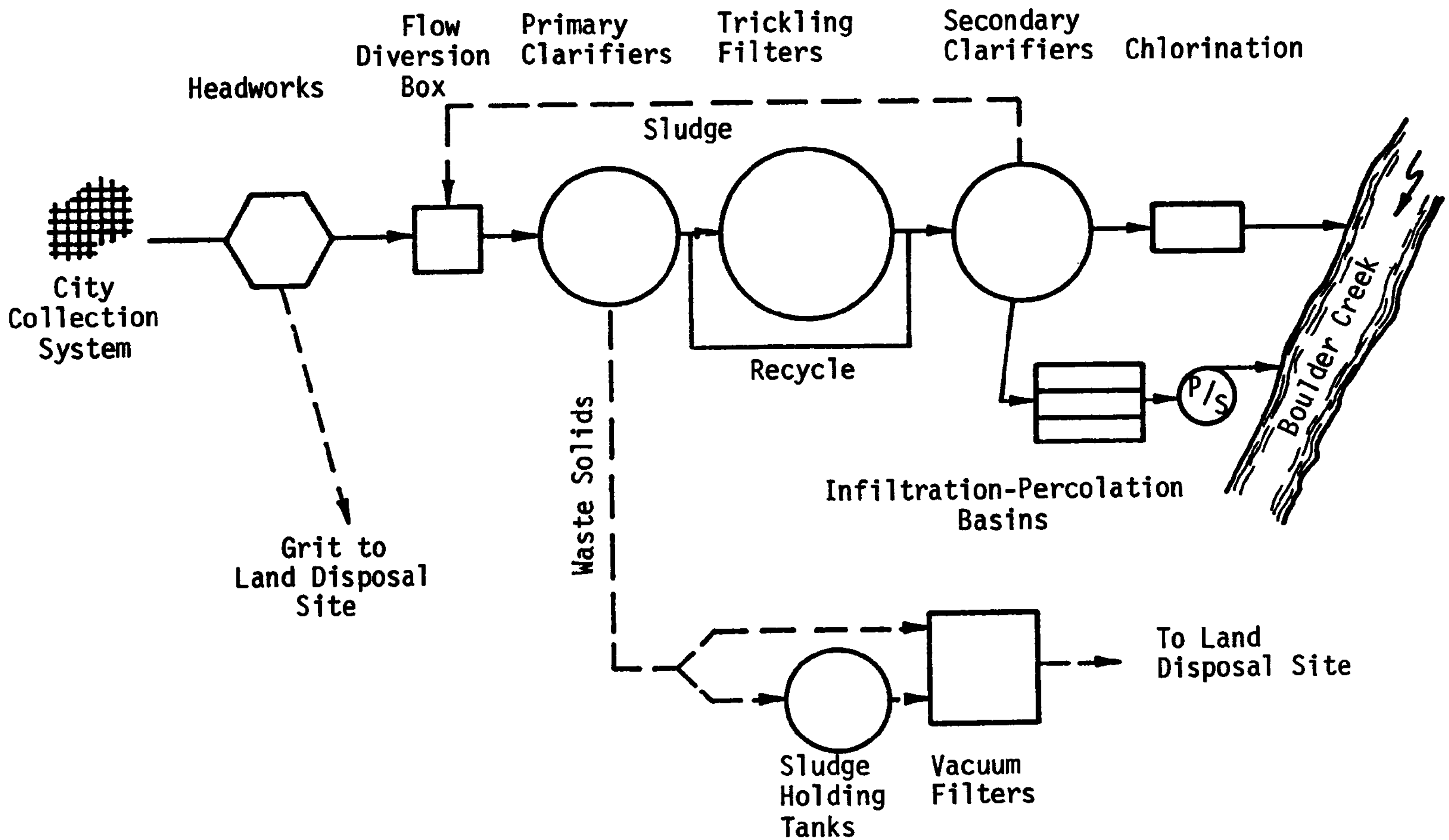


Figure 1. Schematic of Boulder wastewater treatment plant.

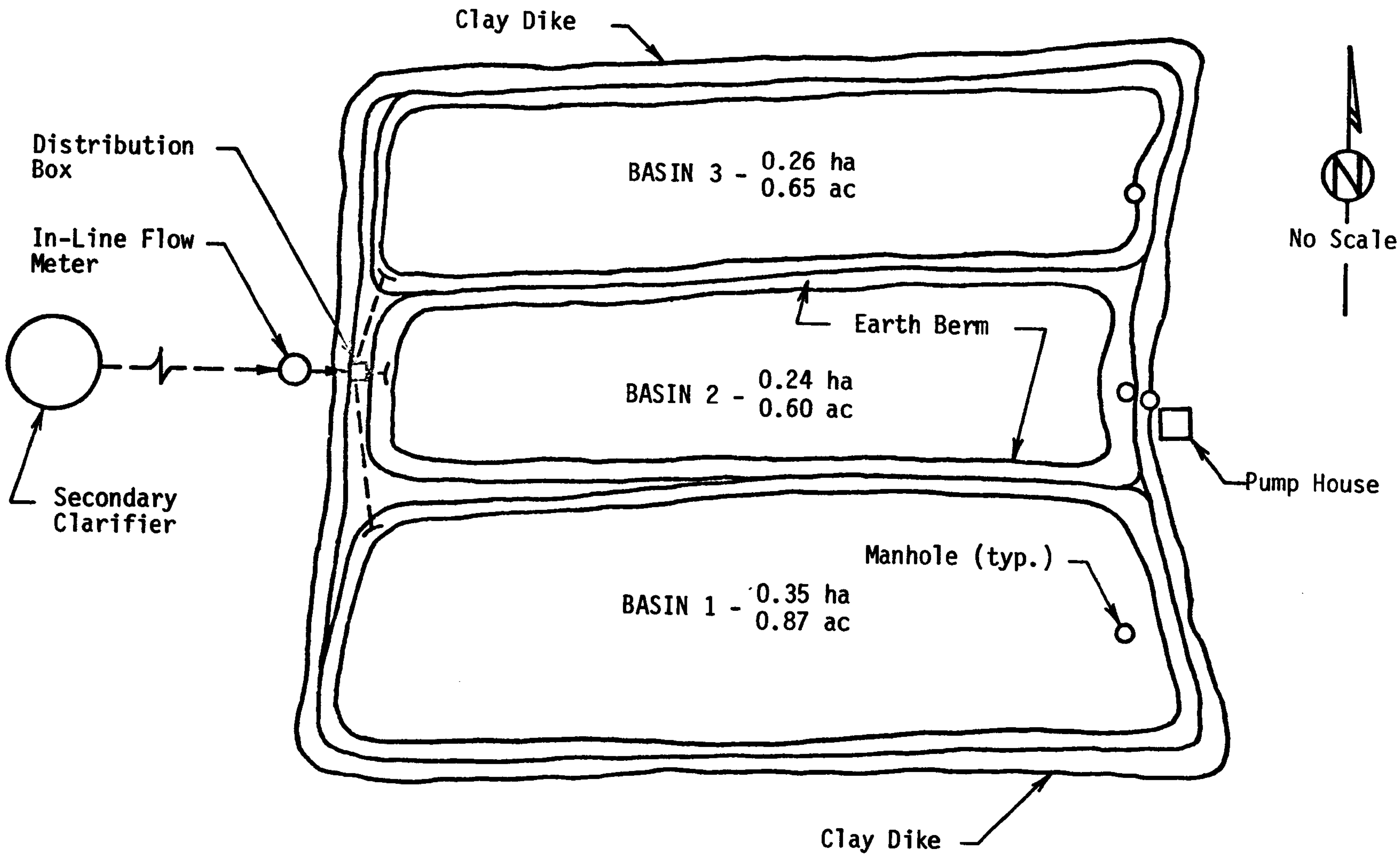


Figure 2. Infiltration-Percolation System Layout

2.43 to 3.05 meters (8 to 10 feet) under the basin surface. The collected water flowed by gravity to a manhole in each basin, and then to a central manhole where the monitoring and sampling took place. From this manhole the water flowed to a wet well and was pumped to its discharge point in Boulder Creek. A plan view showing the underdrain system is shown in Figure 3, and a cross section of the basins is shown in Figure 4.

Well points were installed in each basin to measure the rise and fall of the water table during the loading cycles. These well points were developed by augering holes through the beds, and installing 0.04 meter (1.5 inches) PVC pipes in the holes. These pipes extended from 0.91 meters (3 feet) above the ground surface to the bedrock. The locations and elevations of these well points have been indicated in Figure 5. The depth of the water table was measured by lowering a float down the pipe until the water surface was intercepted, and then recording the depth.

BASIN MODIFICATION

Following six months of treatment operation on the beds which have been described, a decision was made to modify Beds 2 and 3 in an attempt to improve their hydraulic performance. As such, in January, 1977 the top tight loamy layer of soil was removed from both of these beds. This necessitated the removal of the top 0.46 meters (18 inches) of soil from Bed 3, and the top 0.61 meters (24 inches) of soil from Bed 2. Both beds were subsequently graded to a flat surface with a slight eastward slope, similar to that in their initial condition.

While Basin 3 remained in the modified form throughout the rest of the study, Bed 2 was further altered in March, 1977 by the construction of a ridge and furrow system. The furrows were 0.46 meters (18 inches) deep and spaced approximately 1.98 meters (6.5 feet)on center. The furrows averaged 1.02 meters (40 inches) in width, and the ridges averaged 0.91 meters (36 inches) wide. There were 9 furrows constructed in Bed 2, each approximately 85 meters (280 feet) long. A furrow at each end was constructed to facilitate loading. A plan view of the three basins, and the modified areas, is shown in Figure 6. It should be noted that Bed 1 was not altered in any way, after the initial construction.

BASIN LOADING

Loading of the basins was accomplished by pumping the effluent from a secondary clarifier with a centrifugal pump which was powered by a portable gasoline engine. The unchlorinated wastewater from this point was pumped through approximately 297 meters (975 feet) of 0.36 meter (14 inch) PVC pipe to a distribution box which directed the water to the desired basin.

Each basin was loaded twice a week at 3½ day intervals. From October, 1976 to January, 1977, Bed 1 (surface of basin) was loaded at approximately 8:00am on Mondays and 7:00pm on Thursdays; Bed 2, 8:00am on Tuesdays and 7:00pm

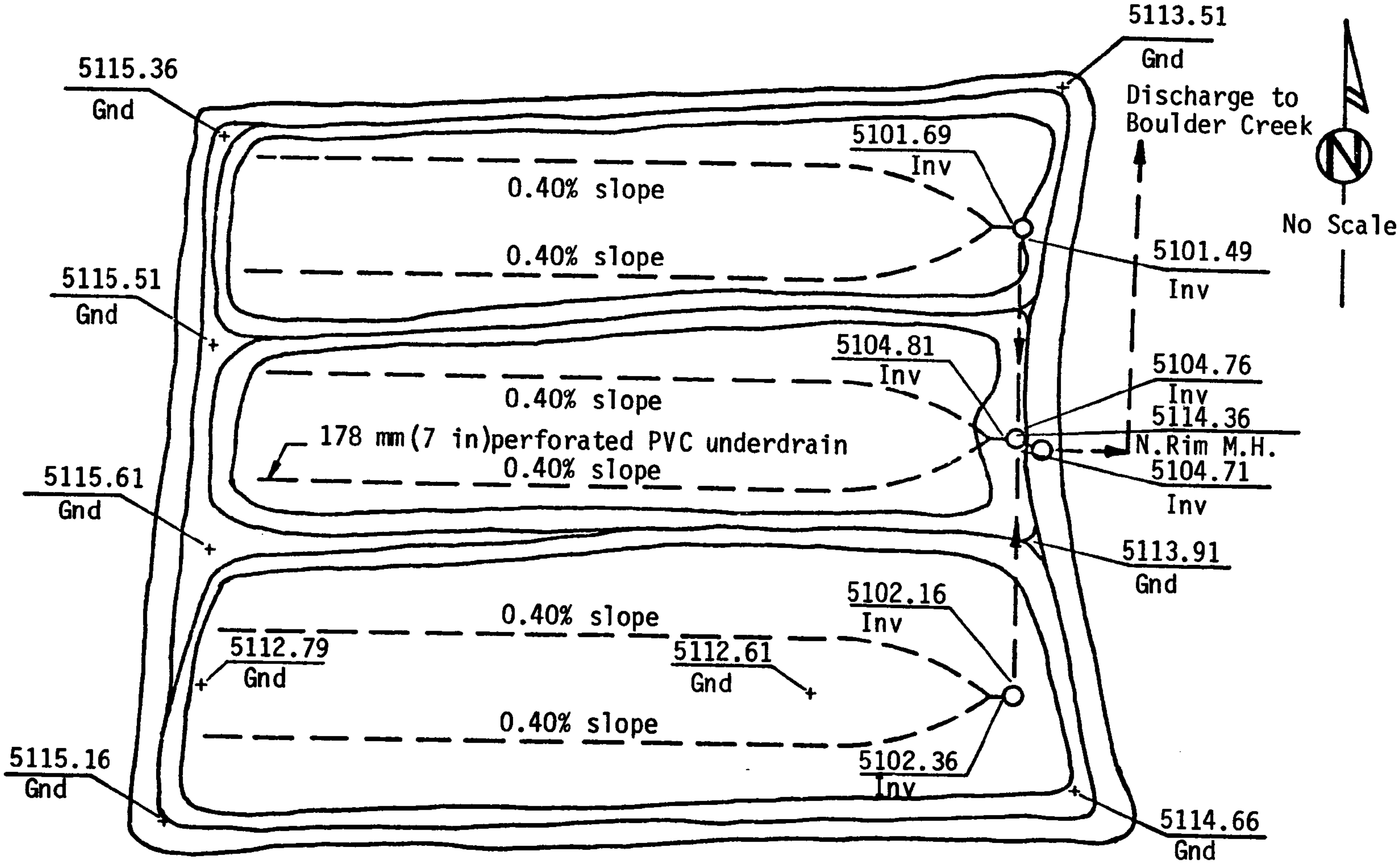


Figure 3. Basin underdrains.
(Elevations in feet.)

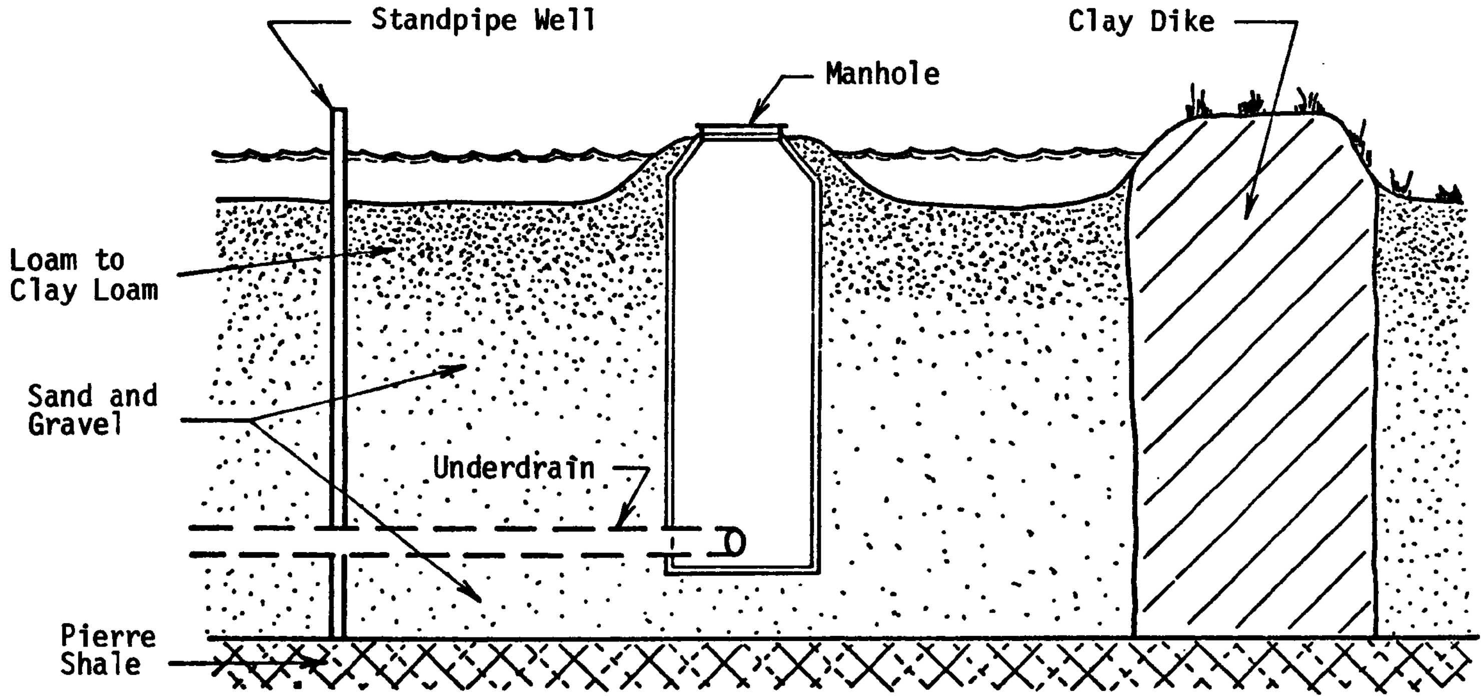


Figure 4. Typical section through basins.

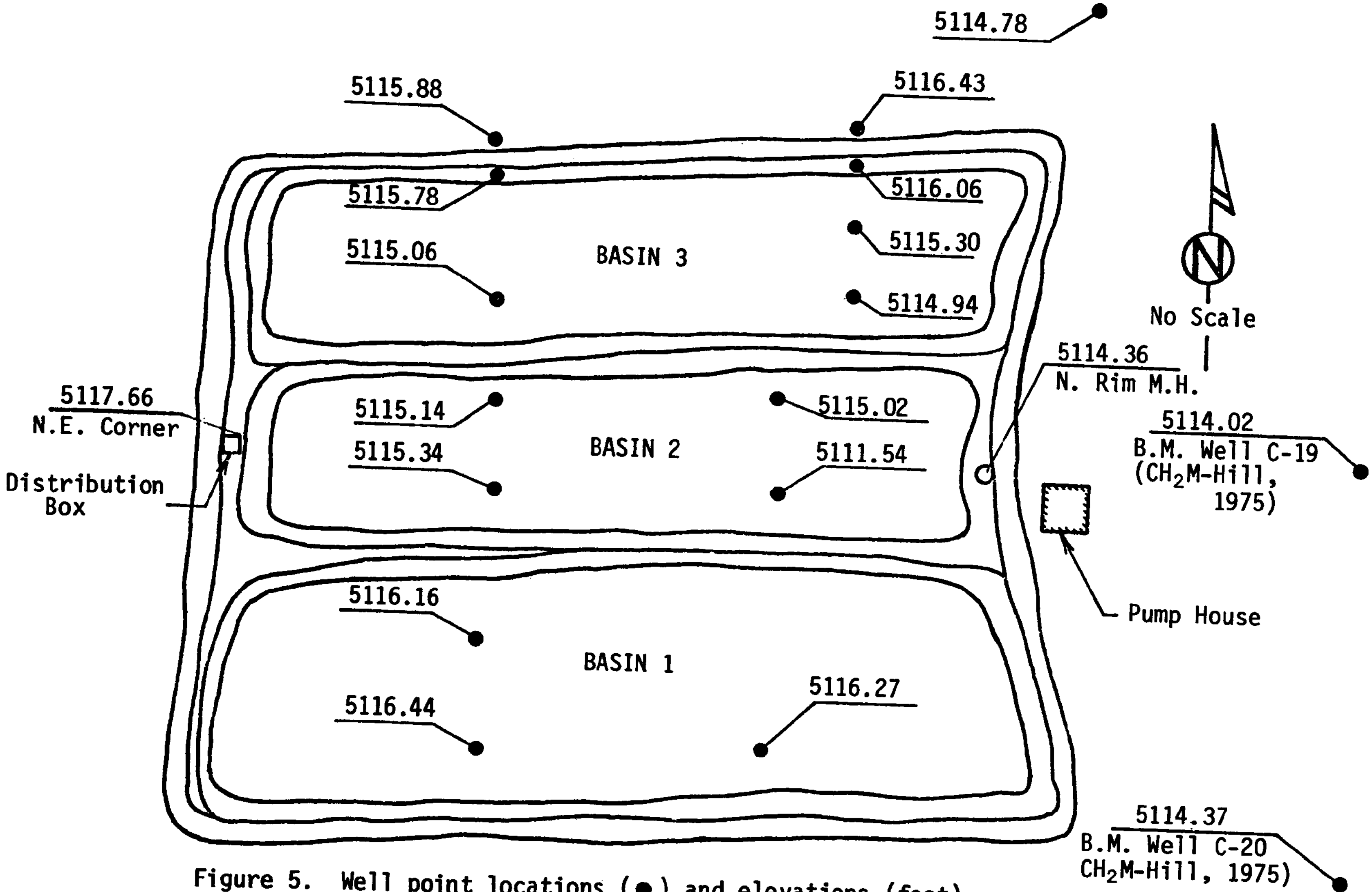


Figure 5. Well point locations (●) and elevations (feet).

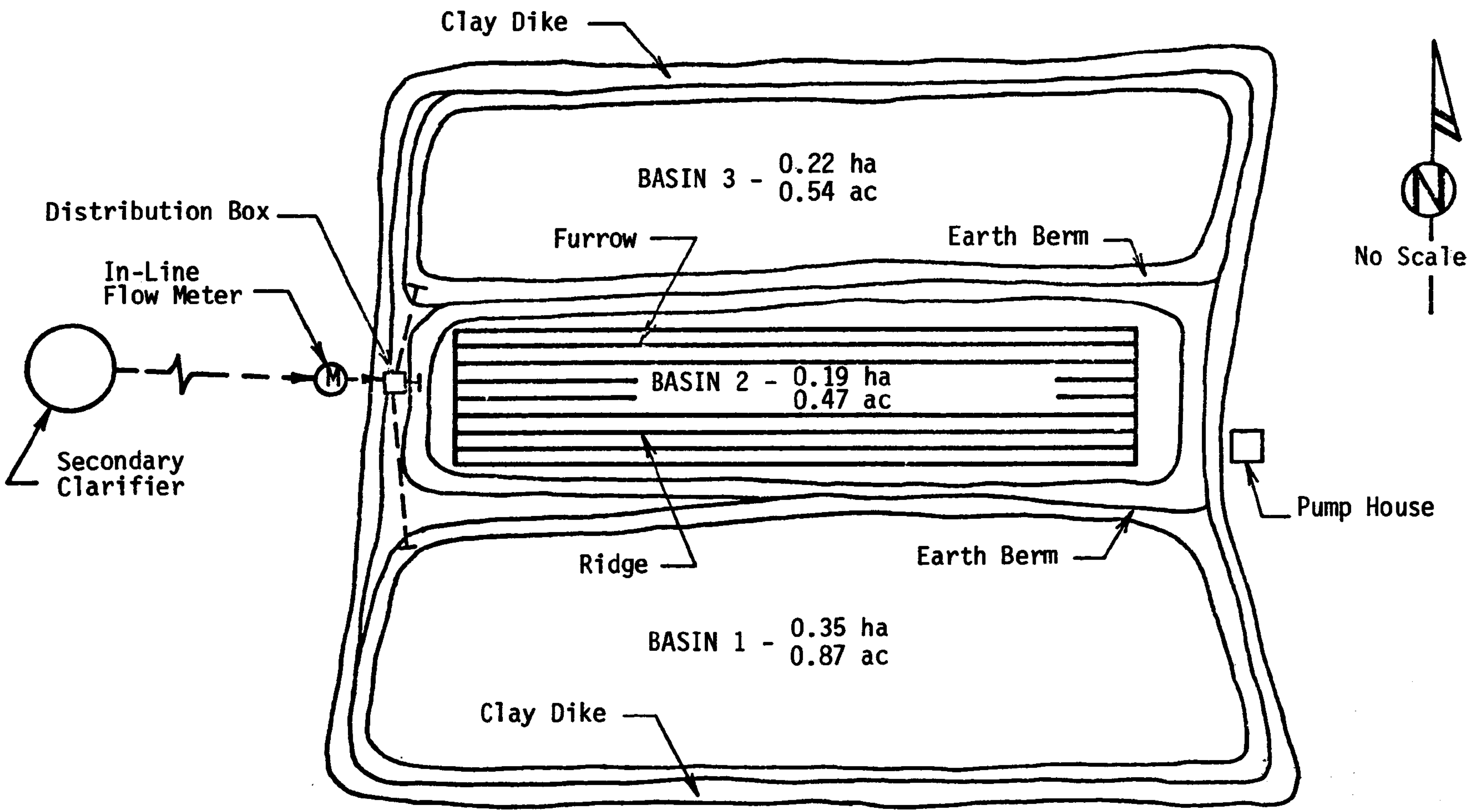


Figure 6. Modified basin configuration.

on Fridays; Bed 3, 8:00am on Wednesdays and 7:00pm on Saturdays. The beds were loaded on alternating days to facilitate the ease of operation, sampling, and analysis. In February 1977, the loading cycle was shifted one day later to begin on Tuesday morning instead of Monday morning due to a change in operator schedules. After six weeks of loading, all basins were dried and then scarified. The required length of drying time varied from one week in the summer to two or three weeks in the winter, depending on the weather conditions during the drying period.

SECTION 5

SAMPLING AND ANALYSIS

SAMPLING

Samples of both the influent and effluent of the three basins were taken at regular intervals. The influent samples were taken at the inlet pipe of each bed during loading. Following renovation treatment, the basin effluent was sampled at the center manhole, with samples collected independently for each basin at specified times after loading. Tables 1 and 2 summarize the loading and sampling schedule. Only the first loading of each week was sampled and monitored. The temperature, dissolved oxygen level, nitrogen, hydraulic flow, and infiltration rate measurements were monitored with time after the loadings. All other parameters were measured only once for each loading cycle on the sample collected 24 hours after loading.

The water samples were preserved immediately after collection. Samples collected for testing of nitrites and nitrates were preserved with 40 mg/ℓ $HgCl_2$, and stored at $4^{o}C$. Samples to be analyzed for organic nitrogen and ammonia were acidified with concentrated sulfuric acid to a pH of less than 2, and refrigerated at $4^{o}C$. These procedures were in accordance with those outlined in <u>Methods for Chemical Analysis of Water and Wastes</u> (3). Samples collected for all of the tests were refrigerated at $4^{o}C$, with the exception of those used in determining temperature and dissolved oxygen.

ANALYTICAL METHODS

All analyses, except those noted, were performed at the Sanitary Engineering Laboratory at the University of Colorado. The samples were subjected to the following series of analyses for purposes of this investigation: total solids, suspended solids, phosphorus, COD, temperature, coliforms, and the nitrogen series. In addition, selected samples were analyzed for Cd, Cu, Cr, Ni, Pb, and Zn during November-December, 1977.

Temperature measurements were taken immediately following the sampling at the pilot plant site. A mercury-filled centigrade thermometer, calibrated to $1^{o}C$, was used for these measurements.

Total kjeldahl nitrogen (TKN) was determined by acid digestion of 250 milliliters (ml) samples and distillation into boric acid. The TKN was titrated with standard sulfuric acid to the pH of the blank carried through the same procedures. The difference between the TKN and ammonium nitrogen was the organic

TABLE 1. LOADING AND SAMPLING SCHEDULE

(OCTOBER 1976 TO JANUARY 1977)

Day	Time	Basin 1	Basin 2	Basin 3
Monday	8:00 a.m.	Load Basin Measure Flow Sample Effluent		
	9:00 a.m.	Measure Flow Sample Influent Sample Effluent		
	11:00 a.m.	Measure Flow Sample Effluent		
	2:00 p.m.	Measure Flow Sample Effluent		
	8:00 p.m.	Measure Flow Sample Effluent		
Tuesday	8:00 a.m.	Measure Flow Sample for Complete Analysis	Load Basin Measure Flow Sample Effluent	
	9:00 a.m.		Measure Flow Sample Influent Sample Effluent	
	11:00 a.m.		Measure Flow Sample Effluent	
	2:00 p.m.		Measure Flow Sample Effluent	
	8:00 p.m.	Measure Flow Sample Effluent	Measure Flow Sample Effluent	
Wednesday	8:00 a.m.	Measure Flow Sample Effluent	Measure Flow Sample for Complete Analysis	Load Basin Measure Flow Sample Effluent
	9:00 a.m.			Measure Flow Sample Influent Sample Effluent
	11:00 a.m.			Measure Flow Sample Effluent
	2:00 p.m.			Measure Flow Sample Effluent

(continued)

TABLE 1. (continued)

Day	Time	Basin 1	Basin 2	Basin 3
	8:00 p.m.	Measure Flow Sample Effluent	Measure Flow Sample Effluent	Measure Flow Sample Effluent
Thursday	8:00 a.m.		Measure Flow Sample Effluent	Measure Flow Sample for Complete Analysis
	7:00 p.m.	Load Basin		
	8:00 p.m.		Measure Flow Sample Effluent	Measure Flow Sample Effluent
Friday	8:00 a.m.			Measure Flow Sample Effluent
	7:00 p.m.		Load Basin	
	8:00 p.m.			Measure Flow Sample Effluent
Saturday	7:00 p.m.			Load Basin

TABLE 2. LOADING AND SAMPLING SCHEDULE

(FEBRUARY 1977 TO JUNE 1978)

Day	Time	Basin 1	Basin 2	Basin 3
Tuesday	8:00 a.m.	Load Basin Measure Flow Sample Effluent		
	9:00 a.m.	Measure Flow Sample Influent Sample Effluent		
	12 Noon	Measure Flow Sample Effluent		
	4:00 p.m.	Measure Flow Sample Effluent		
	8:00 p.m.	Measure Flow Sample Effluent		
Wednesday	8:00 a.m.	Measure Flow Sample for Complete Analysis	Load Basin Measure Flow Sample Effluent	
	9:00 a.m.		Measure Flow Sample Influent Sample Effluent	
	12 Noon		Measure Flow Sample Effluent	
	4:00 p.m.		Measure Flow Sample Effluent	
	8:00 p.m.	Measure Flow Sample Effluent	Measure Flow Sample Effluent	
Thursday	8:00 a.m.	Measure Flow Sample Effluent	Measure Flow Sample for Complete Analysis	Load Basin Measure Flow Sample Effluent
	9:00 a.m.			Measure Flow Sample Influent Sample Effluent
	12 Noon			Measure Flow Sample Effluent
	4:00 p.m.			Measure Flow Sample Effluent

(continued)

TABLE 2. (continued)

Day	Time	Basin 1	Basin 2	Basin 3
	8:00 p.m.	Measure Flow Sample Effluent	Measure Flow Sample Effluent	Measure Flow Sample Effluent
Friday	8:00 a.m.		Measure Flow Sample Effluent	Measure Flow Sample for Complete Analysis
	8:00 p.m.	Load Basin	Measure Flow Sample Effluent	Measure Flow Sample Effluent
Saturday	8:00 a.m.			Measure Flow Sample Effluent
	8:00 p.m.		Load Basin	Measure Flow Sample Effluent
Sunday	8:00 p.m.			Load Basin

nitrogen. The organic nitrogen procedure has been described in detail in Section 135 of Standard Methods (4).

Ammonium determinations were made with an Orion Research ammonium electrode Model 95-10. The instructions provided with the electrode were followed, with a new calibration curve prepared for each group of samples.

The nitrite determinations were initially made by following the method outlined in Section 420 of the 14th Edition of Standard Methods (4). However, after January, 1977 nitrite analyses were modified to facilitate the use of a Technicon Auto Analyzer II. The automated procedure was an adaptation of the diazotization method outlined in Standard Methods (4).

Similarly, nitrates were initially determined by the Brucine method outlined in Standard Methods (4). A Bausch and Lomb Spectronic 70 spectrophotometer was used for transmittance readings with new standards run for each set of nitrate samples. After January, 1977 an automated procedure was adapted using the Technicon Auto Analyzer II. Nitrates were determined by the automated copper-cadmium reduction method described in Section 605 of Standard Methods (4).

Total phosphorus determinations were made by first acidifying the samples, and then digesting them by the Persulfate Digestion Method outlined in Section 223C-III of Standard Methods (4). Color development was accomplished by the Stannous Chloride method outlined in Section 223E of Standard Methods (4). The transmittance was measured with a Bausch and Lomb Spectronic 70 spectrophotometer at a wave length of 690 nanometers (nm).

Total and fecal coliforms were reported as coliforms per 100 ml, and were determined by use of the membrane filter technique described in Section 408 of Standard Methods (4).

Both the suspended solids and total dissolved solids were run using some variations to the process as described in Standard Methods (4). Suspended solids were determined by the modified process described by Harada and his co-workers (5). Total solids were determined by the following procedure: first, evaporating dishes were dried at 103°C for one hour, then dessicated for one-half hour, and weighed for tare weight. One hundred milliliters of sample were then evaporated in the dishes on a hot plate. Following this, they were dessicated for one-half hour and weighed.

The chemical oxygen demand (COD) analyses were determined by the method outlined in Section 220 of Standard Methods (4). The soluble COD was determined by following the indicated procedure after filtration of the sample through a 450 nm millipore filter.

During the loading cycle of November-December 1977, a special set of heavy metal samples was taken from the secondary effluent entering the beds, and the bed product water 24 hours after loading. These samples were stored in new polyethylene bottles which had been acid washed three times in 1:1 HNO_3, 1:1 HCl, and distilled water. The samples were subsequently subjected to the sample treatment indicated in Figure 7.

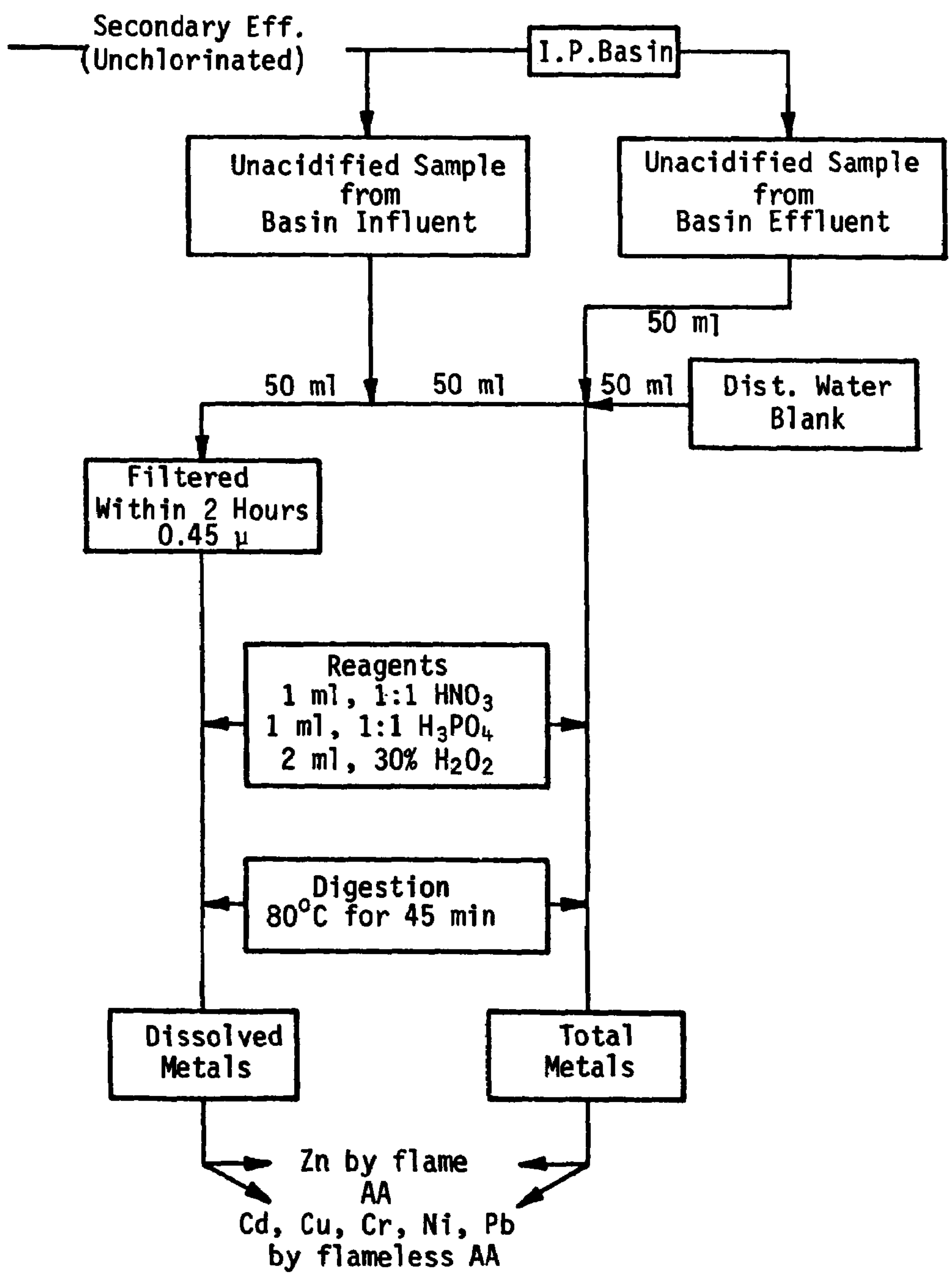


Figure 7. Analytical procedure for heavy metals.

As indicated in this figure, the concentrations of cadmium, copper, chromium, nickel, and lead were determined by flameless atomic absorption. Zinc was determined by flame atomic absorption since it was impossible to use the flameless method for measurement of zinc in these samples because of the very high sensitivity of the instrument and the excessive amounts of background zinc. The use of flameless atomic absorption allowed concentration of the samples to be avoided, as well as the vigorous digestion necessary with flame atomic absorption.

SECTION 6

HYDRAULIC CHARACTERISTICS OF THE SITE

GROUND WATER PROFILE

Prior to basin construction, the ground water characteristics of the proposed location were measured and recorded. From these measurements, the ground water table contour of Figure 8 was developed to define the conditions in May, 1975. From this figure it can be seen that the general pattern of ground water flow was away from Boulder Creek and towards the north and west sides of Basin 3. The flow was also directed towards the west end, and the west and south sides of Basins 2 and 1, respectively. For comparison, Figure 9 shows the ground water levels at selected sites surrounding the basins on August 10, 1976, which was after several months of basin operation. The water levels at that time were actually somewhat lower than those prior to initiation of basin operation, reflecting the change in ground water table with the time of the year.

Within the basins, the ground water elevations were lowered by the underdrain system to approximately 2.4 meters (8 feet) from the surface throughout the area of the infiltration-percolation system. These ground water elevations were monitored on a time profile basis during a typical week of loading, with the elevations reported in Table 3. Mounding occurred to a small degree, but serious ground water mounding was prevented by the underdrain system. The maximum mounding condition is indicated in Figure 10, which shows a cross-section of the area from Boulder Creek to the infiltration-percolation system.

With the steep gradient of ground water elevations across the peripheral clay dike, it was expected that some flow would occur. As a result, prior to application of any wastewater to the beds, the base ground water flow discharged from each basin was measured. These flows are indicated in Table 4, and show that the two outside basins had significant flow across the dike, while the middle basin received essentially no ground water. Under the minimum loading condition on Bed 3, the ground water constituted about 20% of the average underdrain flow following basin loading. However, with the higher loading following modification of the basins, the ground water contribution to the underdrain flow was reduced to less than 5% of the total underdrain flow in all the basins.

BASIN HYDRAULICS

Following flooding of each basin, the wastewater percolated downward

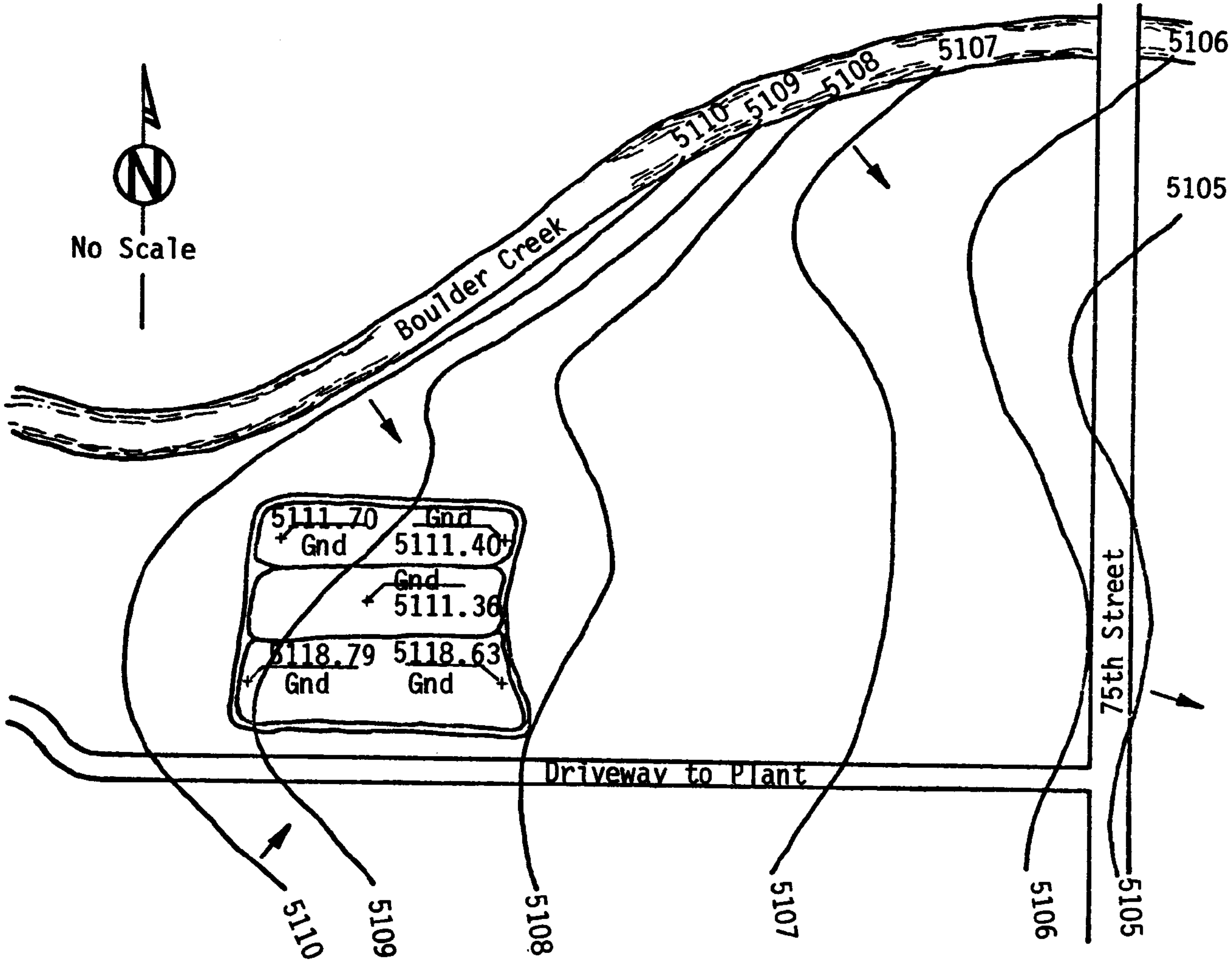


Figure 8. Ground water contours and flow directions (feet) (May, 1975).

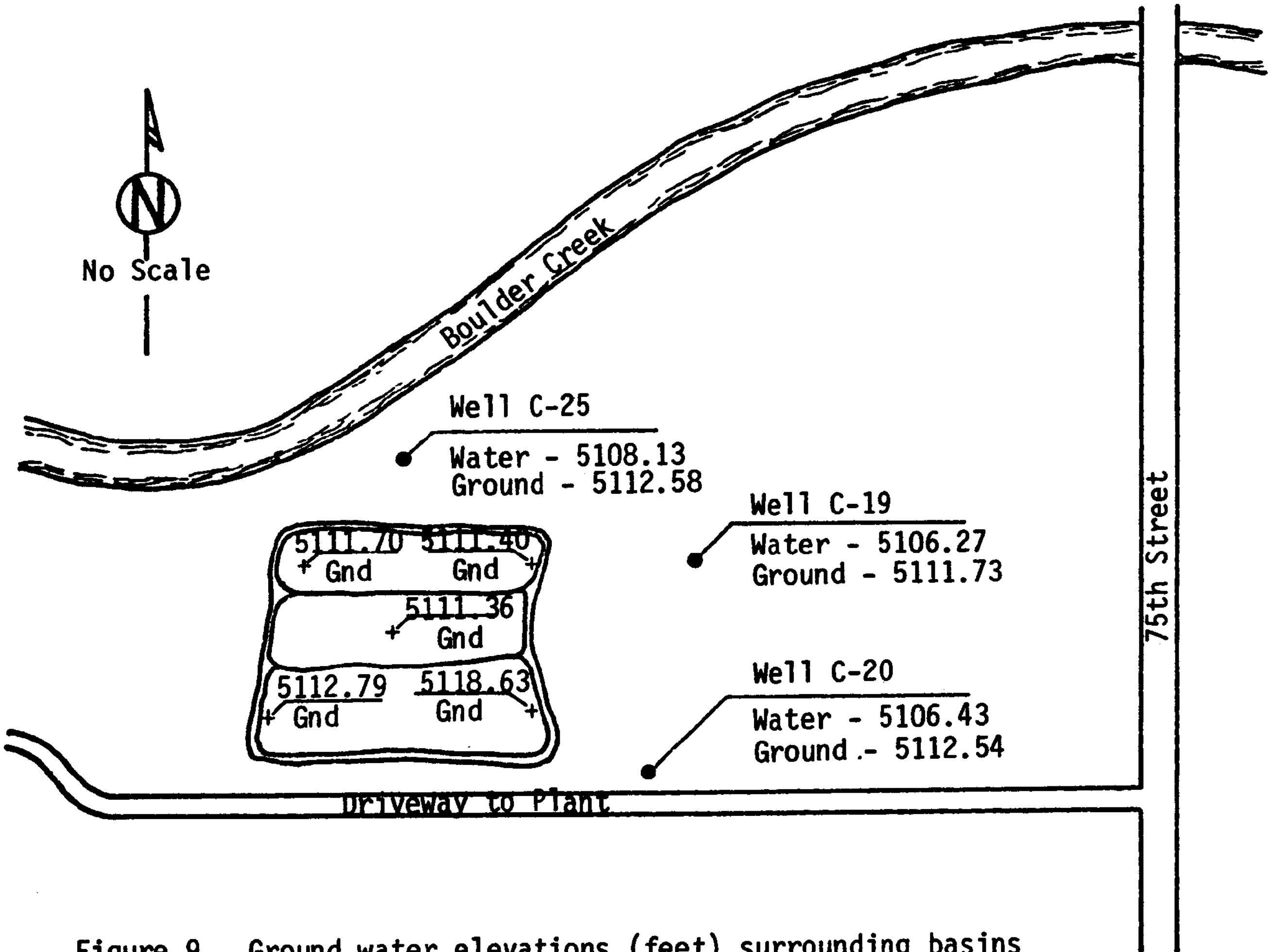


Figure 9. Ground water elevations (feet) surrounding basins (August, 1976)

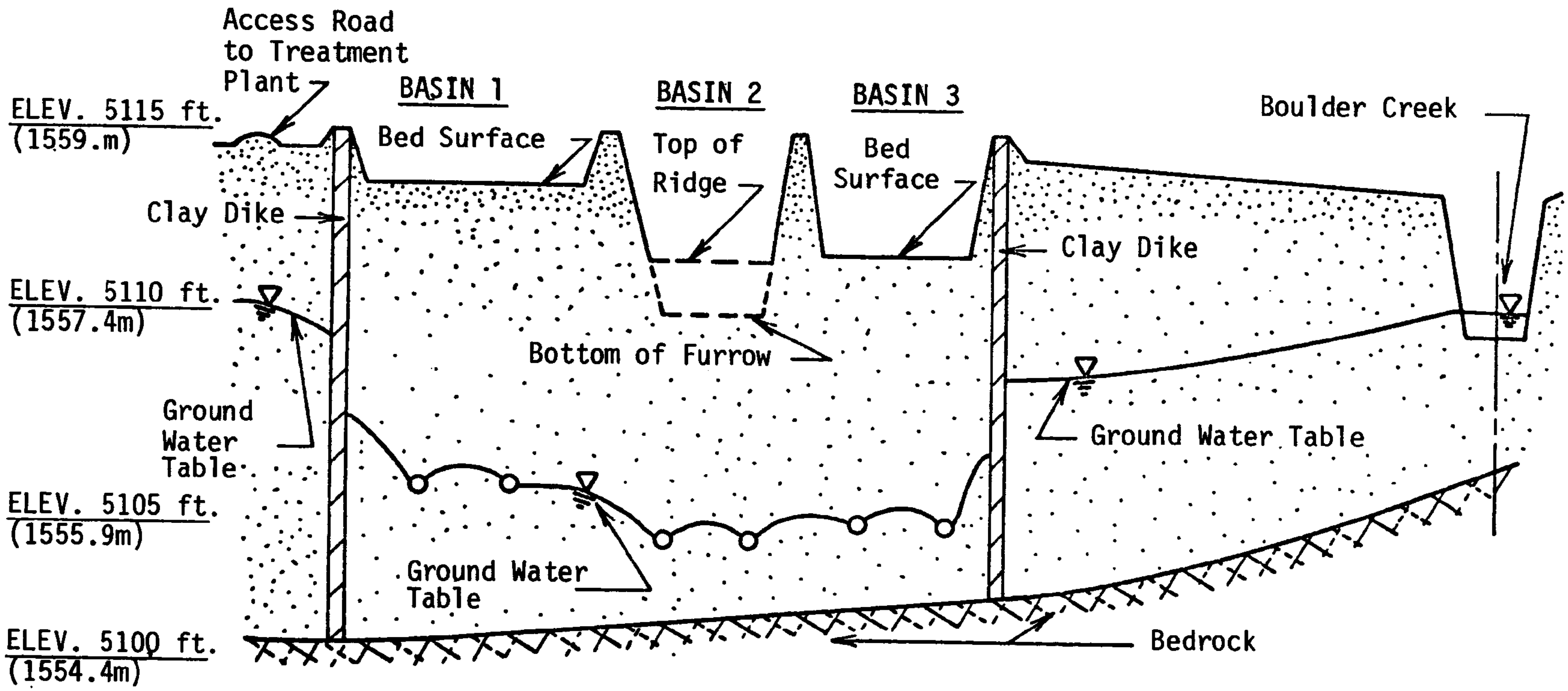


Figure 10. Ground water mounding pattern within basins.

TABLE 3. GROUND WATER ELEVATIONS (feet)

Bed and Time After Loading	Well Point Location																
	1-1	1-2	1-3	2-1	2-2	2-3	2-4	3-1	3-2	3-3	3-4	3-5	3-7	3-8	A	B	C
	Well Point Elevation																
	5116.27	5116.44	5116.16	5111.54	5115.02	5115.34	5115.14	5114.94	5115.30	5116.06	5116.43	5116.06	5115.78	5115.88	5114.78	5114.02	5114.37
Bed 1 - 1@ 0 hrs	5104.60	5105.69	5104.99	5103.54	5104.19	5103.52	5104.31	5103.19	5103.97	5104.39	5107.43	5104.14	5106.70	5108.21	5107.78	5106.77	5107.95
1@ 8	5104.52	5105.86	5104.99	5103.29	5104.10	5103.44	5104.14	5103.44	5104.02	5104.64	5107.43	5104.06	5105.89	5108.13	5107.78	5106.85	5107.70
1@12	5104.69	5106.44	5105.24	5103.37	5103.94	5103.44	5103.97	5103.44	-	-	-	-	-	-	-	-	-
1@24	5104.69	5106.44	5105.24	5103.37	5103.94	5103.44	5103.97	5103.44	5103.52	5104.56	5107.26	5104.14	5105.48	5107.88	5108.03	5107.10	5108.20
1@32	5104.52	5106.44	5105.24	5105.96	5104.85	5104.35	5104.72	5103.44	5103.86	5104.39	5107.43	5104.06	5105.89	5108.38	5108.03	5107.02	5108.37
1@48	5104.77	5106.44	5105.24	5104.46	5104.27	5103.77	5104.22	5103.44	5103.97	5104.56	5107.51	5104.23	5105.73	5108.21	5107.86	5107.10	5108.04
1@72	5104.81	5106.44	5105.33	5104.04	5104.35	5103.69	5104.31	5103.86	5103.94	5105.39	5107.76	5104.48	5106.31	5108.33	5108.03	5107.10	5108.20
Bed 2 - 2@ 0 hrs	5104.69	5106.44	5105.24	5103.37	5103.94	5103.44	5103.97	5103.44	5103.52	5104.56	5107.26	5104.14	5105.48	5107.88	5108.03	5107.10	5108.20
2@ 8	5104.52	5106.44	5105.24	5105.96	5104.35	5104.35	5104.72	5103.44	5103.86	5104.39	5107.43	5104.06	5105.89	5108.38	5108.03	5107.02	5108.37
2@24	5104.77	5106.44	5105.24	5104.46	5104.27	5103.77	5104.22	5103.44	5103.97	5104.56	5107.51	5104.23	5105.73	5108.21	5103.86	5107.10	5108.04
2@32	5104.77	5106.44	5105.33	5104.29	5104.02	5103.52	5104.22	5103.61	5104.19	5105.31	5107.43	5104.39	5106.39	5108.30	5107.70	5106.94	5108.37
2@48	5104.81	5106.44	5105.33	5104.04	5104.35	5103.69	5194.31	5103.86	5103.94	5105.39	5107.76	5104.48	5106.31	5108.33	5108.03	5107.10	5108.20
Bed 3 - 3@ 0 hrs	5104.77	5106.44	5105.24	5104.46	5104.27	5103.77	5104.22	5103.44	5103.97	5104.56	5107.51	5104.23	5105.73	5108.21	5107.86	5107.10	5108.04
3@ 8	5104.77	5106.44	5105.33	5104.29	5104.02	5103.52	5104.22	5103.61	5104.19	5105.31	5107.43	5104.39	5106.39	5108.30	5107.70	5106.94	5108.37
3@24	5104.81	5106.44	5105.33	5104.04	5104.35	5103.69	5104.31	5103.86	5103.94	5105.39	5107.76	5104.48	5106.31	5108.39	5108.03	5107.10	5108.20
3@72	5104.60	5105.61	5105.16	5105.04	5104.19	5103.52	5104.31	5103.87	5103.77	5104.73	5107.68	5104.27	5105.73	5108.21	5107.78	5107.02	5107.87

ft x 0.305 = m

TABLE 4. BASE FLOWS

Basin	Base Flow (cfs)	(m^3/sec)
1	0.013	3.68×10^{-4}
2	0	0
3	0.025	7.08×10^{-4}

through the soil, was collected by underdrains, and was pumped to the surface and discharged into Boulder Creek. With each wastewater application, the discharge flow of Basin 1 exhibited a rapid increase to a peak within 12 hours after loading. This was followed by a gradual decline in flow as is shown by a representative discharge hydrograph in Figure 11. Prior to basin modification, a representative hydrograph for Basin 2 exhibited a rise and fall similar to that shown in Figure 12. While the peak occurred at about the same time after loading, the curve was much broader, indicating a lower percolation rate, or slower water mass flow through the soil. Similarly, Basin 3 had comparable hydraulic characteristics, but with a dampened peak occurring at about 15 hours as shown in Figure 13. To facilitate comparison of the profiles for each bed, the flows from Basins 2 and 3 were normalized to that of Basin 1 by multiplying their flows by the fraction of loading time for Basin 1 divided by the loading time for each of the other basins. This was done to correct the hydrographs for the different amounts of water applied to each basin. With this normalization, the hydraulic dampening of Basins 2 and 3 was more apparent, as can be seen in Figure 14. The lower infiltration rates which are suggested by these curves for Basins 2 and 3 were also indicated by ponding of the water in these basins for several days after loading. It can be speculated that this extended ponding time further lowered the infiltration rates because of increased algae growth, and the accompanying fouling of the surface with suspended solids.

In addition to the differences in infiltration rates which have been noted for the three basins, the infiltration rates of all the basins gradually declined during the first few months of operation. This trend is indicated in Figure 15, which is a plot of the peak flow of the underdrain discharge during each of the first several loading cycles. The reduced peak flow rates were characteristic of similar reductions which occurred in the acceptable application rates during the first months of operation. The initial loading rates in May, 1976 were equivalent to 48.5 m/yr (159 ft/yr) on Bed 1, 36.0 m/yr (118 ft/yr) on Bed 2, and 53.6 m/yr (176 ft/yr) on Bed 3. After two weeks of loading at these rates, ponding conditions developed on all three beds. As a result, all of the beds were allowed to dry for one week and then scarified. Subsequently, the loading rates were reduced to 27.4 m/yr (90 ft/yr), 12.2 m/yr (40 ft/yr) and 15.2 m/yr (50 ft/yr) for beds 1, 2, and 3, respectively. Bed 1 functioned acceptably at this loading rate, and the rate was increased to 30.5 m/yr (100 ft/yr) in October,

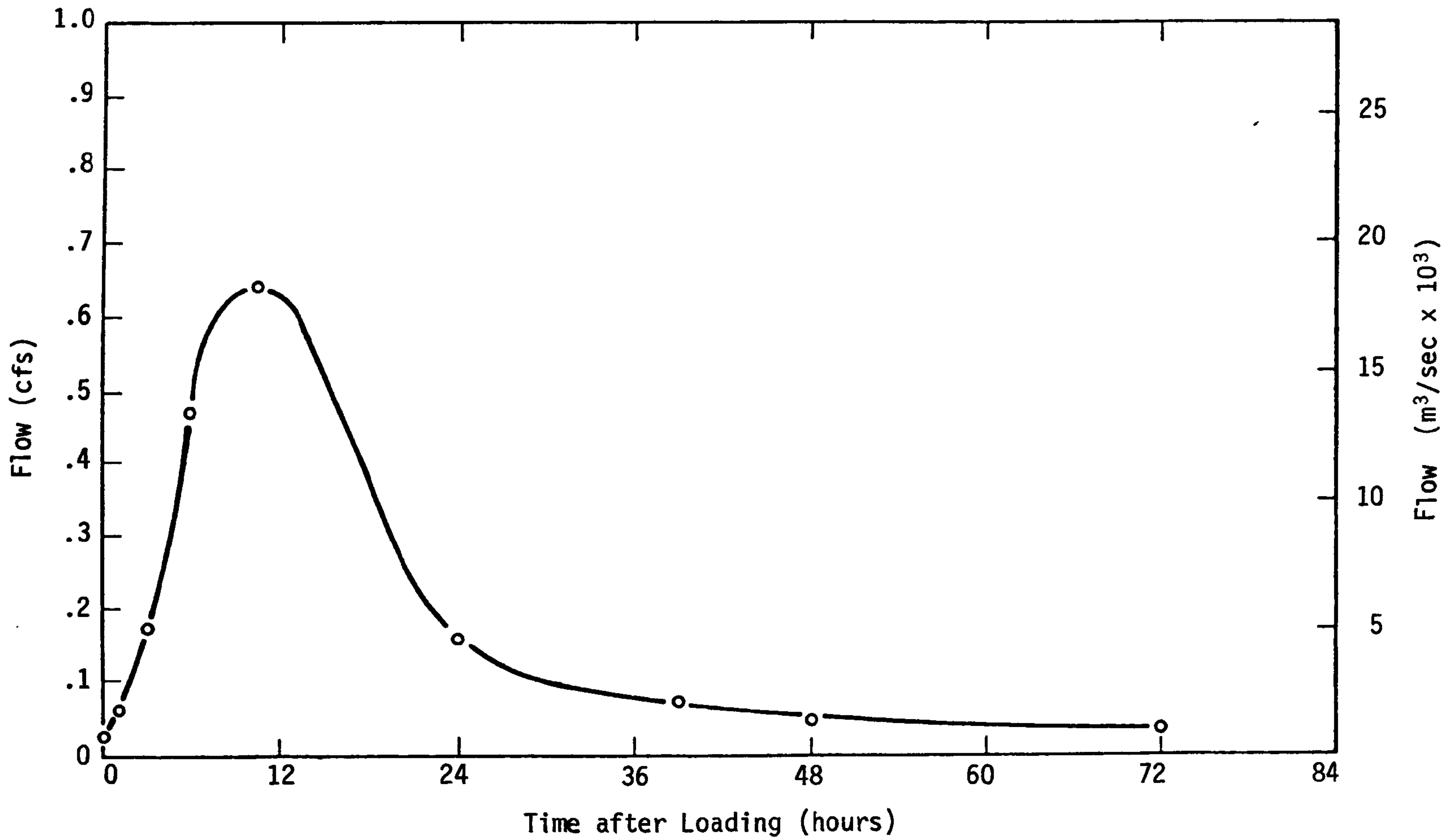


Figure 11. Typical discharge hydrograph for Basin 1.

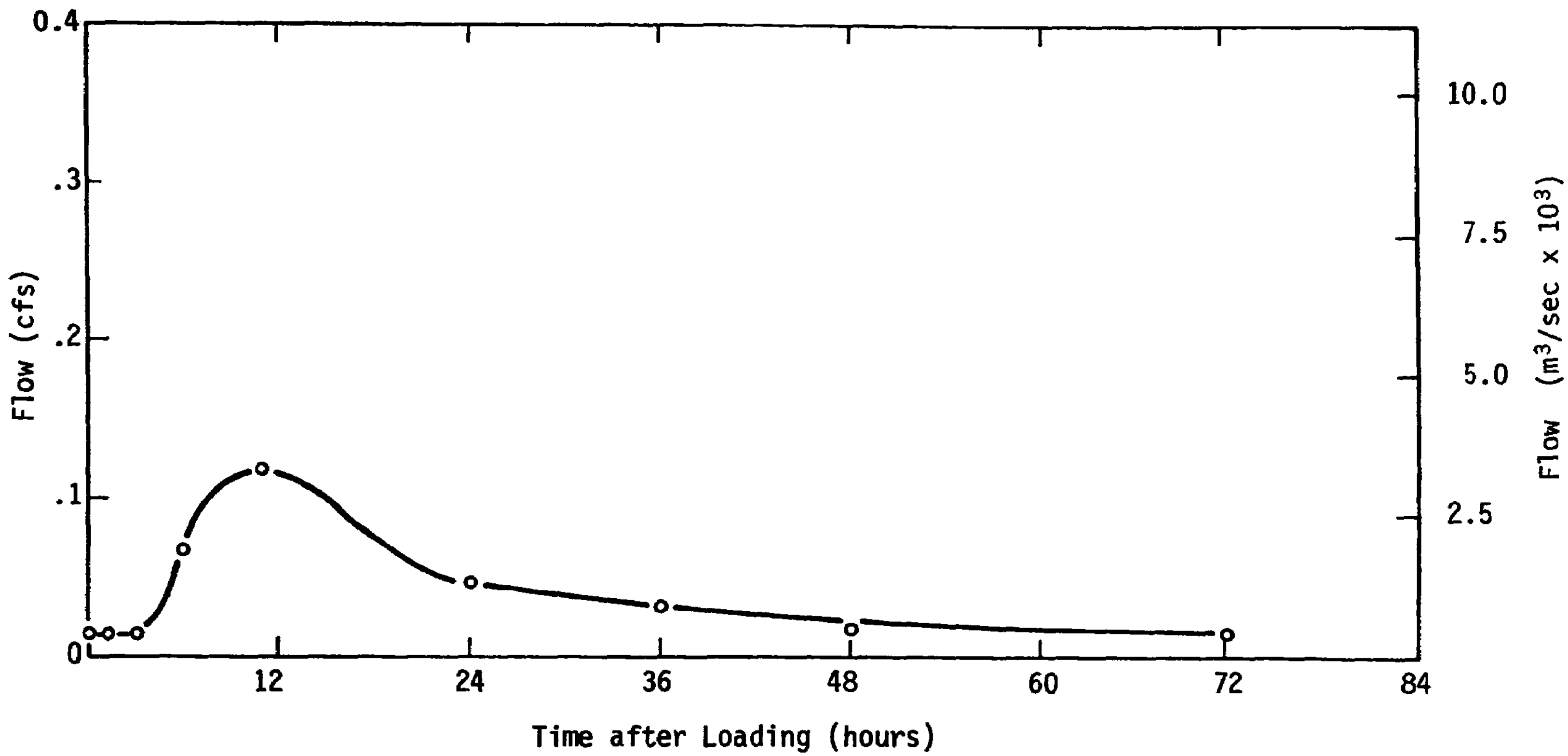


Figure 12. Typical discharge hydrograph for Basin 2 prior to modification.

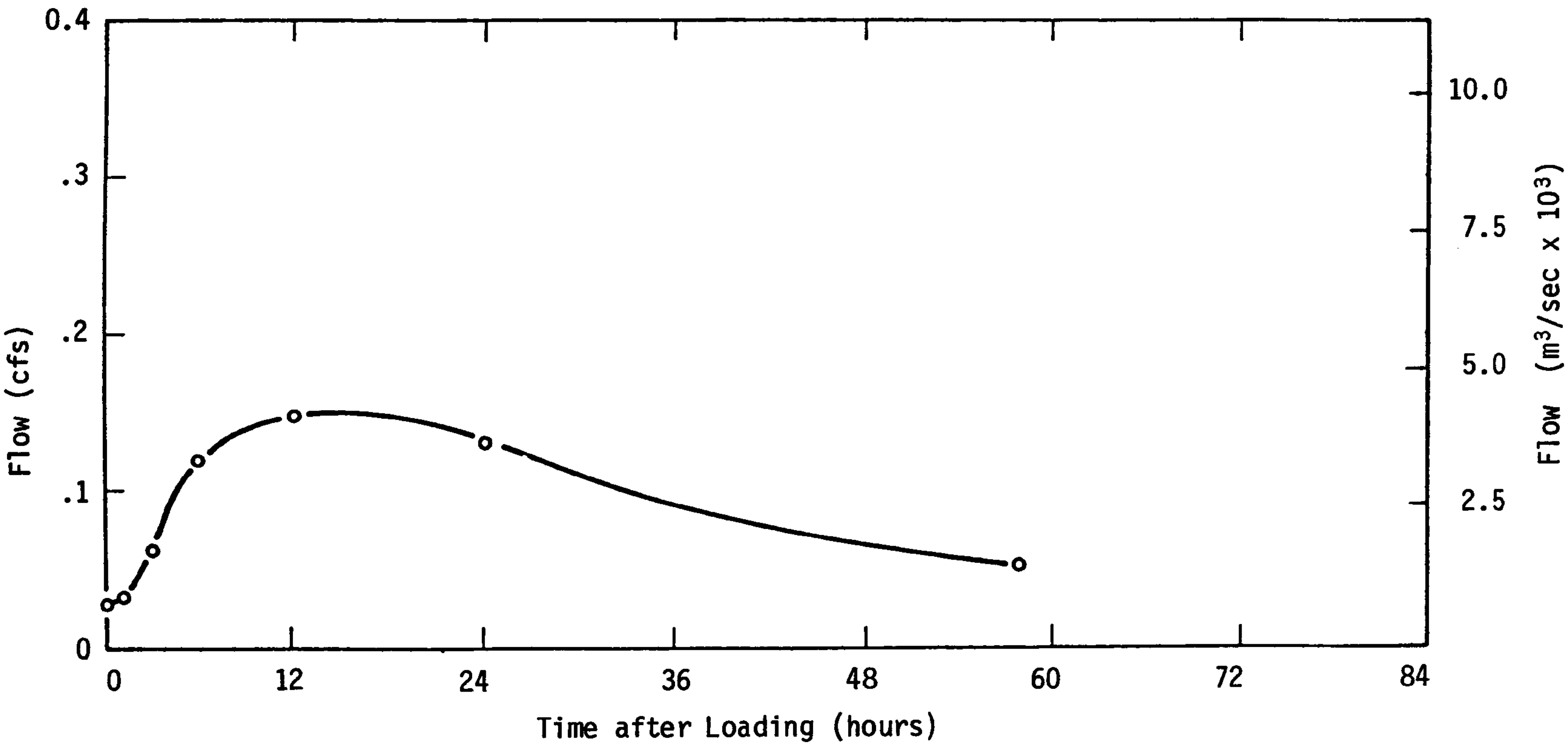


Figure 13. Typical discharge hydrograph for Basin 3 prior to modification.

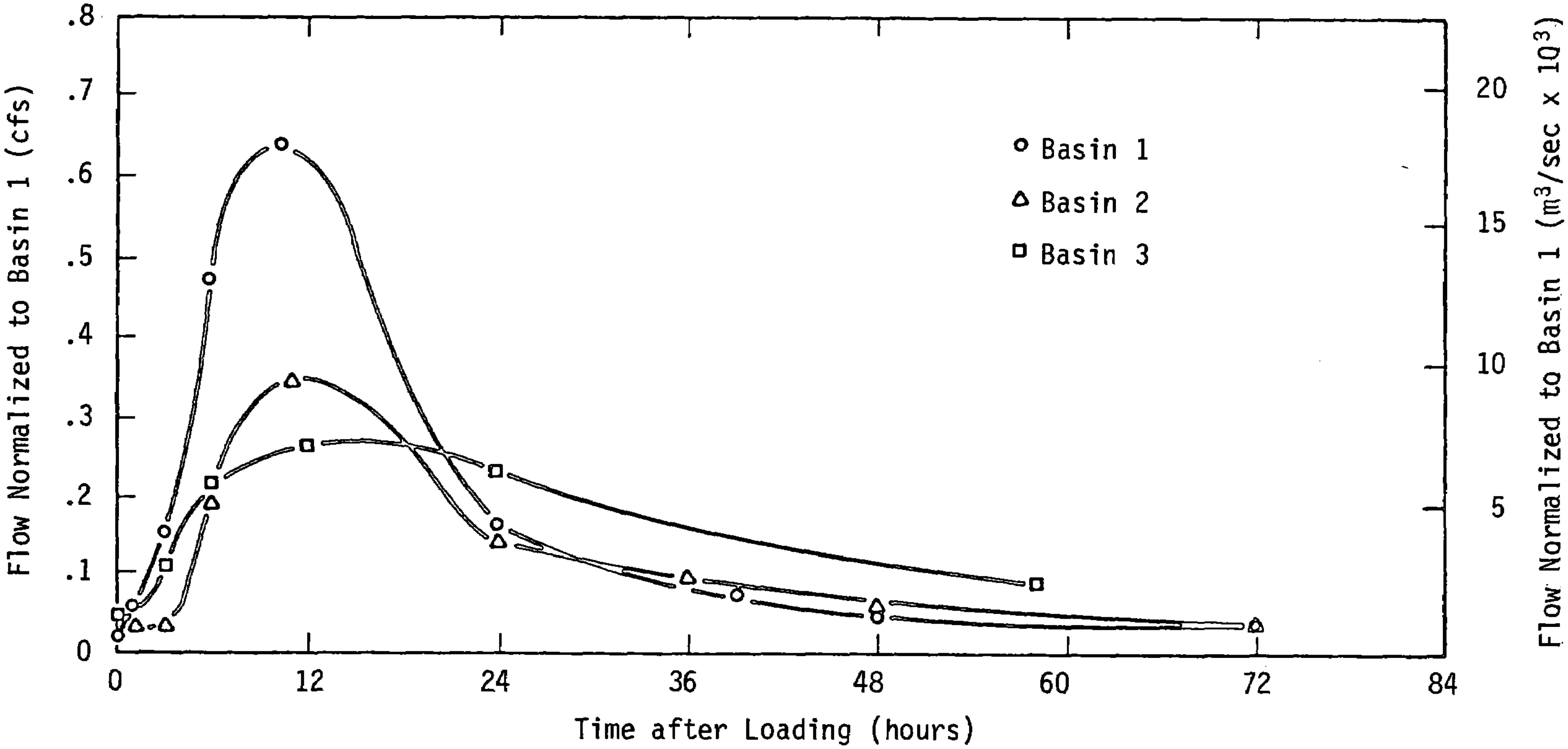


Figure 14. Normalized discharge hydrographs prior to modification.

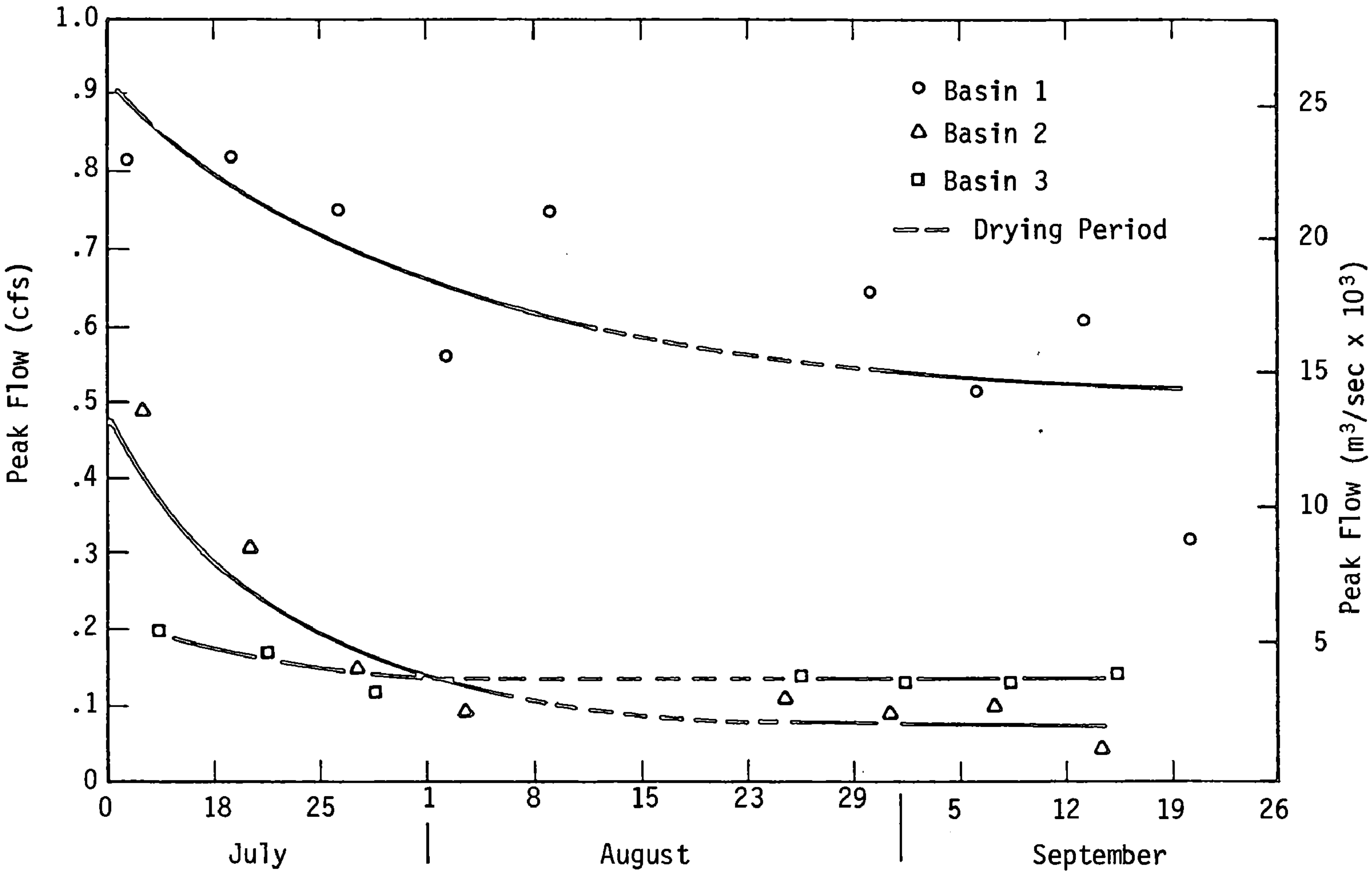


Figure 15. Peak discharge flows prior to basin modification.

1976 without causing any hydraulic problems. However, Beds 2 and 3 continued to demonstrate hydraulic difficulties, and by December, 1976 their loading rates had been effectively reduced to 3.96 m/yr (13 ft/yr) and 5.79 m/yr (19 ft/yr), respectively.

INFILTROMETER STUDIES

It was theorized that the surface layer of soil was restricting the wastewater infiltration and, thus, was responsible for the poor hydraulic performance of Basins 2 and 3. The Niwot Series soil of the area has been characterized by the Soil Conservation Service as consisting of 0-0.30 meter (0-12 inch) depth loams to clay loams, and 0.30-1.52 meter (12-60 inch) depth coarse sand (1). If the surface soils of Beds 2 and 3 were indeed clay loams, it was suspected that the theory would be correct. As a result, infiltrometer tests were performed on all three beds to determine: (1) the existing infiltration rates on the surface, (2) the type and depth of the overlying soil layer, and (3) the infiltration rate of the underlying soil and its soil classification. A double ring infiltrometer was used for performance of these tests. The inner ring was 0.20 meters (8 inches) in diameter and the outer ring was 0.38 meters (15 inches) in diameter, with the length of both at 0.36 meters (14 inches). The rings were concentric, and were connected by metal plates welded between the two rings.

Test sites were located within each of the basins which had no significant surface disturbance, and which had soil textures representative of the area. At these sites, the infiltrometer was pressed into the soil to a depth of approximately 0.15 meters (6 inches). Installation was performed with care to minimize the soil disturbance around the cylinder. The area between the inner and outer ring provided a buffer pond which served to minimize the radial flow of water away from the inner ring. This area was filled first with water and kept at a constant level throughout the testing. The water used in the testing was unchlorinated secondary effluent from the 75th Street Trickling Filter Plant; the same wastewater which was applied to the basins. After the outer ring was filled, the inner ring was filled to a 0.15 meter (6 inch) depth with care taken to minimize the disturbance of the soil surface. The water level in the inner ring was measured at the start of testing and at time intervals ranging from 1 minute to 30 minutes, depending upon the infiltration rate. Sites with high infiltration rates were tested several times, with the last test used to determine the average infiltration rate. The average infiltration rate was determined by dividing the total drop in the water level by the corresponding elapsed time. These infiltration rates were reported in centimeters per hour. The above procedure was patterned after similar procedures described by Haise and Johnson (6,7).

Infiltration tests were performed at each of the locations indicated by a dot in Figure 16. The lettered dots represent sites tested prior to surface modification of the beds, and the numbered dots represent sites tested after some soil was removed from the surface of Beds 2 and 3. Surface infiltration tests were performed in Bed 1 at the locations identified by points A, D, and F in Figure 16. These tests yielded infiltration rates ranging from 0.58-7.13 cm/hr (0.23-2.81 in/hr), with an average of 4.42 cm/hr (1.74 in/hr).

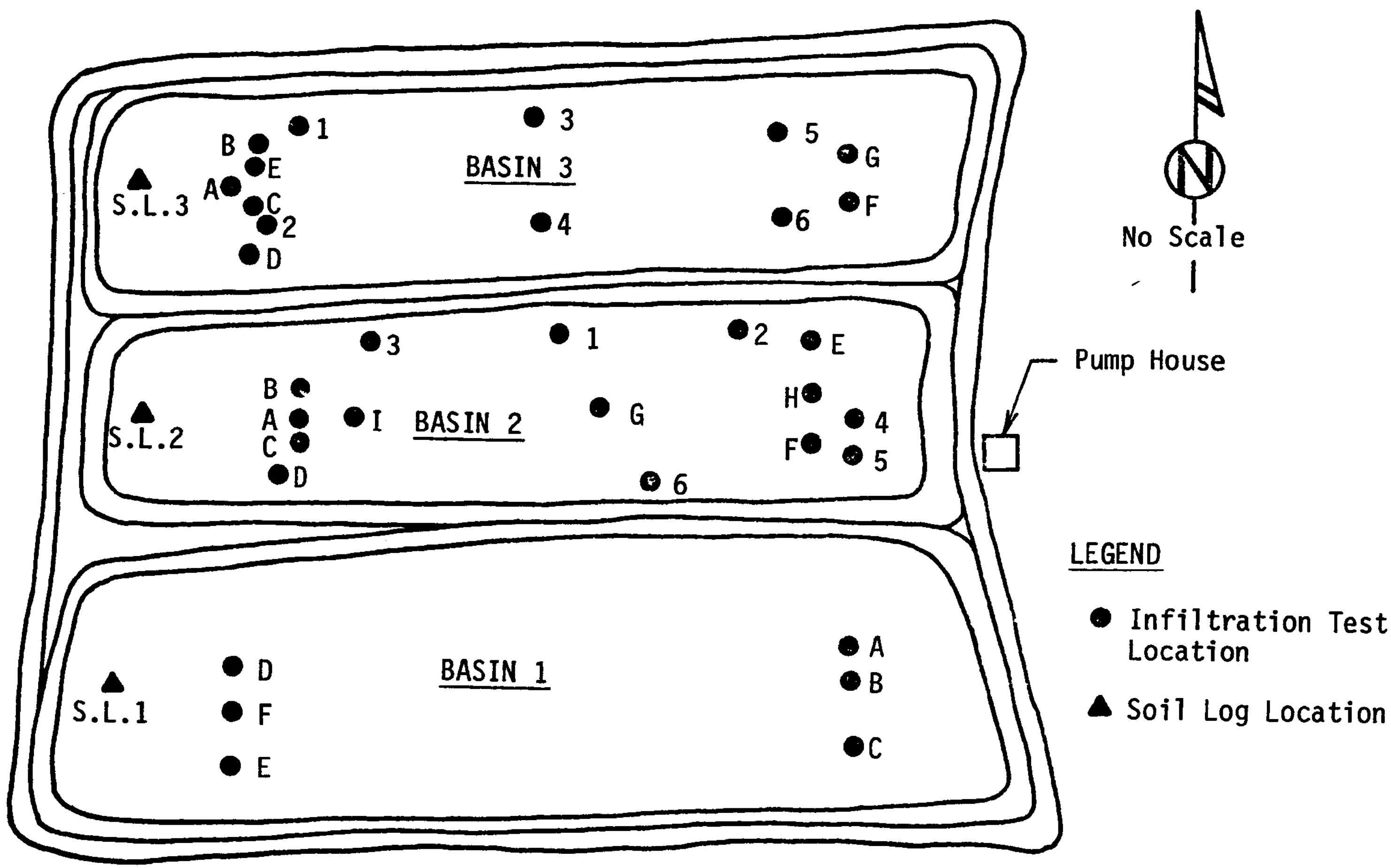


Figure 16. Infiltrometer test locations.

Details of the rates at each site have been included in Table 5. The soil within this basin was a silty loam which extended from the surface to 0.38-0.66 meters (15-26 inches) from the surface. The underlying soil was well rounded gravel mixed with sand and silt-sand loam. Infiltration tests at a 0.38 meter (15 inch) depth at location C and a 0.66 meter (26 inch) depth at location E yielded rates of 110 cm/hr (43.5 in/hr) and 88 cm/hr (34.5 in/hr), respectively. However, the presence of gravel at these depths prevented the desired 0.15 meter (6 inch) penetration of the double ring infiltrometer, which may have contributed to the very high observed rates. A soil log 2.44 meters (8 feet) in depth revealed the same gravel-sand soil mixture from a 0.51 meter (20 inch) depth to the invert of the underdrain system at approximately 2.44 meters (8 feet). It can be seen in Figure 17 that a significant increase in the measured infiltration rate was noted once the surface soil layer of silty loam was penetrated. Because of the relatively homogeneous nature of the soil below this point, it was expected that the infiltration rate would become relatively constant at depths of greater than 0.61 meters (2 feet). Based on the favorable infiltrometer results, and the successful operation at a loading rate of 30.5 m/yr (100 ft/yr), it was decided to continue loading Bed 1 at 30.5 m/yr (100 ft/yr) with no physical alteration of the bed surface.

Infiltration tests on the surface of Bed 2 at points A, D, and E yielded infiltration rates ranging from 0.10-1.41 cm/hr (0.04-0.16 in/hr), with an average rate of only 0.23 cm/hr (0.09 in/hr). The soil on the surface of this bed appeared to be a silty loam with some heavy clay. At depths of 0.46 meters (18 inches) and 0.76 meters (30 inches), the measured infiltration rates increased to 0.89 cm/hr (0.35 in/hr) and 0.99 cm/hr (0.39 in/hr), respectively. These measurements were made at points C and B. The soil texture at each location was similar to the silty-clay loam of the surface soil, but also contained some sand. At point H and a depth of 1.07 meters (42 inches), the soil was a sandy loam, and an infiltration rate of 22.6 cm/hr (8.89 in/hr) was observed. A soil log on Bed 2 showed that gravel and a sandy loam existed from approximately 0.91 meters (3 feet) to the invert of the underdrain system at about 2.44 meters (8 feet). A plot of the infiltration rate as a function of depth is shown in Figure 18. From this figure it was determined that the top 0.91-1.07 meters (3-3.5 feet) of overburden were unsuitable for an infiltration-percolation system and should be removed.

As has been indicated, the method which was ultimately utilized to physically alter Bed 2 consisted of stripping the top 0.61 meters (2 feet) of soil and then constructing a ridge and furrow system, as was outlined in Section 4. Following bed modification, six infiltration tests were performed on the bottom of the 0.46 meter (18 inch) deep furrows, at the locations numbered 1 through 6 in Figure 16. The results of these tests yielded infiltration rates which varied from 3.2 cm/hr (1.25 in/hr) to 27.9 cm/hr (11.0 in/hr), with an average of 12.1 cm/hr (4.76 in/hr). The predominant soil type found in the bottom of the furrows was a sandy loam. Loading on Bed 2 following the surface modification was limited to the amount of wastewater which filled the furrows without overflowing onto the ridges. These loadings were applied on a bi-weekly basis. This practice

TABLE 5. SUMMARY OF INFILTROMETER TEST RESULTS

Bed No.	Test Location	Depth from Original Surface (meters)	Infiltration Rate (cm/hr)
1	A	Surface	5.5
Soil Log 1:	D	Surface	0.6
0-0.51 m - silty-loam	F	Surface	7.1
0.51-2.42 m - sand and gravel	B	0.051	0
	C	0.381	110.5
	E	0.660	87.6
2	A	Surface	0.20
Soil Log 2:	D	Surface	0.10
0-0.91 m - silty-loam w/heavy clay	E	Surface	0.41
0.91-2.43 m - gravel and sandy loam	F	0.051	0.13
	G	0.051	0.30
	C	0.457	0.89
	B	0.762	0.99
	I	0.864	-
	H	1.067	22.6
	1	1.067	27.9
	2	1.067	41.9
	3	1.067	9.1
	4	1.067	3.5
	5	1.067	16.7
	6	1.067	3.2
3	A	Surface	2.0
Soil Log 3:	F	Surface	0.53
0-0.46 m - silty-sandy loam w/clay	C	0.254	0.97
0.46-2.44 m - sandy loam	D	0.254	4.2
	G	0.457	52.0
	E	0.330	18.1
	2	0.457	4.7
	3	0.457	3.0
	4	0.457	1.3
	5	0.457	3.5
	6	0.457	15.2
	1	0.559	11.4
	B	0.610	38.1

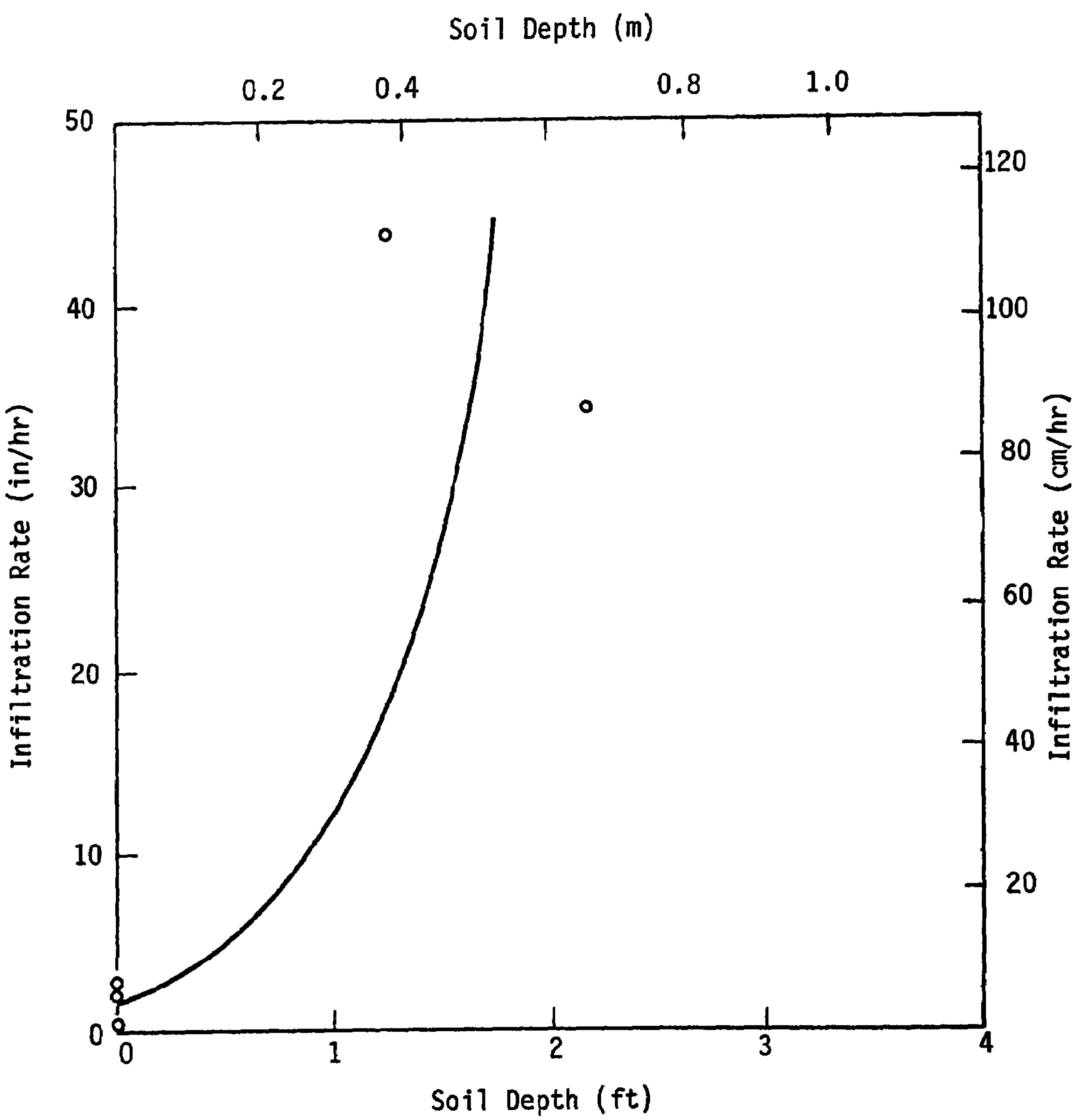


Figure 17. Infiltrometer test results for Basin 1.

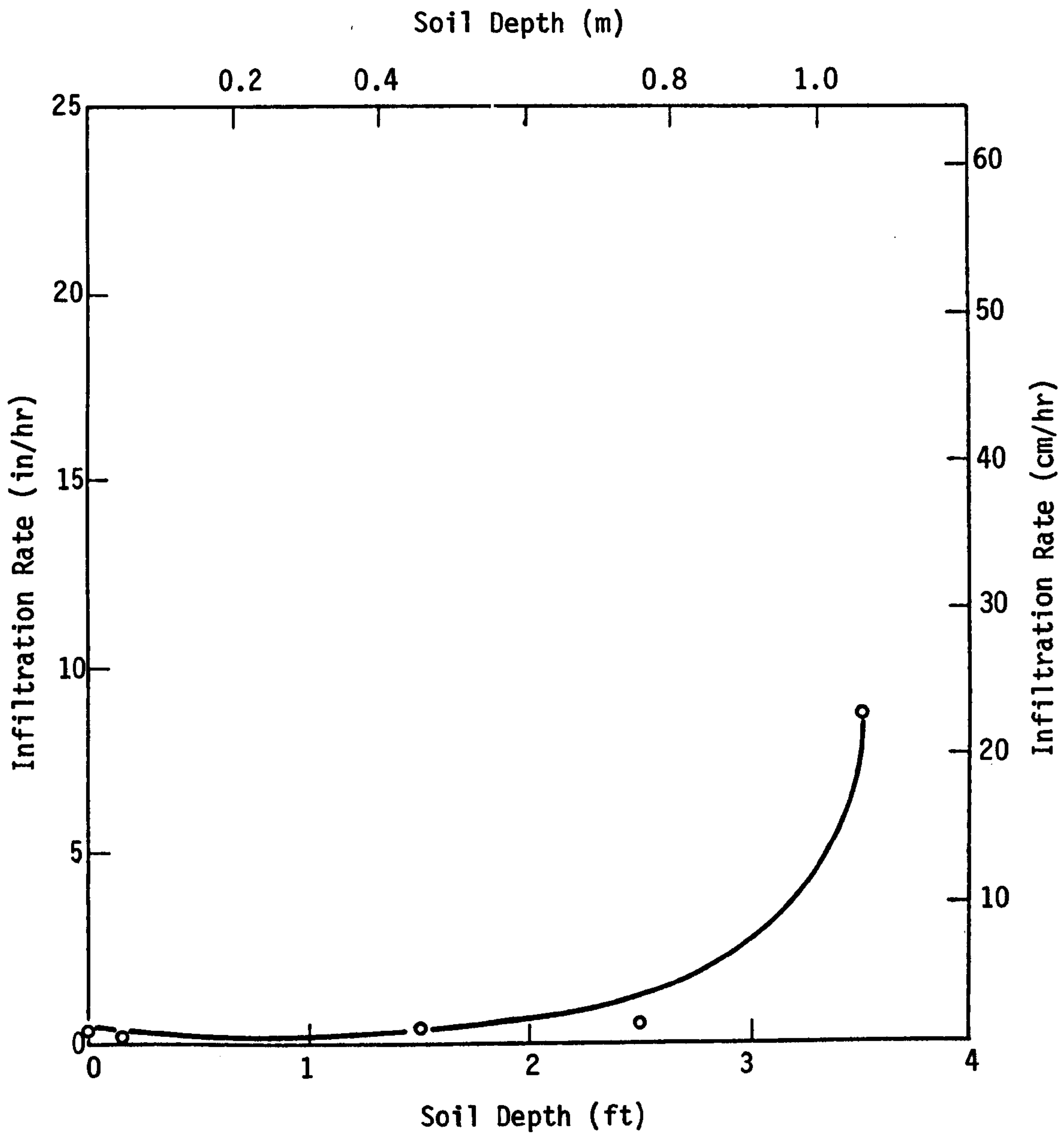


Figure 18. Infiltrometer test data on Basin 2 prior to modification.

resulted in an effective loading rate of 41.2 m/yr (135 ft/yr) over the furrow bottom area, or 12.8 m/yr (42 ft/yr) over the entire area of Bed 2. This proved to be a successful operating level.

Surface infiltration tests performed on Bed 3 at locations A and F resulted in infiltration rates of 2.0 cm/hr (0.78 in/hr) and 0.53 cm/hr (0.21 in/hr), respectively. Location A consisted of silty-sandy loam, while location F consisted of a clay-silt loam. At a depth of 0.25 meters (10 inches), an average infiltration rate of 2.59 cm/hr (1.02 in/hr) was observed, and the soil was predominantly a silty-sandy loam. These tests were made at points C and D. Infiltration tests performed at sites E and G were at a depth of 0.46 meters (18 inches), and yielded an average rate of 35.0 cm/hr (13.8 in/hr). At the 0.61 meter (24 inch) depth, an infiltration rate of 38.1 cm/hr (15.0 in/hr) was measured at point B, with the soil consisting primarily of a sandy loam containing some clay. Figure 19 summarizes these results graphically as a plot of the variation of infiltration rate with soil depth.

In order to improve the hydraulic performance of Basin 3, it was decided that the top 0.46 meters (1.5 feet) of tight soil should be removed. The stripping operation was performed without developing a ridge and furrow system as was constructed in Bed 2. Following the stripping, surface infiltration tests were performed at each of five representative locations, points 2-6, within Bed 3. The infiltration rates measured at these sites yielded an average infiltration rate of 5.6 cm/hr (2.19 in/hr). The soil at this depth was predominantly a sandy loam with varying amounts of clay also present. The loading rate of 48.8 m/yr (160 ft/yr) which was applied following bed modification was determined by an empirical method used for estimating loading rates for infiltration-percolation systems (Personal communication, R.E. Thomas). The loading rate was established at ten percent of the theoretical loading for one year at the average infiltration rate of 5.6 cm/hr (2.19 in/hr) as shown below:

$$\text{Loading rate} = \underline{0.1(2.19\ \text{in/hr})\ (8760\ \text{hr/yr})}.$$

$$\text{Loading rate} = 160\ \text{ft/yr} = 48.8\ \text{m/yr}$$

When loading was resumed, Bed 3 was erroneously loaded at a rate of 57.3 m/yr (188 ft/hr) for 3 weeks because of a failure to account for the reduced surface area developed as a result of incorporating a side slope into the excavation. At this loading rate, the bed failed to drain between successive loading cycles. When the loading was reduced to 48.8 m/yr (160 ft/yr) on the actual area, the basin operated in a successful manner.

Following the physical alteration of Beds 2 and 3, infiltration rates of the system were recorded during the operation of all three beds. This was done by measuring the change in water level with time after inundation of each of the beds. The average infiltration rates were determined by dividing the decline in water level by the elapsed time. A summary of these infiltration rates is shown in Table 6 for the three beds. The average

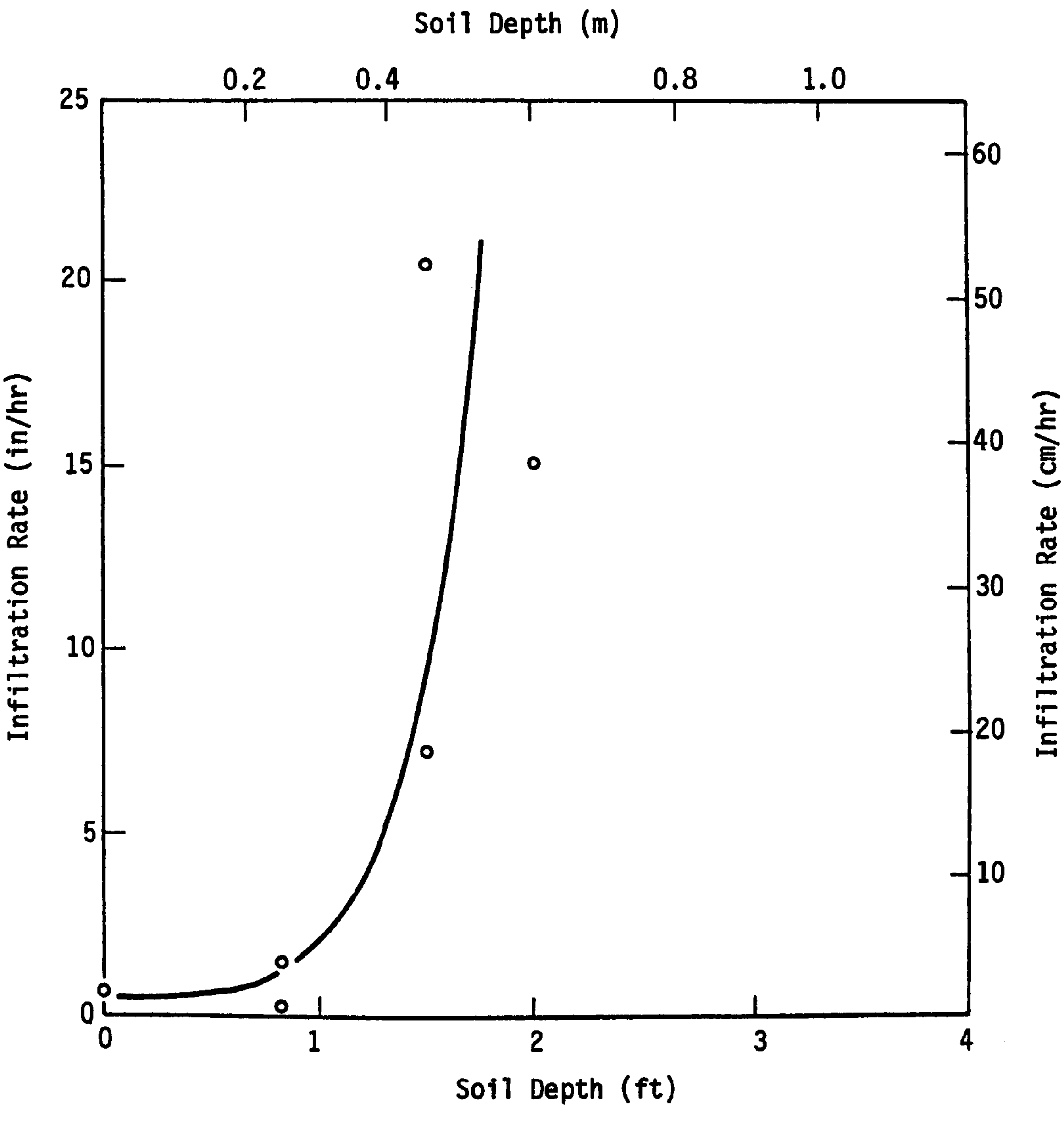


Figure 19. Infiltrometer test data on Basin 3 prior to modification.

TABLE 6. SUMMARY OF INFILTRATION RATES

Loading Date	M^3 Loaded	Infiltration Rate (cm/hr)	Loading Date	M^3 Loaded	Infiltration Rate (cm/hr)	Loading Date	M^3 Loaded	Infiltration Rate (cm/hr)
Bed 1			Bed 2			Bed 3		
2-8-77	1024	0.99	4-13-77	255	9.5	2-10-77	1262	3.2
2-15-77	1256	0.84	4-20-77	273	2.2	2-17-77	997	0.58
2-22-77	1158	0.84	4-27-77	-	2.4	2-24-77	1330	0.58
3-1-77	1166	0.64	5-4-77	263	2.5	4-14-77	1220	0.61
3-8-77	1193	0.48	5-18-77	261	4.8	4-21-77	984	0.56
3-15-77	1240	0.48	6-1-77	238	5.1	5-12-77	1191	2.3
4-12-77	817	2.5	6-8-77	320	4.8	5-19-77	1080	1.1
4-19-77	1146	0.38	6-15-77	248	5.4	6-2-77	1186	2.3
5-10-77	1195	2.9	6-22-77	250	5.4	6-9-77	1160	1.9
5-17-77	957	0.71	6-29-77	525	3.3	6-23-77	946	1.6
5-31-77	986	2.8	7-6-77	253	5.1	6-30-77	1169	1.4
6-7-77	1079	2.0				7-7-77	1147	1.2
6-14-77	1212	1.7	Average	289	4.6			
6-21-77	1257	1.6				Average	1139	1.4
6-28-77	1239	1.4						
7-5-77	1114	1.3						
Average	1128	1.3						

infiltration rates are also shown for Beds 1, 2, and 3 at 1.4 cm/hr (0.53 in/hr), 4.6 cm/hr (1.81 in/hr), and 1.45 cm/hr (0.57 in/hr), respectively.

Comparison of the recorded infiltration rates with the average rates obtained in the infiltrometer tests shows a consistent relationship between the two. The average infiltration rate for the surface of Bed 1 was found to be 4.4 cm/hr (1.74 in/hr) in the infiltrometer tests. The corresponding recorded infiltration rate for Bed 1 was 1.4 cm/hr (0.53 in/hr), or 30% of the infiltrometer test value. Similarly, the recorded infiltration rate for Bed 3 was 26% of the infiltrometer test value, and the recorded infiltration rate for Bed 2 was 38% of the infiltrometer test value.

The higher infiltration rates obtained from the infiltrometer tests may have been associated with any of several factors. However, the major factor which contributed to the high rates was believed to be lateral flow from the infiltrometer rings. The combination of lateral and vertical flow through the soil medium served to increase the rate at which the soil would accept the applied water. The U.S. Salinity Laboratory Staff has reported that the impact of lateral flow increases as the infiltration area decreases (7). The infiltration area used in the Boulder infiltration-percolation project was very small compared to the total area of the basins. Although lateral flow was minimized by the outer buffer pond in the double ring infiltrometer, it was still felt to be primarily responsible for the higher rates observed in the infiltrometer studies.

FACTORS AFFECTING INFILTRATION RATES

The infiltration rates in an infiltration-percolation system are apparently affected by operation and management practices, and wastewater characteristics, in addition to site conditions at a specific location, as reflected by the data collected during this study.

Operation and management practices of an infiltration-percolation system consist mainly of the loading schedule, the basin surface management, and the method of wastewater application. Varying the loading schedule, i.e., the inundation and drying period of an infiltration-percolation system, has been shown to affect the infiltration rates of that system. The extent of this impact is difficult to determine, but it has been stated in the literature that maximum infiltration rates are achieved by shorter inundations and longer drying times (8,9,10,11).

The longer drying times allow for aeration of the soil, and enhance the decomposition and dessication of organic material deposited during the inundation period. The microbial population will also decrease as substrate is utilized, thereby increasing the pore space available for infiltration (11). In the Boulder study, the loading schedule was not intentionally varied, but observations made during bed operation showed the approximate drying times for Beds 2, 1, and 3 to be 3 days, 2 to 2.5 days, and 1 to 2 days, respectively. A plot of the recorded infiltration rate as a function of time is shown for Basins 1, 2, and 3 in Figures 20, 21, and 22, respectively. From these figures it can be seen that the infiltration rate following drying and scarification was the highest. In Basins 1 and 3, the high initial infiltration rate was followed by a significant rate decrease as the loading cycle progressed. The high initial infiltration rate was attributed to the soil aeration and scarification procedure which served to remove any organic matter that may have formed during the preceding loading cycle. The highest infiltration rate for Basins 2 and 3 followed a long drying period which included the period of bed modification. These high rates, observed on February 10, 1977 for Basin 3, and April 13, 1977 for Basin 2, resulted primarily from the bed modifications which created a new soil-water interface for infiltration. Since the literature suggests that soil clogging is a surface phenomena, it follows that stripping of the beds removed the clogged surface and created a "fresh" surface (12, 13).

The effect of basin surface management on the infiltration rate was investigated to only a limited degree at the Boulder site. Basin surface management practices may include the growth of vegetation on the surface, placement of a layer of sand or gravel, or operation with a bare soil surface. Surface management may also encompass the type and frequency of scarification during the drying periods. All beds in the Boulder system had bare surfaces, so comparison with vegetated or gravel surfaces was not possible. The scarification practiced in this study undoubtedly contributed to the restoration of high infiltration rates observed at the beginning of each loading cycle. However, since the same type and frequency of scarification was used throughout the operation of the basins, comparison with other methods was not possible.

A gradual decline in infiltration rates was exhibited as the loading cycle progressed on Beds 1 and 3. Many factors likely contributed to this decline in infiltration rates. Other investigators have suggested that the major factors contributing to such declines are the accumulation of suspended solids, and the microbial activity which is effective in degrading the applied organics. However, Bed 2 did not exhibit these declining trends, even though it was subjected to application with the same strength wastewater. The differences in operation between basins which may have caused the infiltration rate in Basin 2 to remain constant were: (1) the water was applied to Beds 1 and 3 by total surface flooding, and (2) Bed 2 was loaded at a lower loading rate than Beds 1 and 3. Although Bed 2 was not scarified after the drying period, while Beds 1 and 3 were, this fact would only affect the initial infiltration rate and would not be responsible for maintaining infiltration rates during the loading cycle. The fact that Bed 2 was a ridge and furrow system may explain why infiltration rates were maintained during the loading cycle. It has been suggested by McGaughey that a ridge and furrow system would not be affected by suspended solids to the same degree as a flat surface (8). Since the solids that settle will only alter the surface at the bottom of the furrow, the sides of the furrow remain relatively free of clogging solids and, therefore, allow the initial infiltration rate to be maintained. Figures 23, 24, and 25 show the infiltration rate plotted as a function of accumulated suspended solids for each of the three basins. From these figures it can be seen that the accumulated

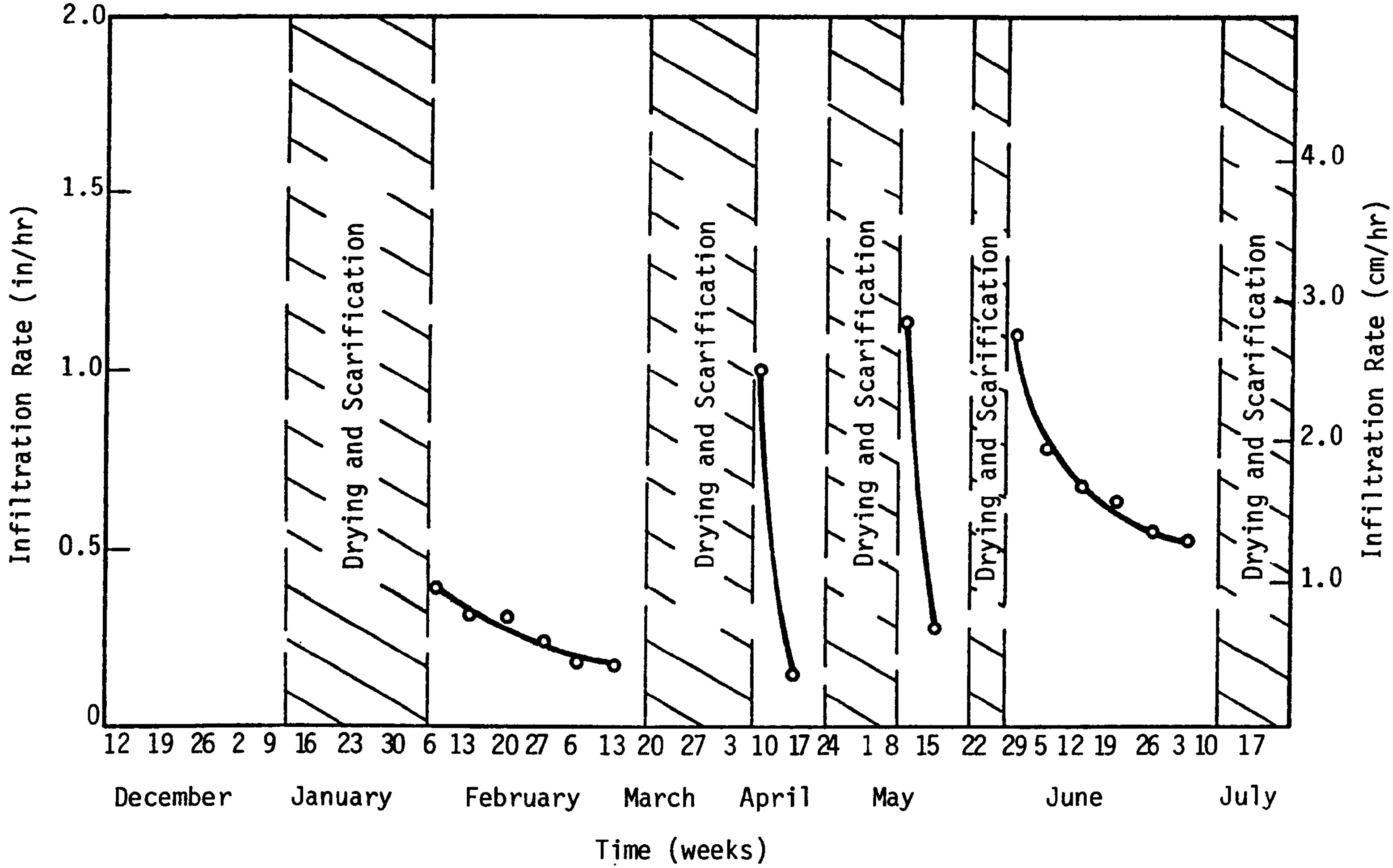


Figure 20. Pattern of infiltration rate decline in Basin 1.

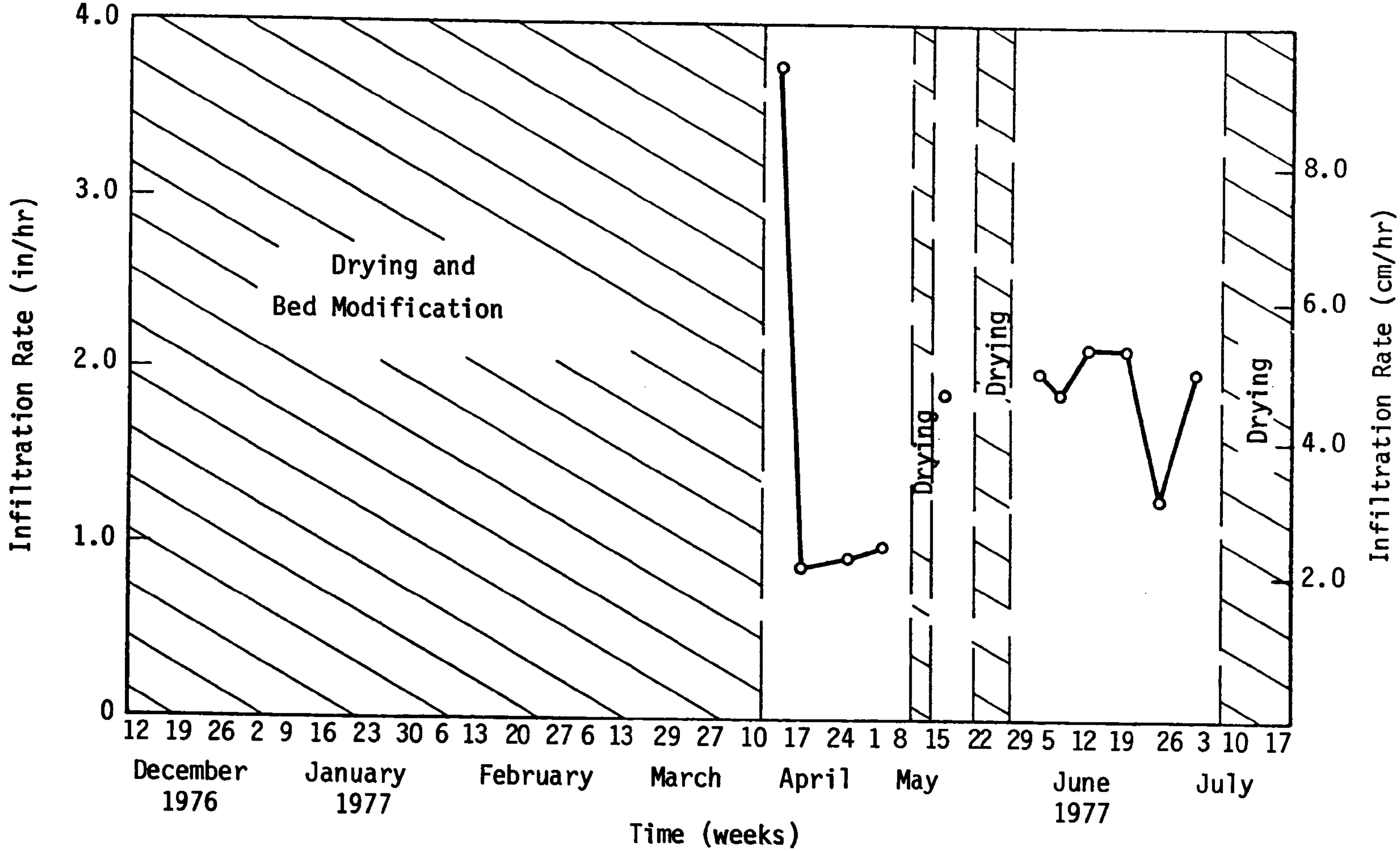


Figure 21. Pattern of infiltration rate decline in Basin 2.

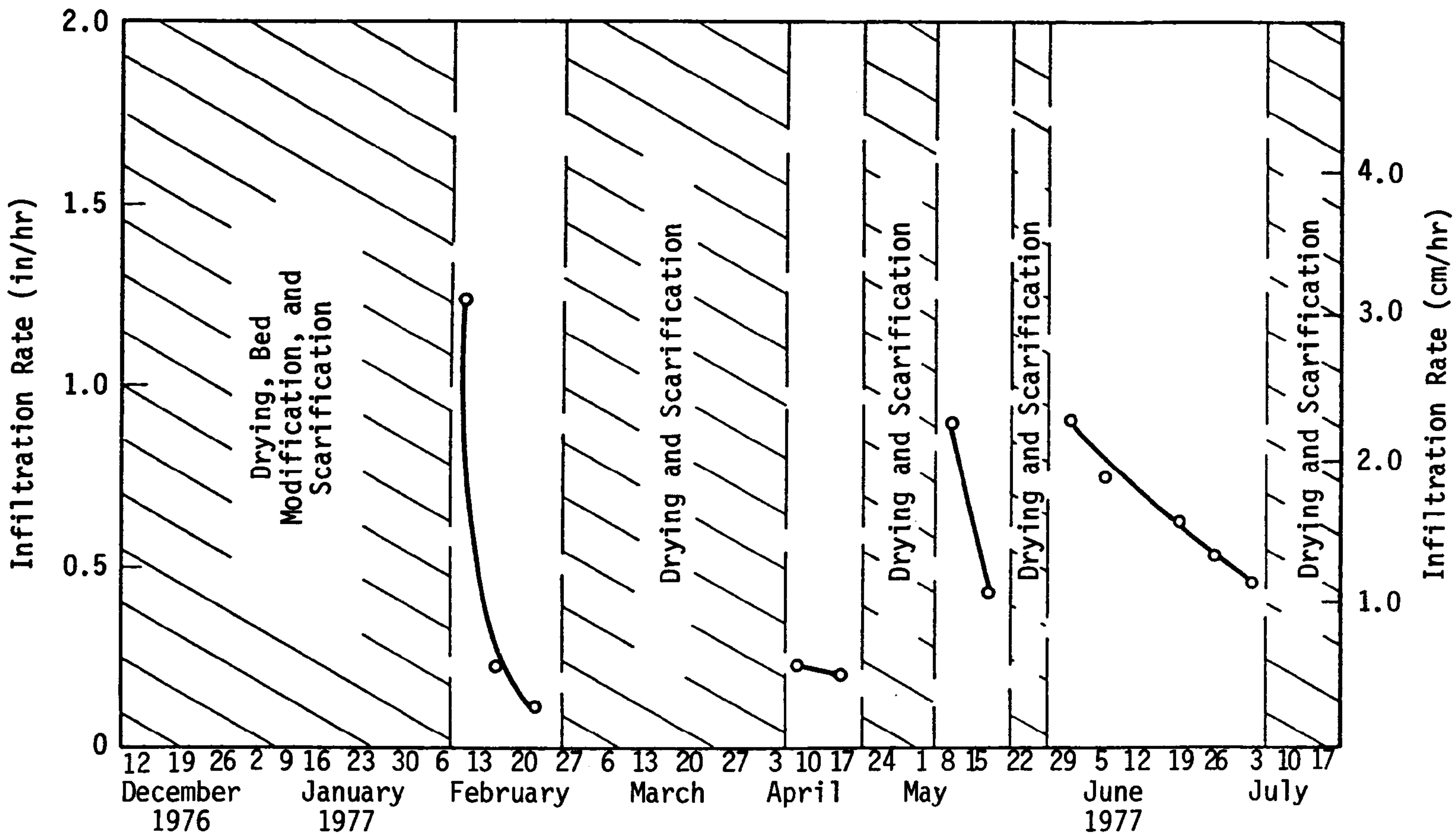


Figure 22. Pattern of infiltration rate decline in Basin 3.

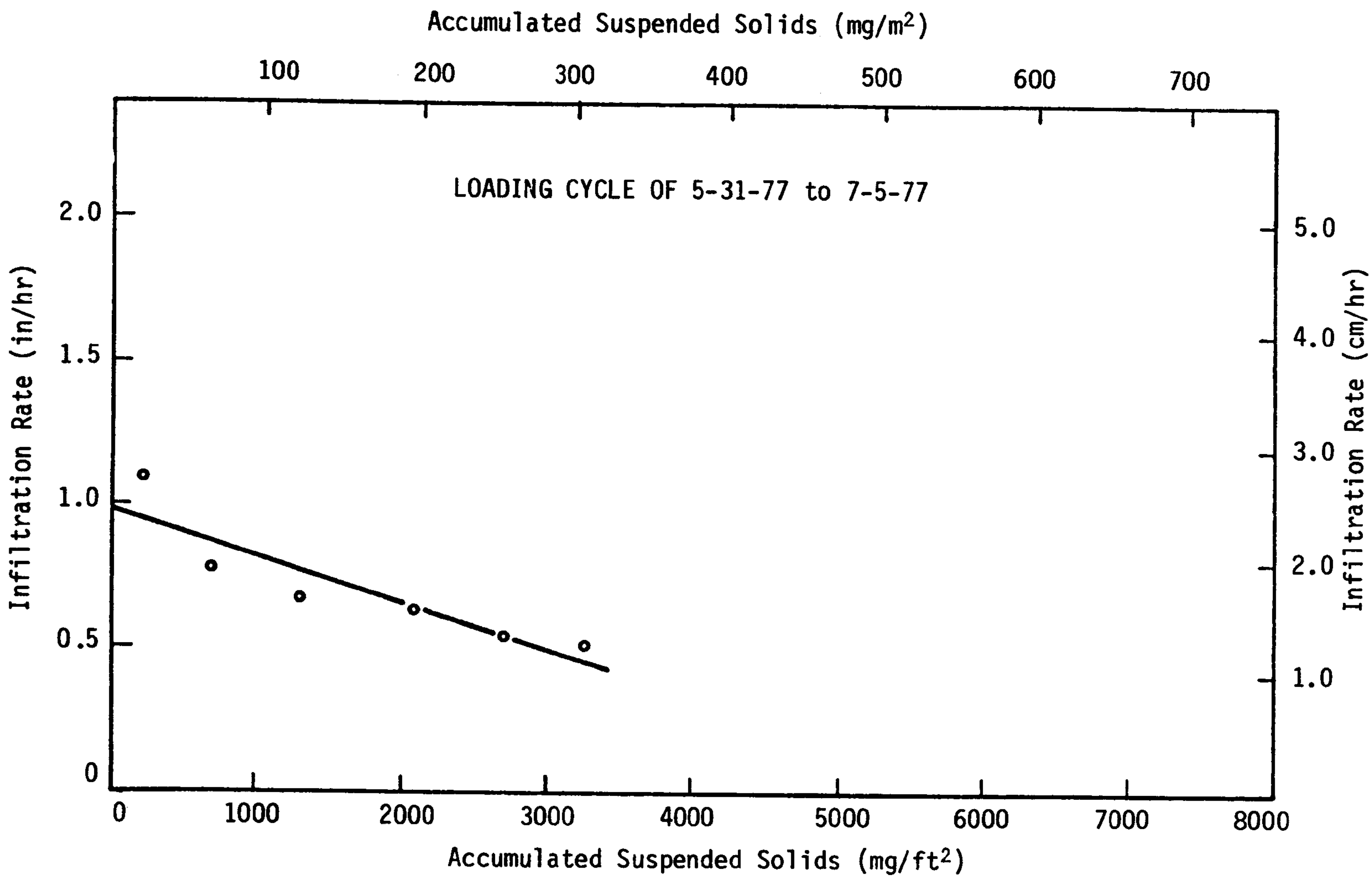


Figure 23. Infiltration rate as a function of suspended solids loading in Basin 1.

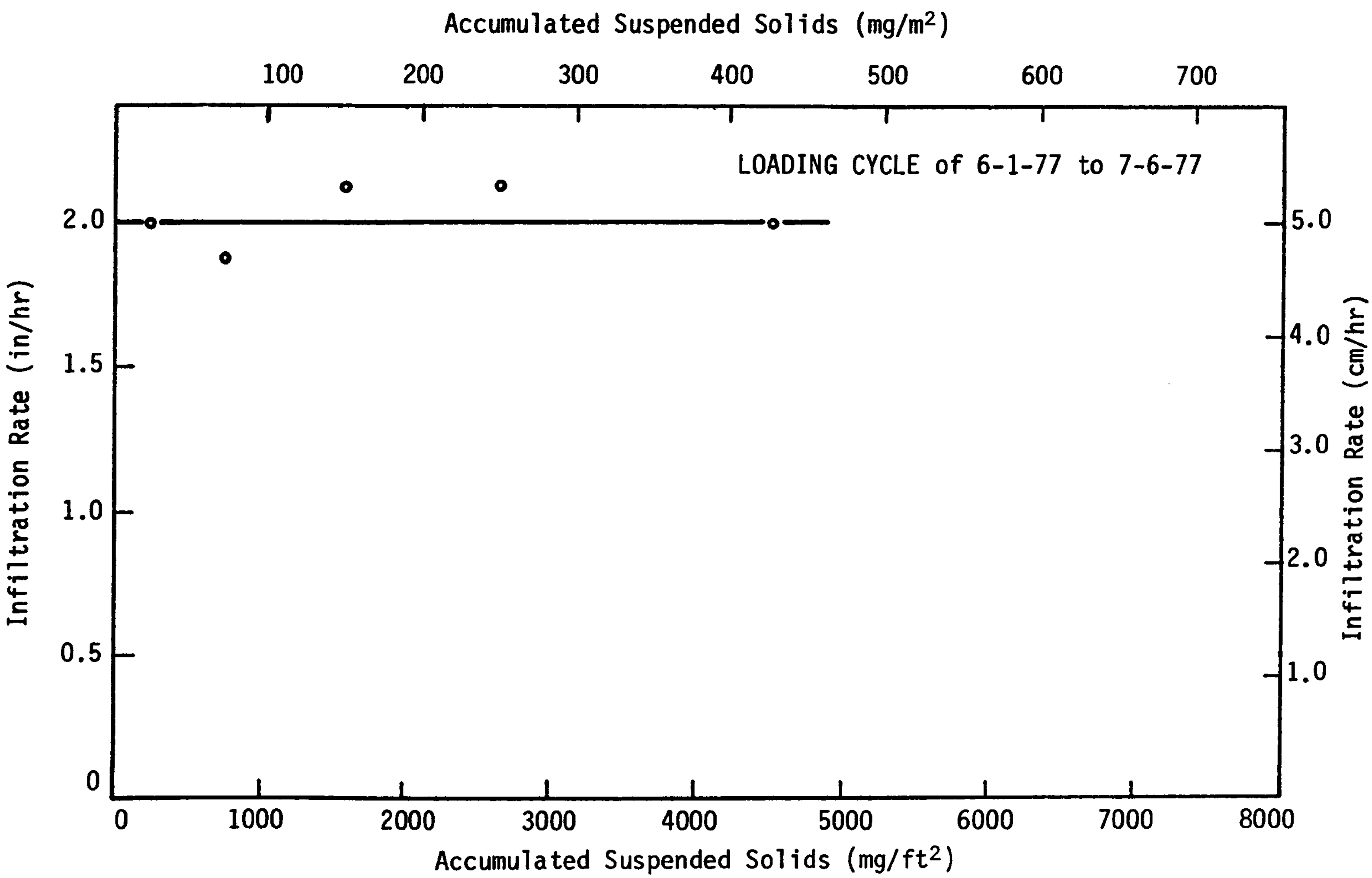


Figure 24. Infiltration rate as a function of suspended solids loading in Basin 2.

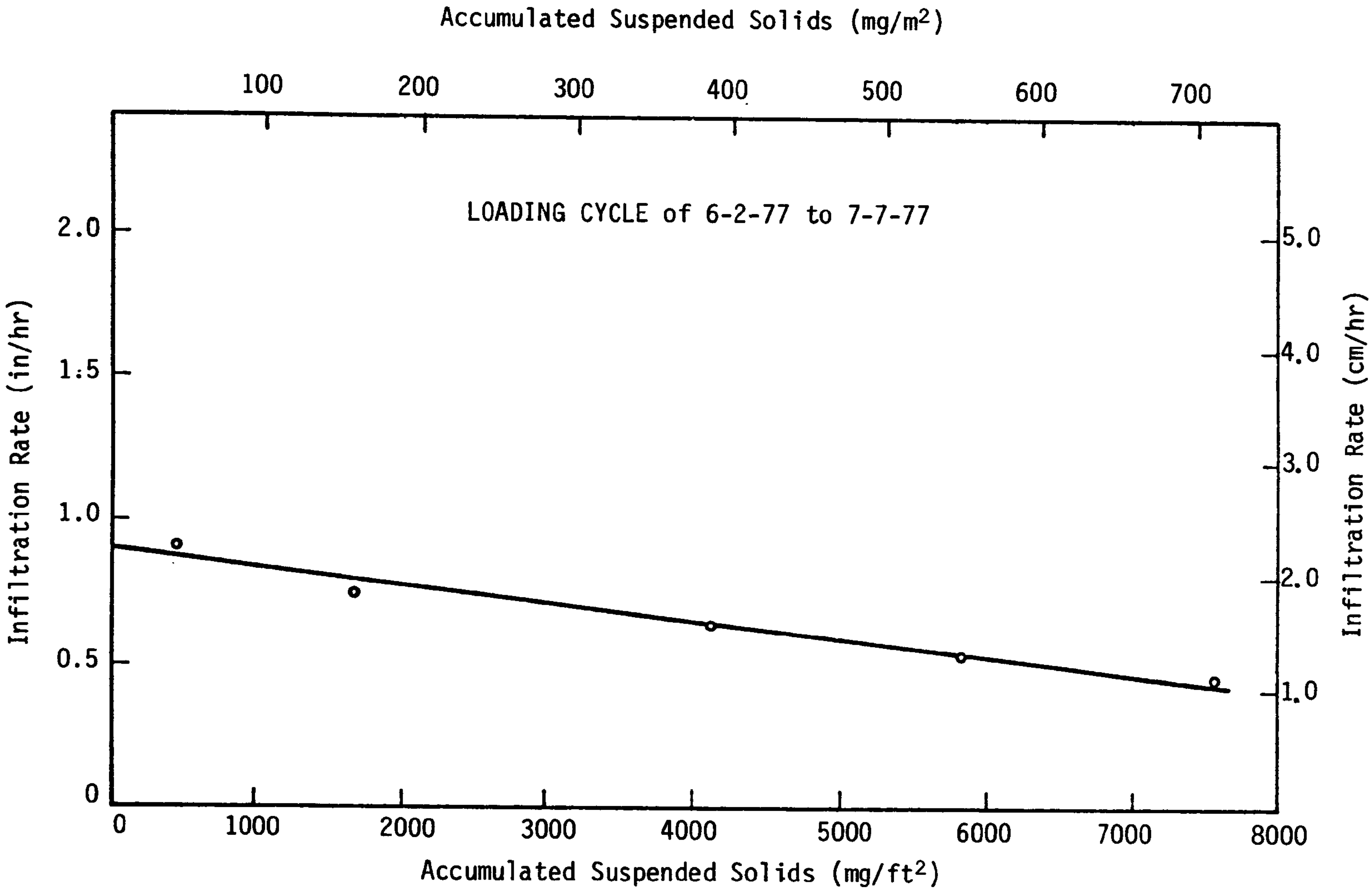


Figure 25. Infiltration rate as a function of suspended solids loading in Basin 3.

suspended solids did not appear to have an effect on the infiltration rates in Basin 2, but appeared to affect the infiltration rates of Basins 1 and 3. Based on these data, it was concluded that the existence of a ridge and furrow system in Basin 2 was partially responsible for maintaining the relatively high initial infiltration rates.

The lower loading rate applied to Bed 2 was undoubtedly responsible, in some part, for the relatively constant infiltration rates. This lower loading rate permitted longer drying times between the loading times on Bed 2, when compared to Beds 1 and 3. As stated previously, Bed 2 was dry for approximately 3 days while Beds 1 and 3 were dry from 2 to 2.5 days, and 1 to 2 days, respectively. These longer drying times allowed for a more complete aeration of the soil in Basin 2 and enhanced the decomposition and dessication of the organic material deposited during the flooding. Thus, more pore volume became available for the transport of water, resulting in the maintenance of initial infiltration rates throughout the cycle.

The influent temperature also appeared to have some effect on the infiltration rate. Figure 26 indicates the average infiltration rate over a complete loading cycle as a function of the average temperature of the influent over the same period. The high infiltration rates observed in Basins 2 and 3 immediately following their modification were not included in these averages because these high rates resulted mainly from the bed modification, as was previously discussed. The trends illustrated in Figure 26 were attributed to the lower viscosity of the water at higher temperatures, and to the fact that the quality of the wastewater influent to the basins was generally better during the warmer weather.

While temperature did seem to affect the infiltrative capacity of the soil, the cold temperatures encountered did not necessitate discontinuing the operation of the infiltration-percolation system. Even though there was some freezing of the beds during February and March, the ice did not appear to interfere with bed performance. The ice layer apparently served to insulate the underlying water and collapsed as the water level declined. Subsequent loadings melted the broken ice, after which a new ice layer formed and the cycle was repeated. Prolonged periods of sub-freezing temperatures could freeze the beds solid, which would cause severe operational problems. However, this did not occur in the Boulder situation.

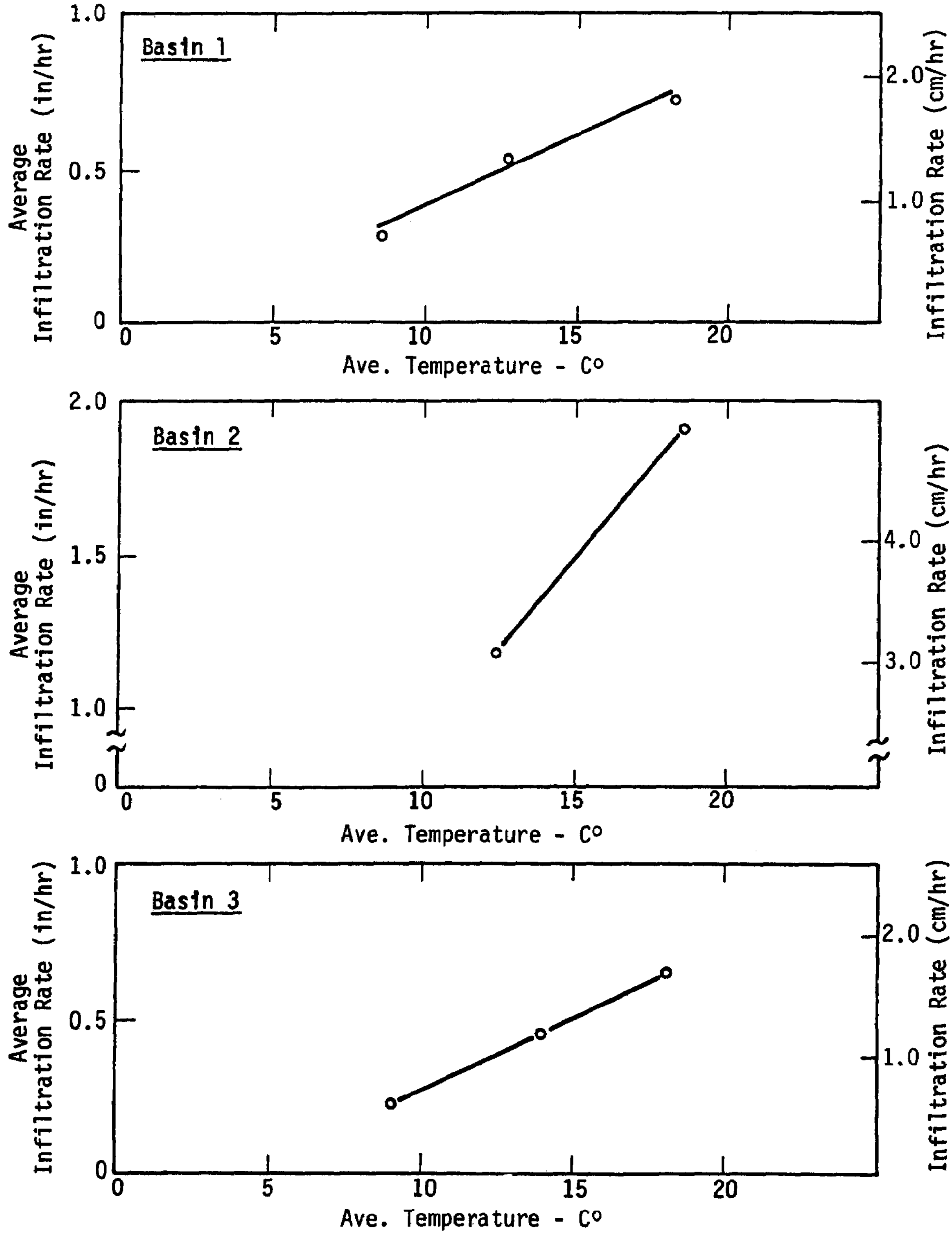


Figure 26. Infiltration rate as a function of wastewater temperature.

SECTION 7

TREATMENT PERFORMANCE OF THE INFILTRATION-PERCOLATION SYSTEM

GENERAL CONSIDERATIONS

In assessing the treatment performance of the infiltration-percolation system, the primary focus was directed toward monitoring the fate of the following major wastewater constituents: phosphorus, organics, nitrogen, hardness, alkalinity, and coliform organisms. In addition, the behavior of selected heavy metals was assessed in a specific short term study. The aim of the study was to evaluate the treatment efficiency and its variation with seasonal and climatic changes, as well as long term equilibration effects.

Table 7 summarizes the average seasonal constituent concentrations in the secondary effluent applied to the beds. Seasonal values were derived by averaging the constituent concentrations of all the secondary effluent samples taken during a particular season. For example, the average fall COD concentration was the average of the COD concentrations in all of the secondary effluent sampled between September 22 and December 22. Some of the seasonal trends observed in the renovated water quality, e.g., COD, were a result of the seasonal variation of the pollutant concentrations in the applied secondary effluent. Other seasonal trends observed, such as the phosphate level in the renovated water, seemed to be less affected by the average seasonal pollutant concentration in the applied secondary effluent.

Figure 27 summarizes the annual precipitation pattern as well as the minimum weekly temperature in the renovated water from Basin 1 for the first year of study. These data have been presented for reference, since the minimum renovated water temperature reflected an environmental condition within the soil system that may have affected the removal efficiencies for certain constituents. The weekly precipitation represents a moisture effect that may have been effective in causing some of the observed seasonal variations.

PHOSPHORUS BEHAVIOR

Data compiled during the course of this study showed the phosphorus concentrations in the basin effluents to vary over quite a broad range. This variation is demonstrated in the total phosphorus concentration profiles of Figure 28.

In all three of the basins, there appeared to be a significant seasonal

TABLE 7. SEASONAL AVERAGES FOR WASTEWATER CONSTITUENTS IN BOULDER SECONDARY EFFLUENT

Constituent	Spring	Summer	Fall	Winter	Annual
Total Nitrogen[1]	13.8	13.2	14.6	24.5	16.5
Ammonia	7.3	6.0	5.3	11.7	7.6
Organic Nitrogen	4.7	4.7	6.6	12.2	7.0
Nitrate	1.8	2.5	2.7	0.6	1.9
Total Phosphate[2]	4.8	5.1	7.2	7.9	6.2
COD	58.0	58.1	72.1	118.3	76.6
Alkalinity[3]	128	118	125	164	134
Hardness[3]	132	154	119	112	129
Calcium	34.6	35.6	-	32.2	34.1
Magnesium	11.0	14.8	-	7.4	11.1
Chlorides	27.4	23.2	32.1	27.3	27.5
Suspended Solids	14.6	11.7	21.6	25.4	18.3
Total Solids	311	333	286	302	308

[1]All nitrogen concentrations in mg/ℓ as N

[2]mg/ℓ as P

[3]mg/ℓ as $CaCO_3$

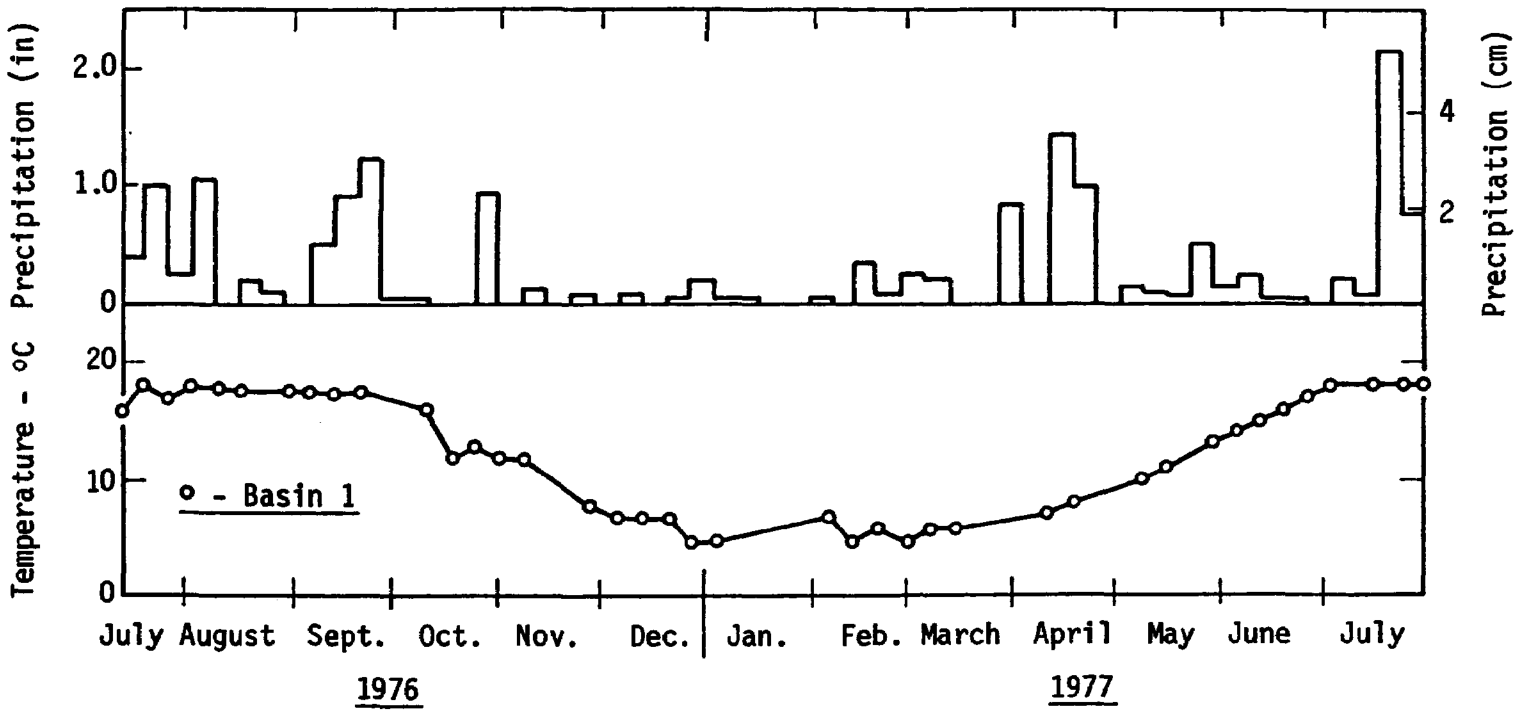


Figure 27. Weekly precipitation and minimum temperature of renovated water.

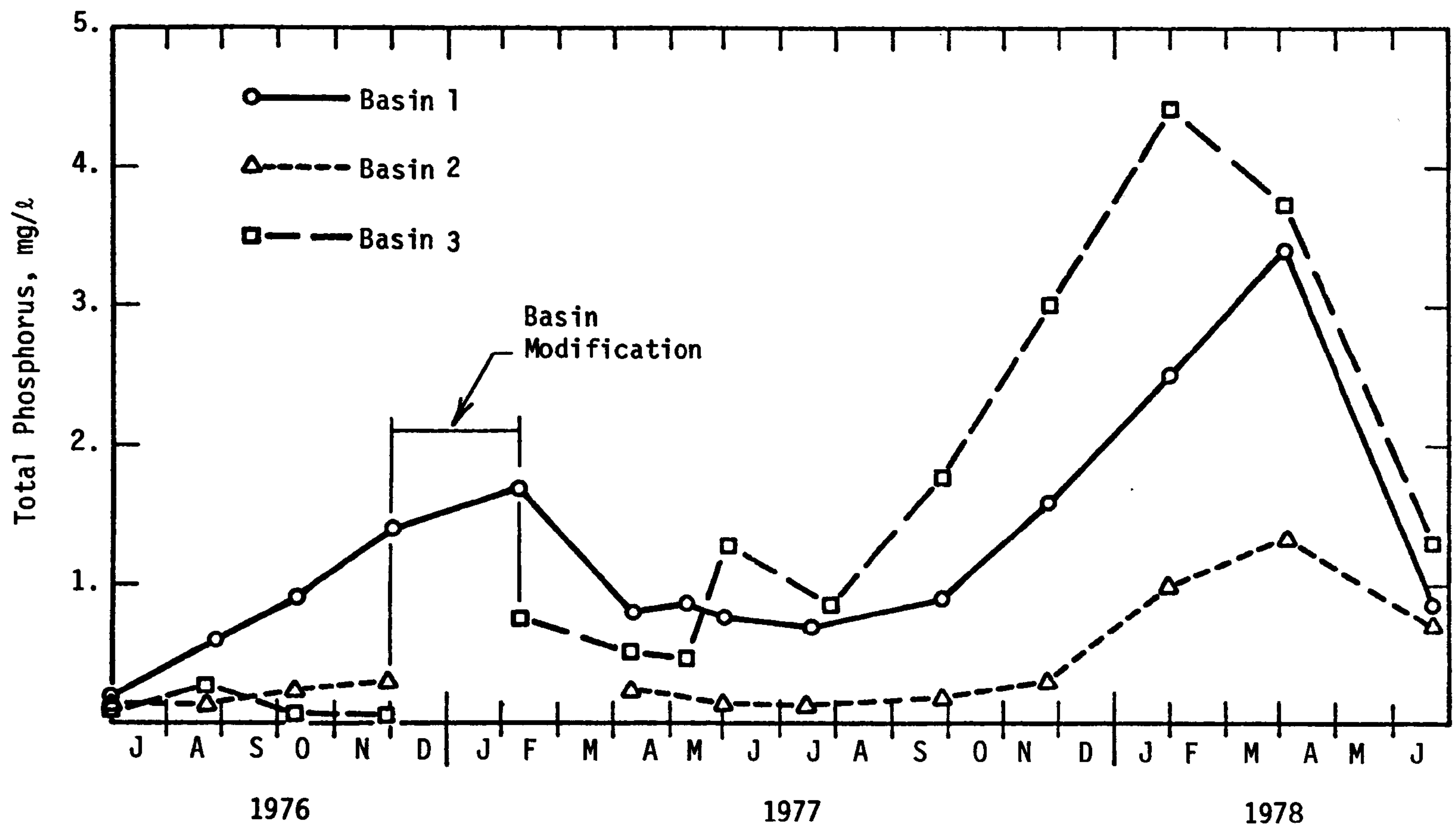


Figure 28. Effluent phosphorus as a function of time.

pattern to the leakage of phosphorus through the basins. The concentration of total phosphorus appearing in the renovated water increased markedly during the winter loading cycles when the wastewater temperatures were the lowest, and the influent phosphorus concentration was the highest because of reduced infiltration in the municipal collection system and lower phosphorus uptake in the trickling filter system.

It has been demonstrated that phosphates introduced to the soil system are initially adsorbed in the mineral or organic fraction of the soil (14, 15). The adsorbed phosphate is subsequently precipitated or mineralized as inorganic compounds into the soil matrix by a slower reaction. This precipitation reaction apparently serves to release the adsorption sites within the soil for additional phosphorus removal. The adsorption process will limit the rate at which phosphorus can be removed by the soil until the soil adsorption sites become saturated. Further phosphorus removal will be controlled by the slower rate of precipitation of relatively insoluble compounds (16). The increase in the phosphorus levels applied to the beds during the fall and winter apparently saturated the phosphorus adsorption capacity within the soil system. At this point, the slower precipitation reaction effectively controlled the rate and limited the effectiveness of the phosphorus removal process. This effect was shown most dramatically in the winter of 1977-78 after Basins 2 and 3 had been modified. During modification, Basin 3 had most of the fine grained soil removed from the upper portion of the soil profile. Following modification, this basin was loaded at the highest equivalent hydraulic loading rate of 48.9 m/yr (160 ft/yr). With this combination of basin modification and loading characteristics, the phosphorus removal efficiency was reduced in Basin 3 to about 40% during January of 1978. Conversely, Basin 2, which was loaded at the lowest rate, demonstrated a much higher and more consistent phosphorus removal capability. These data suggest that by increasing the detention time in the soil system, the phosphorus removal efficiency might be controlled at a very high level.

Analysis of the phosphorus profile for Basin 1 indicates that the phosphorus leakage peaked during both winters of operation. However, the effluent concentration during the second year of operation was about twice that observed during the first winter. These data suggest that the system had not fully equilibrated in the first year of operation. This finding is consistent with the findings reported from the Lake George, N.Y. system by Aulenbach (17, 18). Apparently, the regeneration of adsorption sites by mineralization is not complete, and a continual degradation of system performance can be expected in phosphorus removal.

In addition to the seasonal trends which have been discussed, significant variation was also noted in the phosphorus leakage during different loadings within a cycle. In most of the loading cycles, the concentration of total phosphate in the renovated water increased from the first loading of the cycle to the last. Selected examples showing this trend have been presented in Figure 29. In keeping with this trend, the total phosphate concentration in the renovated water from the first loading of a loading cycle was usually less than the phosphate concentration in the renovated water from the last loading of the previous loading cycle. These

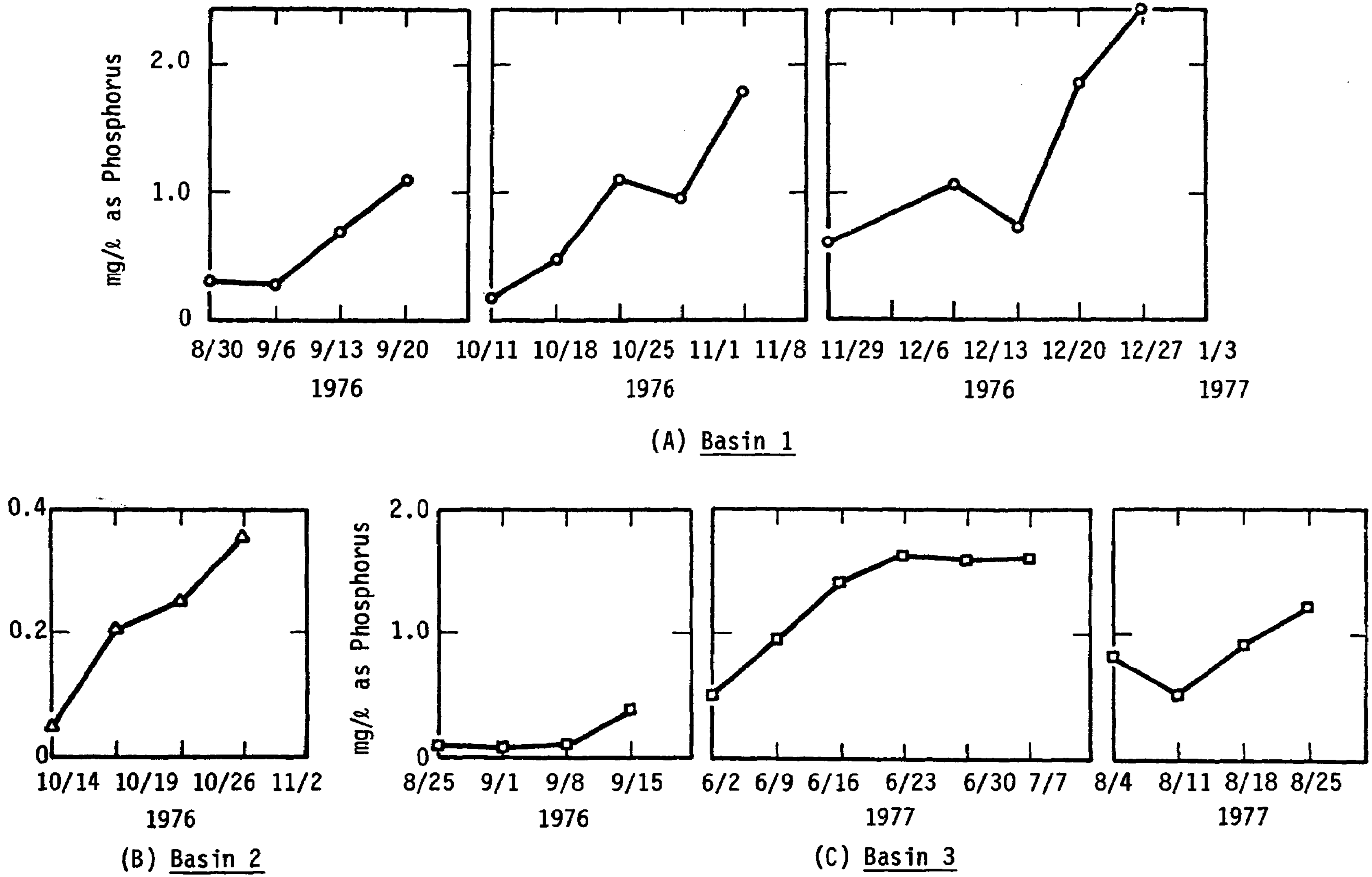


Figure 29. Effluent phosphorus variation within loading cycles.

observations can be explained in terms of the slow precipitation of phosphate in the soil matrix. During the resting period between loading cycles, rejuvenation of the soil phosphate adsorption capacity apparently occurred. The soil was able to remove the initial application of phosphate most efficiently during the first loading of the new cycle. The subsequent applications of effluent during the cycle showed that the capacity of the soil to remove phosphate was reduced as the cycle continued. By the last loading of the loading cycle, the soil adsorption capacity for phosphates was in its most exhausted state, and the greatest leaching of phosphates through the soil system occurred at that time.

REFRACTORY ORGANICS

Refractory organics were monitored during the study period by sampling the secondary effluent during application to each bed, and the renovated water from the basin 24 hours after application commenced. Figure 30 presents a summary of the average COD level for each loading cycle in the basin effluents. As can be seen from this figure, the infiltration-percolation system was generally effective in reducing the product COD to a residual level of 10-20 mg/ℓ. These levels represented a reduction of the influent COD by 70-80%. The only significant deviation from this treatment pattern occurred in the December-February period of 1976-77. During that time the influent had unusually high levels of COD which were attributed to highly concentrated industrial waste discharges into the Boulder Treatment Plant.

The successful operation of the basins through the winter demonstrated the fact that cold conditions did not reduce the capability of the soil for removing organics from the applied wastewater effluents. This performance supports the findings of other authors regarding the temperature insensitivity for organic removal in high rate systems. In one study of this phenomenon, Thomas, et al. loaded six silica sand lysimeters with secondary and septic tank effluent to evaluate the effect of temperature on COD removals (19). Two lysimeters were incubated at 28°C, two at 18°C to 30°C, and two at 18°C to 35°C. All six lysimeters produced comparable COD removal efficiencies. Schwartz, et al. also reported that cold climate did not restrict the COD removal capability of a high-rate land treatment system.(20).

The relative positions of the three curves in Figure 30 suggests that the organic content of the product water may bear some relationship to the hydraulic loading of the basin. Following bed modification, Basin 3 was loaded at the highest rate, and the product water from this basin was typically the highest in COD. Consistent with this observation, Basin 1, the next most heavily loaded, had a lower effluent COD, and Basin 2 had both the lowest hydraulic loading and the lowest product COD level.

NITROGEN

Nitrogen was the most difficult constituent to evaluate with respect to the treatment performance of the infiltration-percolation basins. The reason

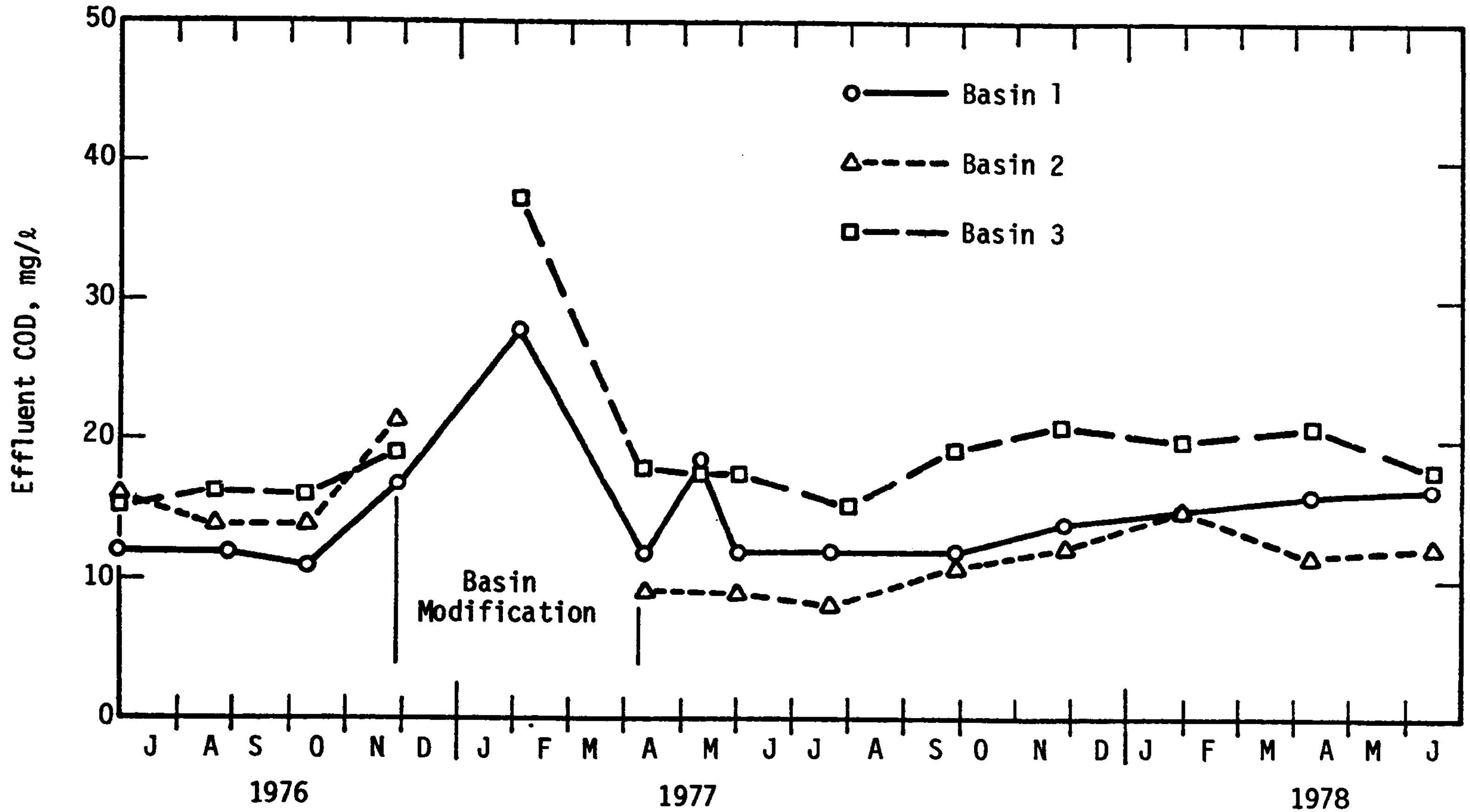


Figure 30. Effluent COD as a function of time.

for this is that the nitrogen cycle contains several pathways for transformation of the organic and ammonium nitrogen applied to a soil system in secondary effluent. The three major sequences involve the following reactions: (1) The ammonium ions may be held in the soil by adsorption or fixation processes; (2) The ammonium ions retained within the soil by adsorption or fixation may be nitrified to nitrate and leached through the soil system with the renovated water; and (3) The nitrate, once formed, can be denitrified in the presence of a carbon source and anoxic conditions. The The adsorption and nitrification processes were clearly the most important during the operation of the infiltration-percolation basins in this study. However, the relative importance of the fixation process, which allows indefinite nitrogen storage, and the denitrification process, was difficult to evaluate.

Nitrate and ammonium ions were the principal nitrogen forms discharged in the basin effluents, with the nitrate ion accounting for about 98% of the total nitrogen discharged from Basin 1. Figures 31 and 32 show the average ammonium ion and nitrate concentrations, respectively, for each of the loading cycles during this study. The average nitrate and ammonium ion levels for a given weekly loading period were calculated by a flow weighting process. This involved development of a hydraulic discharge curve by plotting the hydraulic discharge from the basin as a function of sampling time. The total basin discharge was determined from this figure by integrating under the hydraulic discharge curve. The total pounds of nitrogen discharged by a basin during the week were determined by a similar method. A nitrogen mass flow curve was generated by plotting the product of the ammonium or nitrate nitrogen concentration and the flow rate as a function of sample time. Following normalization of units, the area under this curve yielded the total pounds of the ammonium or nitrate nitrogen discharged by the basin during the loading period. The average ammonium or nitrate concentration was found by dividing the pounds of ammonium or nitrate nitrogen discharged by the total hydraulic discharge.

Through an analysis of the input and output nitrogen quantities, a nitrogen balance was made for Basin 1 which covered the first full year of operation. Figure 33 shows the cumulative nitrogen balance through Basin 1 during this period. The upper line is the cumulative total nitrogen applied, and the lower line is the cumulative nitrogen discharged from the basin. The difference between the lines at any given date shows the total nitrogen storage or loss in the basin from July 12, 1976 through the loading cycle beginning on that date.

From these curves it is apparent that the basin tended to store nitrogen during the cold winter months of the first year and release it as the water temperature warmed in the spring. This figure suggests that the nitrification process was inhibited during the coldest period and ammonium nitrogen was stored by fixation or exchange in the basin. With the onset of the warming trend, nitrification of the ammonium stored in the basin seemed to be more complete, and nitrogen was released from the basin as the mobile nitrate ion. This observation is reinforced by the increased NH_4^+ leakage

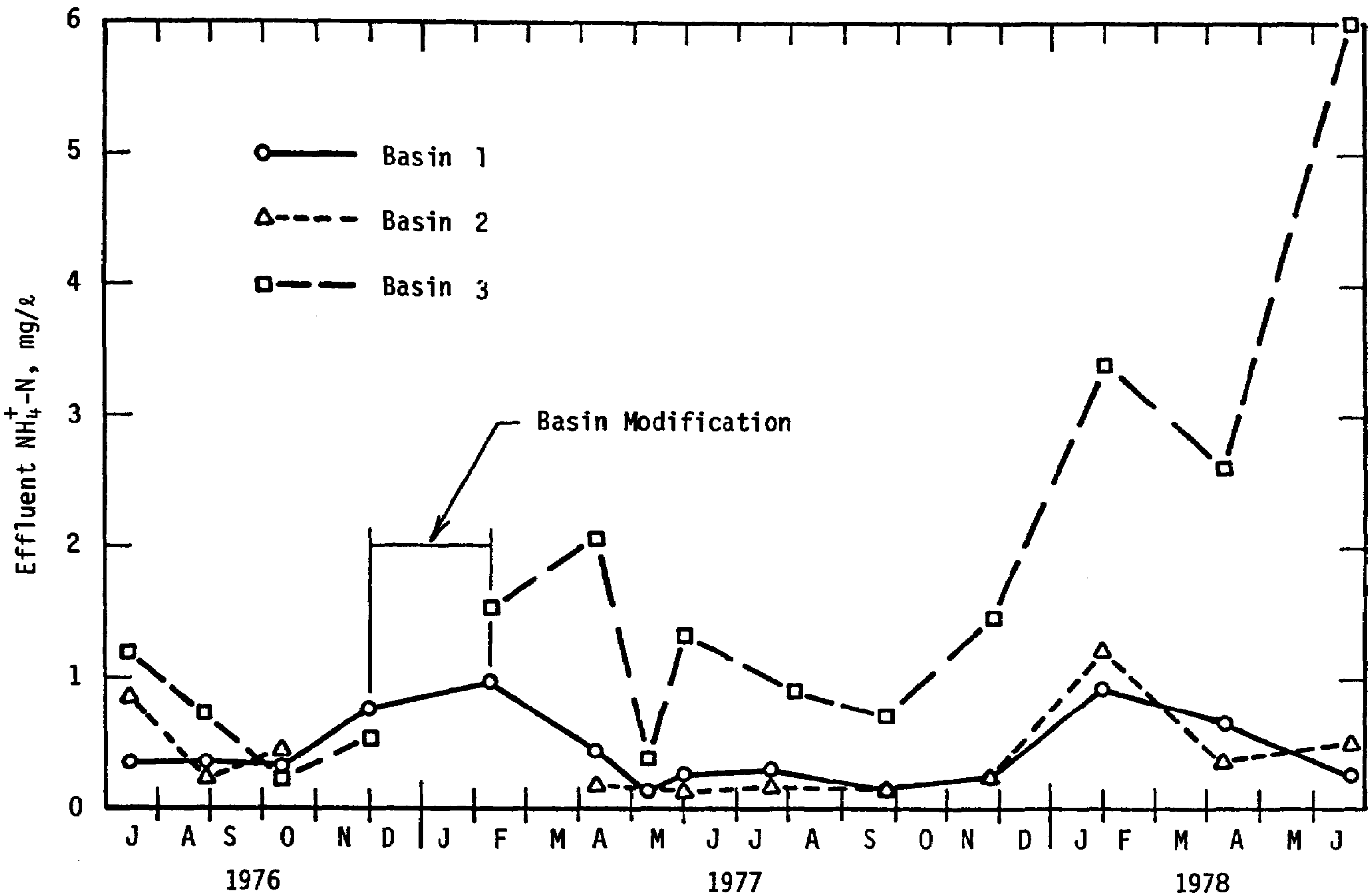


Figure 31. Effluent ammonium nitrogen as a function of time.

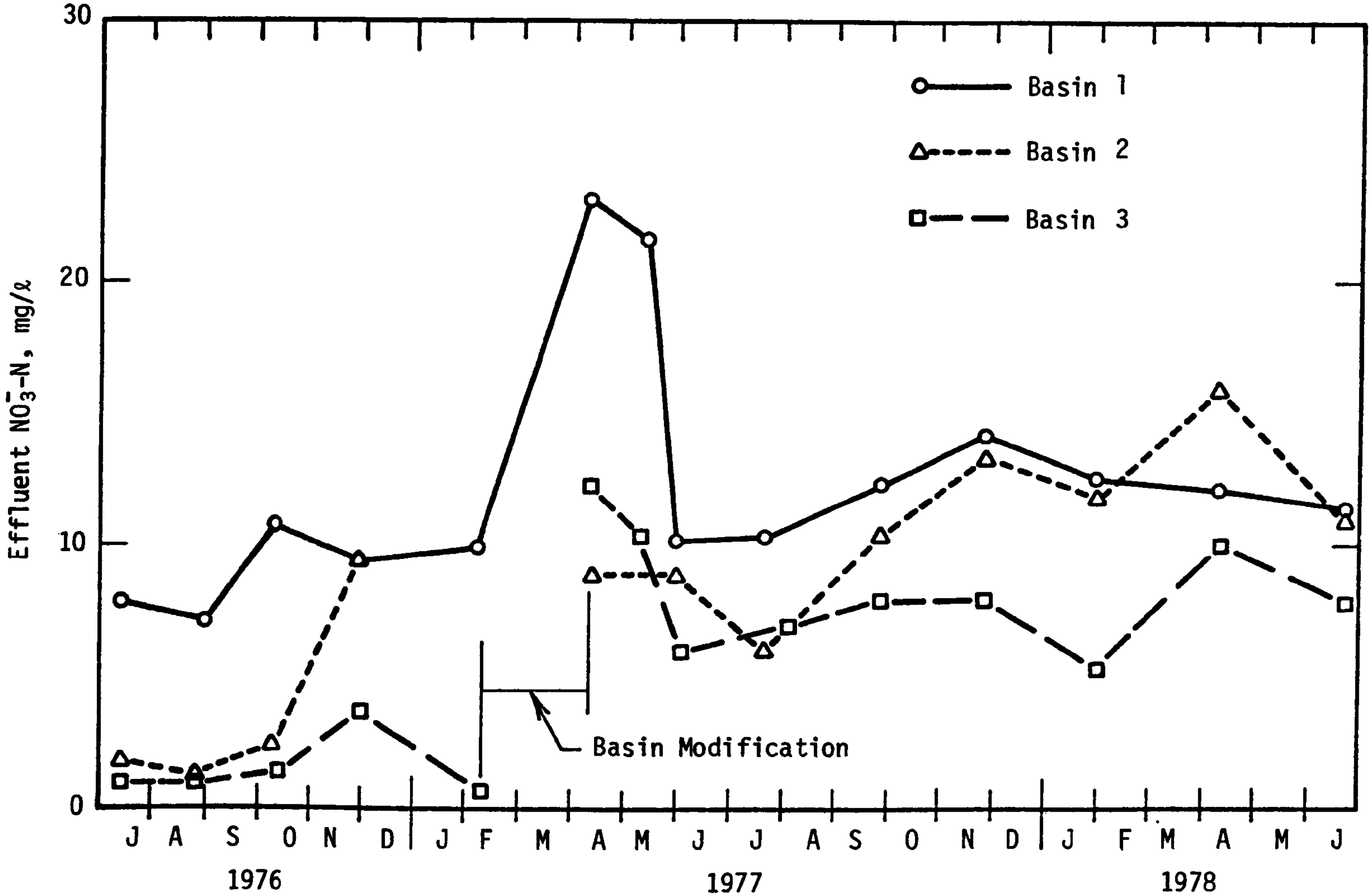


Figure 32. Effluent nitrate nitrogen as a function of time.

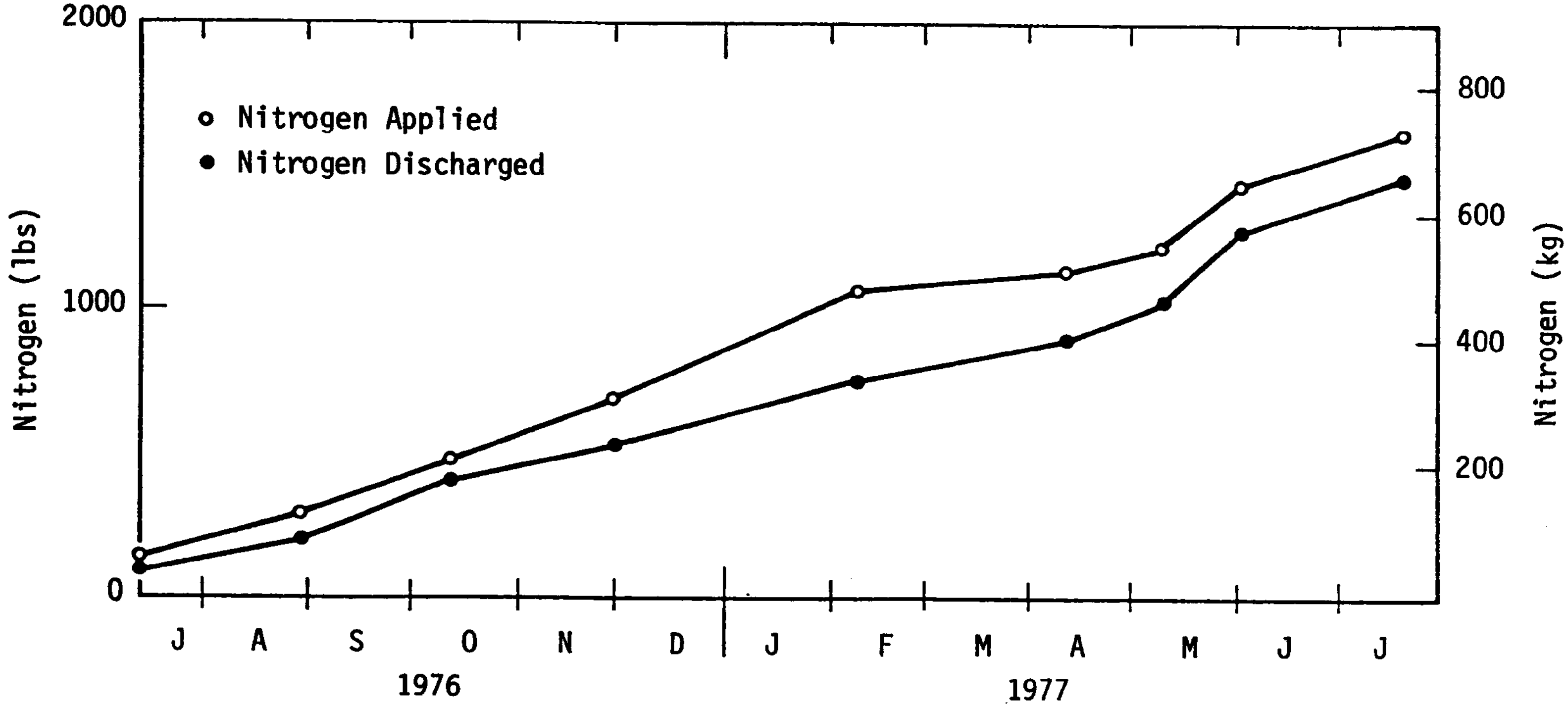


Figure 33. Cumulative nitrogen applied and discharged from Basin 1.

from Basins 1 and 2 during the colder winter months. A similar observation and explanation has been offered by Bouwer, et al. (21).

> "This seasonal trend is probably caused by greater drying of the soil between flooding periods in the summer than in the winter, which allowed more oxygen to diffuse to greater depths in the soil in the warmer months. Thus, more adsorbed NH_4^+ could be nitrified in the summer, causing more NH_4^+ to be adsorbed during flooding and, consequently, a decrease in the NH_4^+ level in the renovated water. This continued until the fall when poorer drying limited the amount and depth of penetration of oxygen. The seasonal trend could also be the result of the higher amount of SS in the effluent in fall and winter. These solids began to accumulate on the soil during winter and early spring. The resulting sludge layer could have acted as an oxygen sink during drying, thus reducing the amount of oxygen entering the soil."

This comment was made in relation to the Flushing Meadows system, but also appears applicable to the infiltration-percolation system operated by the City of Boulder. A plot of the suspended solids in the secondary effluent during the study period, Figure 34, did show that an increase in suspended solids occurred in the winter, as was the case at Flushing Meadows.

Seasonal nitrogen data from the Lake George system demonstrated the same general variations in bed performance which have been noted in this study. Ammonium concentrations taken from two sample wells were highest in the winter and lowest in the summer. Conversely, the nitrate concentrations were generally higher in the summer and fall. A similar ammonium leakage trend occurred during the operation of the Brookings infiltration system (22). The ammonium ion concentrations in the product water were lowest in the summer and highest in the winter. This trend likely reflects both the variation of the ammonium concentration in the applied wastewater, as well as the seasonal variation in the ability of the basins to convert ammonium to nitrates.

The nitrogen behavior in the basins during the second year of operation showed a somewhat less predictable pattern than that of the first year. The level of nitrification seemed to be higher and somewhat more consistent throughout the year, although there was a slight tendency toward peaking of effluent nitrates in the spring, and increased leakage of ammonium ions during the coldest winter months. However, in Basin 3, which was the most heavily loaded during this period, there seemed to be some very significant departures from the expected pattern. The ammonium level in the effluent was significantly higher than in the other two basins, and increased rather dramatically during the last 7-8 months of operation. This was accompanied by a nitrate level that was substantially below that of the discharge from Basins 1 and 2. These data indicate that the basin was unable to effect nitrification of an increasing amount of the applied nitrogen. This pattern seems to suggest that the basin was being stressed to the point that the hydraulic, ammonium, and organic loads exceeded the capacity of the system for transferring adequate oxygen to permit complete oxidation of the applied wastewater ammonium. Clearly, this loading would have to be reduced to provide acceptable ammonium removals on a long term basis.

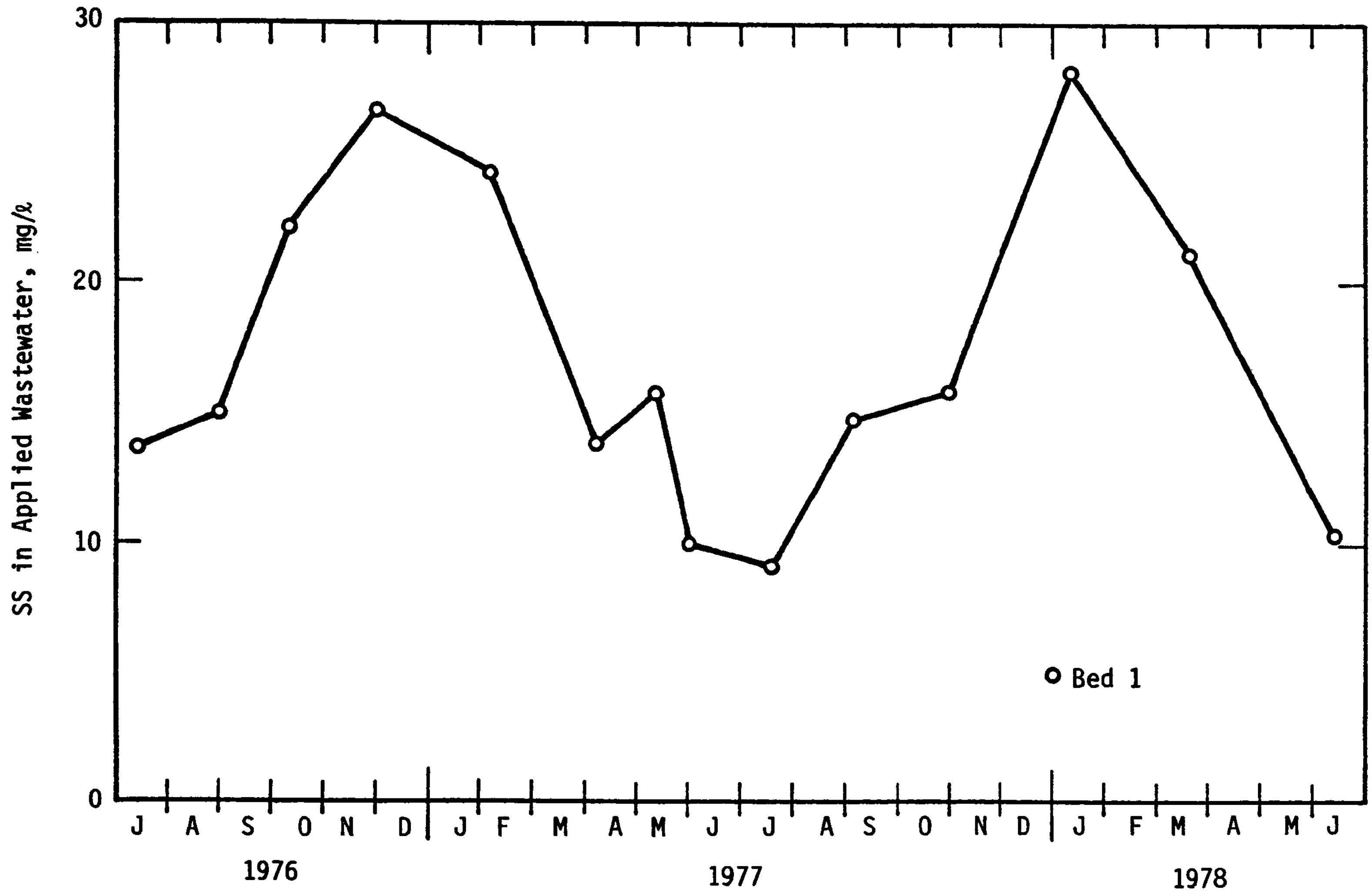


Figure 34. Suspended solids variation in applied wastewater.

In addition to the observed seasonal trends, nitrate and ammonium concentrations in the renovated water followed a general pattern within the individual loading cycles. The average nitrate concentrations tended to decline with successive loadings within a cycle. Figure 35 shows data selected from loading cycles that clearly illustrate this trend. The maximum nitrification occurred when oxygen penetrated deepest into the basins during maximum drying. For this reason, the maximum nitrate concentrations in the renovated water occurred during the first loading following the extended drying periods. Subsequent loadings during the cycle tended to limit the basin aeration and reduced the degree of nitrification that could occur. As a result, the basins produced lower nitrate levels as the loading cycle continued.

A reverse trend was observed with respect to the ammonium ion concentration, particularly during the winter loading cycles. This is shown for two cycles in Figure 36. As shown in this figure, the ammonium ion level in the product water increased with each successive inundation within a loading cycle. This trend was apparently caused by a reduction in the available ammonium adsorption capacity within the soil. Rejuvenation of the soil ammonium adsorption capacity generally occurred during the drying period, when the ammonium ions held on adsorption sites were nitrified. This nitrification freed the adsorption sites in the soil for readsorption of ammonium ions during subsequent loadings of the basin. When ample drying of the basin did not occur between inundations, rejuvenation of the soil ammonium adsorption capacity did not occur. In these cases, greater amounts of ammonium leached through the soil profile with each succeeding loading period, resulting in an increase in ammonium concentration in the renovated water with successive loadings of the loading cycle. The most obvious examples of this trend occurred during the winter when an abnormally high ammonium concentration was present in the applied effluent. During this time, the ammonium adsorption capacity was the most heavily taxed because of high ammonium loading and low rates of nitrification. Apparently, the ammonium adsorption capacity of the soil was not fully taxed during the other seasons, and the indicated trend did not occur except in the winter.

DISSOLVED SALT SPECIES

Water percolating through the infiltration-percolation soil system carried with it soluble salts. The dissolved inorganic constituents monitored in this study were hardness, calcium, magnesium, alkalinity, and chloride. An effort was made to correlate the concentrations of the various salt constituents leached from the soil with operating conditions and the seasonal variations of environmental conditions.

The average hardness concentrations of the renovated water, and the changes observed in hardness as the water percolated through the basins are shown in Figures 37 and 38, respectively. The most pronounced characteristic of both these curves was the peaking pattern observed during the period February-May, 1977. This peaking was apparently related to the addition of alum in the secondary treatment system for suspended solids control during

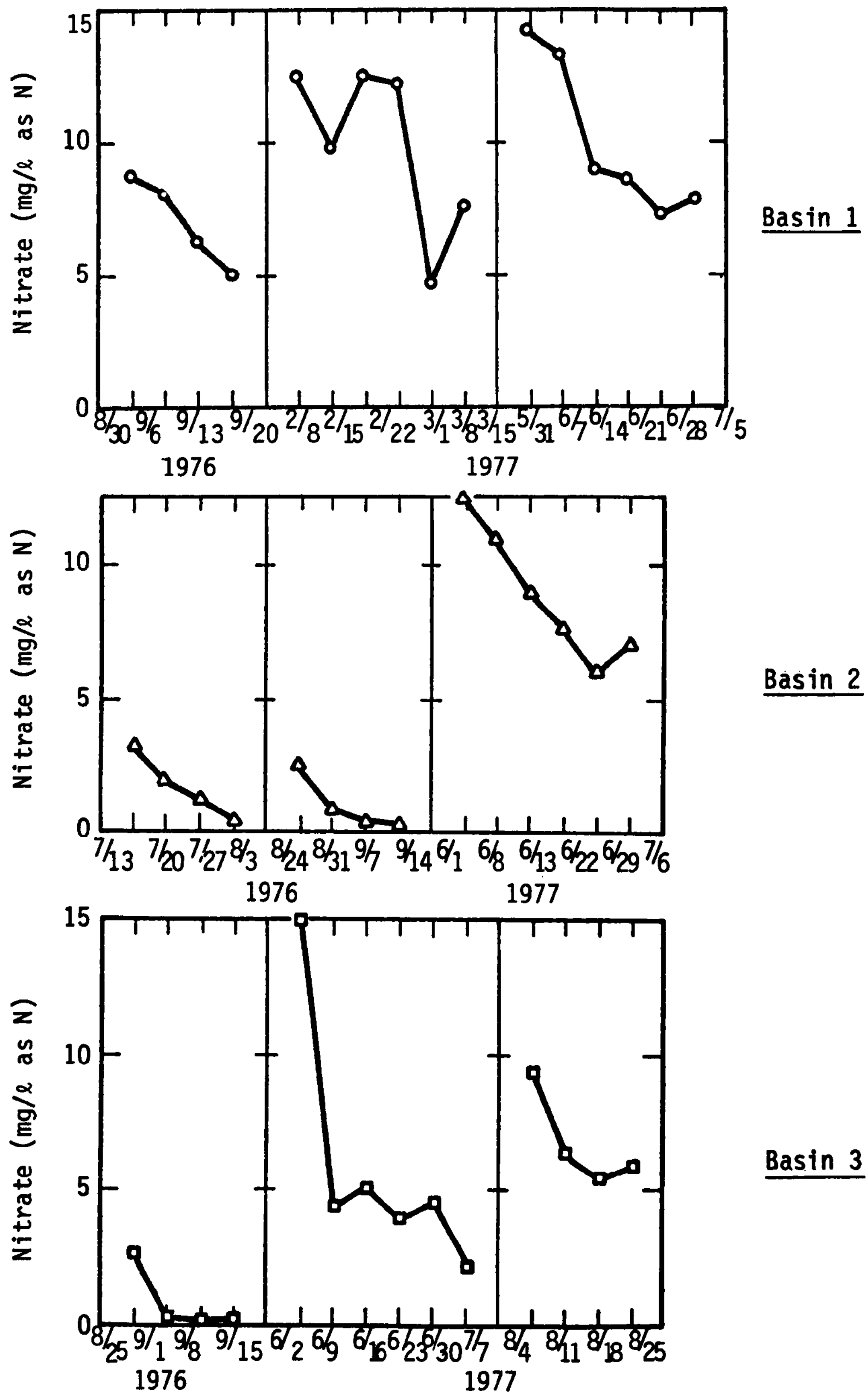


Figure 35. Nitrate variation in effluent within loading cycles.

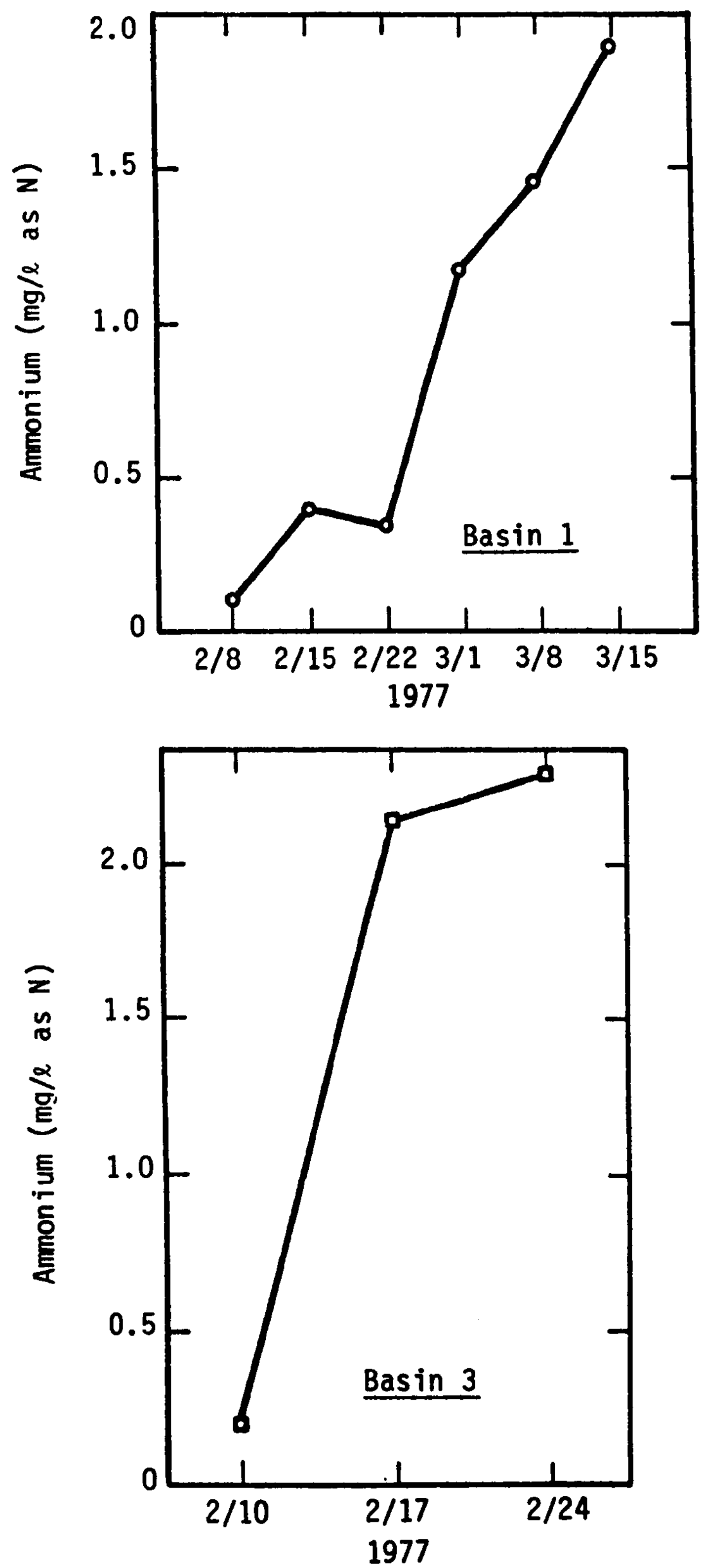


Figure 36. Effluent ammonium variations with loading cycles.

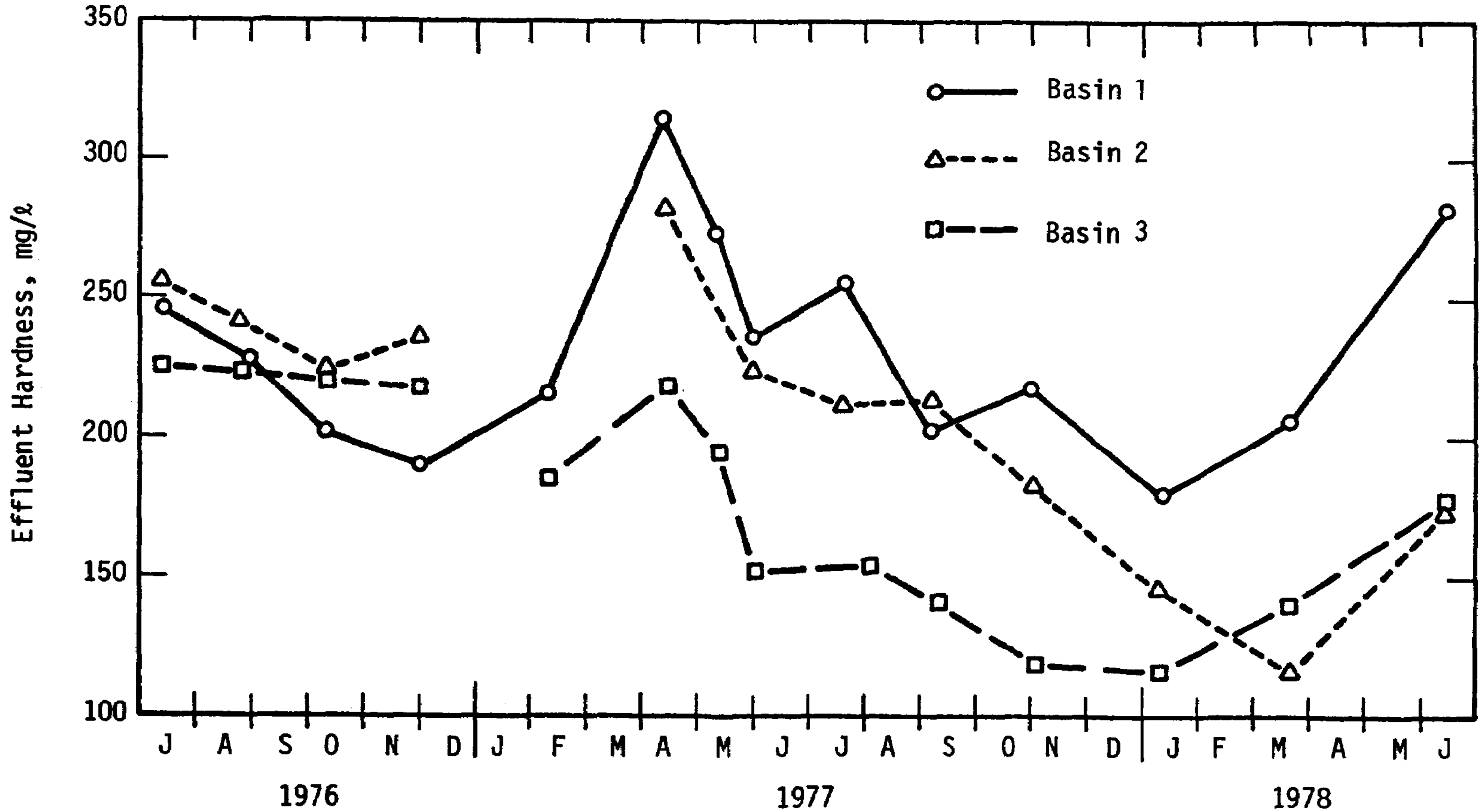


Figure 37. Effluent hardness as a function of time.

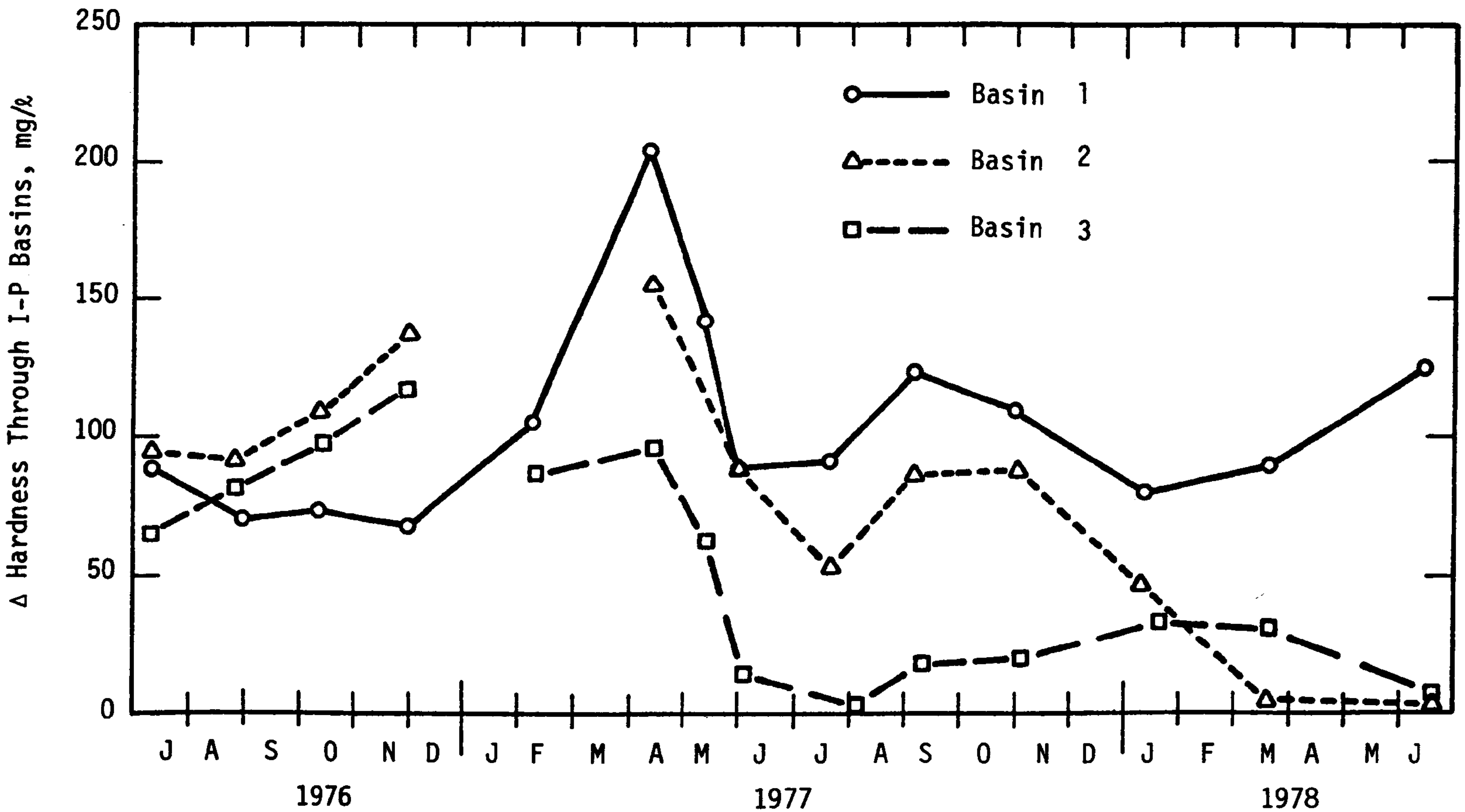


Figure 38. Change in hardness concentration through the basins.

this period. With the reduction in pH which accompanies alum addition, it was not surprising to observe increased leaching of hardness ions. The increased uptake of calcium and magnesium from the soil profile can also be seen by analyzing the changes in concentration of these two constituents from the influent to the effluent, as is shown in Figures 39 and 40.

Accompanying the release of the cations causing hardness, the alkalinity of the water collected in the underdrain system also increased markedly during the period February-May, 1977. This is shown very clearly in the curves of alkalinity change presented in Figure 41. The behavior of both the hardness and alkalinity constituents during the period of chemical addition apparently showed the impact of solution pH variations on the inorganic quality of the percolate water.

Apart from the unusual data developed during the spring of 1977, there seemed to be a general decline in the level of leaching of hardness and alkalinity from the soil profile. In fact, during the last two loading cycles of Basins 2 and 3, there was little change in the concentration of hardness and alkalinity from the influent to the effluent. This would seem to indicate that an equilibrium condition was being approached between the applied wastewater and the soil profile. The reasons for the deviation from this pattern in Basin 1 were not readily apparent.

The chloride concentration of the renovated water was monitored to assess the level of interaction between the ground water and the infiltration-percolation system. As can be seen in Figure 42, the chloride concentration in the water percolating through the basin seemed to show little change over the period of this study. The only significant exception to this was during the first six months of operation of Basin 3. During this period before the bed modification, the infiltration rate was very low, and the rate of application was continually reduced to the point that the equivalent loading in December, 1976 was only about 4.6 m/yr (15 ft/yr). With this low wastewater loading, dilution of the wastewater chloride concentration by the limited flow of ground water coming through the clay dike apparently made a detectable impact on the underdrain chloride concentration. When the basin surface was modified and the loading rate increased, this effect was not apparent.

COLIFORM ORGANISMS

During the early stages of the study, grab samples were collected periodically to monitor the performance of the infiltration-percolation system in removing bacterial organisms. The data which summarize these analyses are presented in Table 8. The influent values in this table represent averages for each of the three basins, while the effluent levels are specific for each basin. As indicated in these data, the basins were very effective in making a substantial reduction in the coliform concentrations. The efficiencies shown compare well with those which have been reported for similar high rate systems. Nevertheless, the effluent from each basin contained a significant concentration of residual fecal organisms. The difference in the effluent fecal coliform levels between the Basin 1 effluent and those in

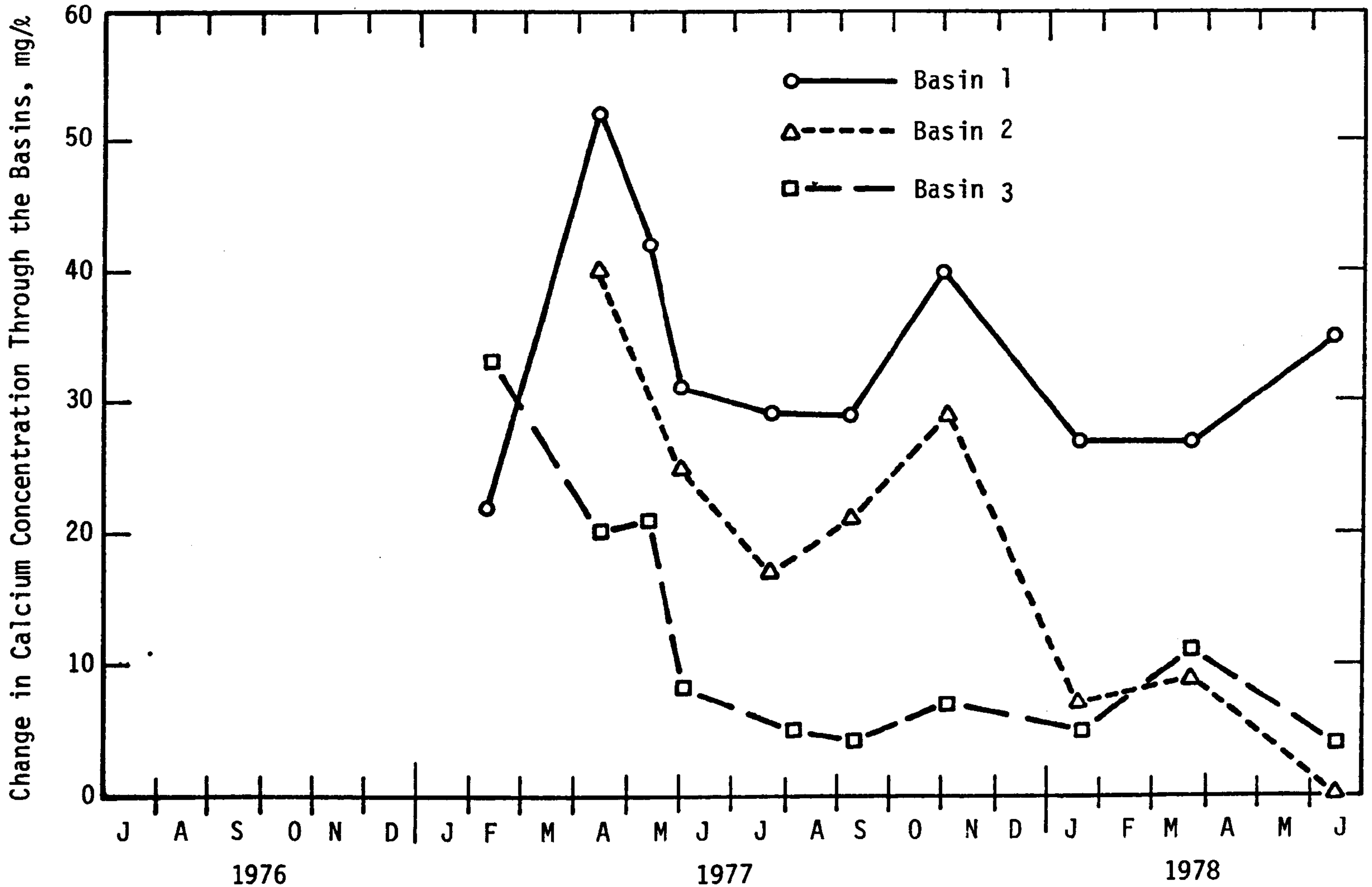


Figure 39. Change in calcium concentration through the basins.

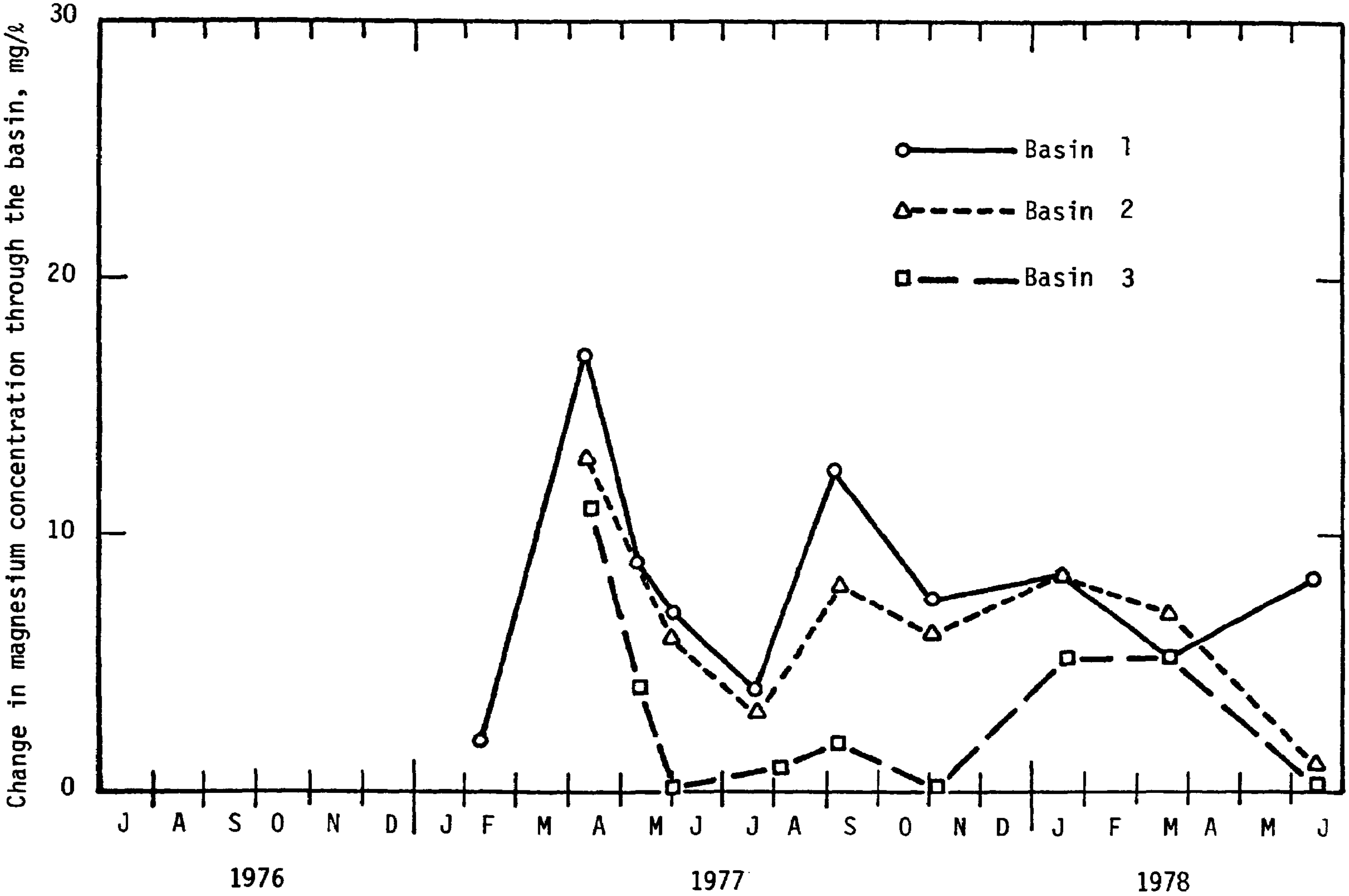


Figure 40. Change in magnesium concentration through the basins.

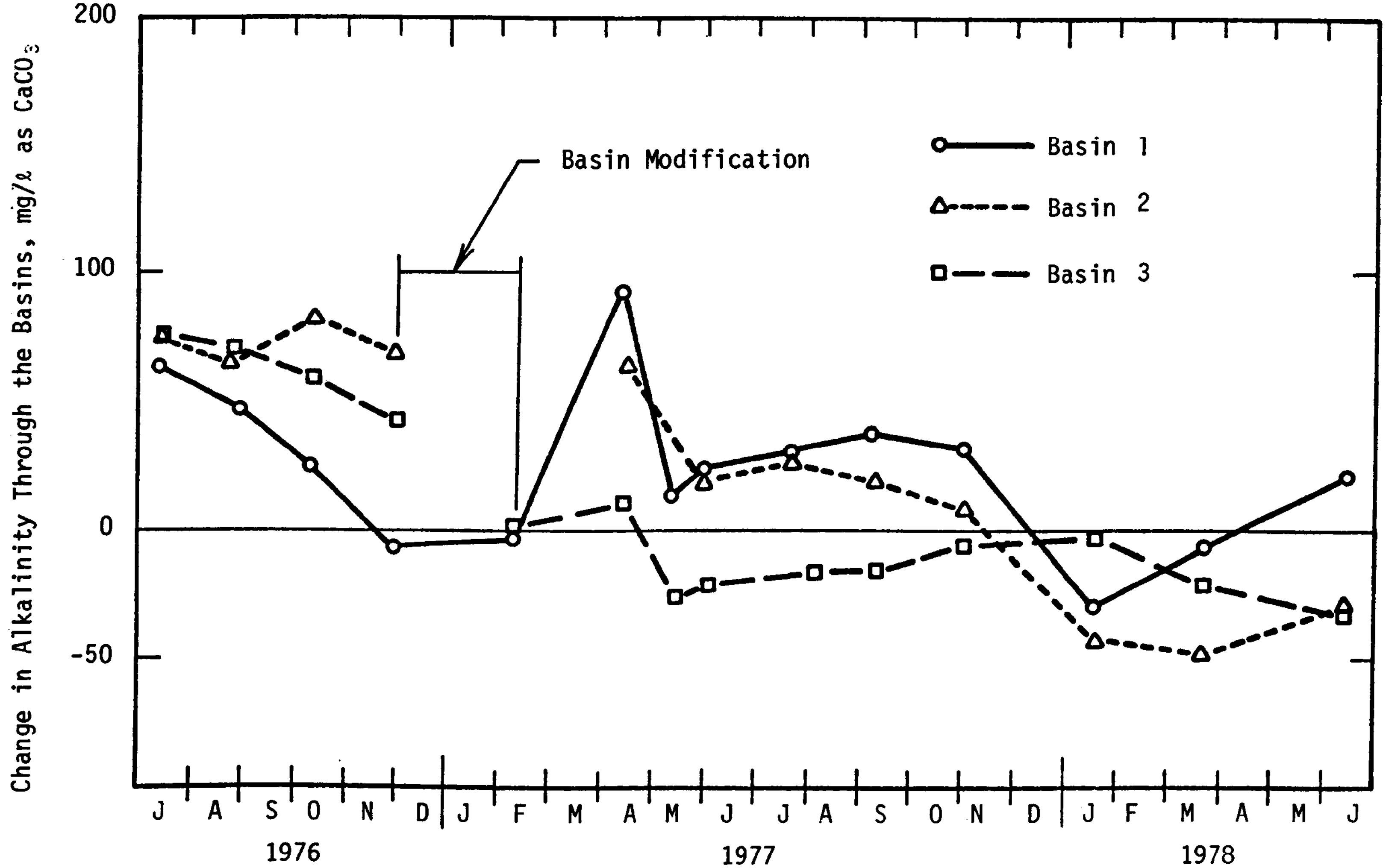


Figure 41. Change in alkalinity through the basins.

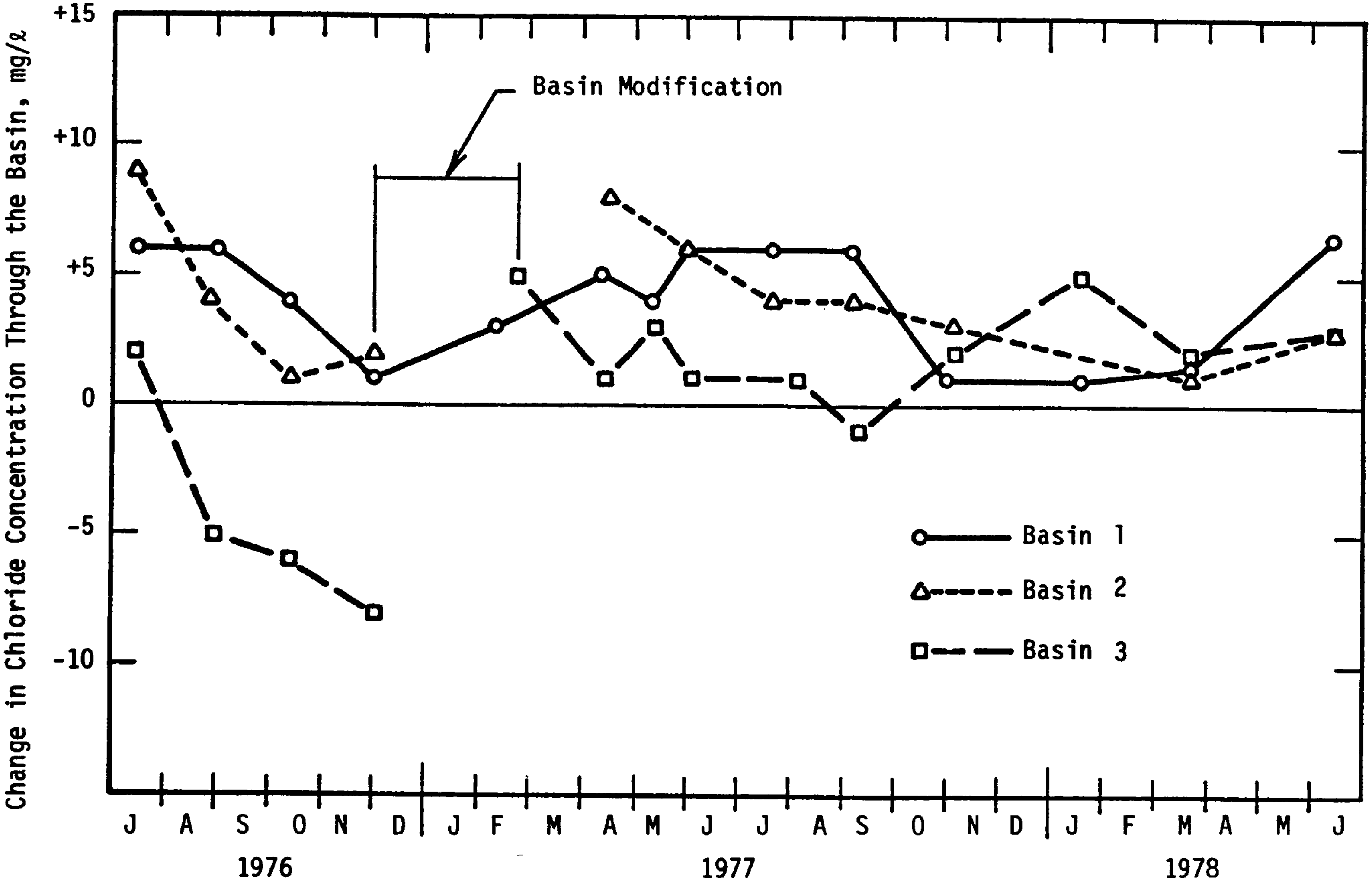


Figure 42. Change in chloride concentration through the basins.

TABLE 8. COLIFORM REMOVAL DATA

Parameter	Total Coliforms (per 100 ml)	Fecal Coliforms (per 100 ml)
Composite Influent Avg.	2.175×10^6	1×10^5
Effluent Avg.		
Basin 1	5700	3800
Basin 2	4025	900
Basin 3	4250	1025

% Removal	Total Coliforms	Fecal Coliforms
Basin 1	99.7	96.2
Basin 2	99.8	99.1
Basin 3	99.8	99.0

Basins 2 and 3 was likely related to the greater soil permeability and higher hydraulic loading rate in this basin during the period of collection of these data.

HEAVY METALS

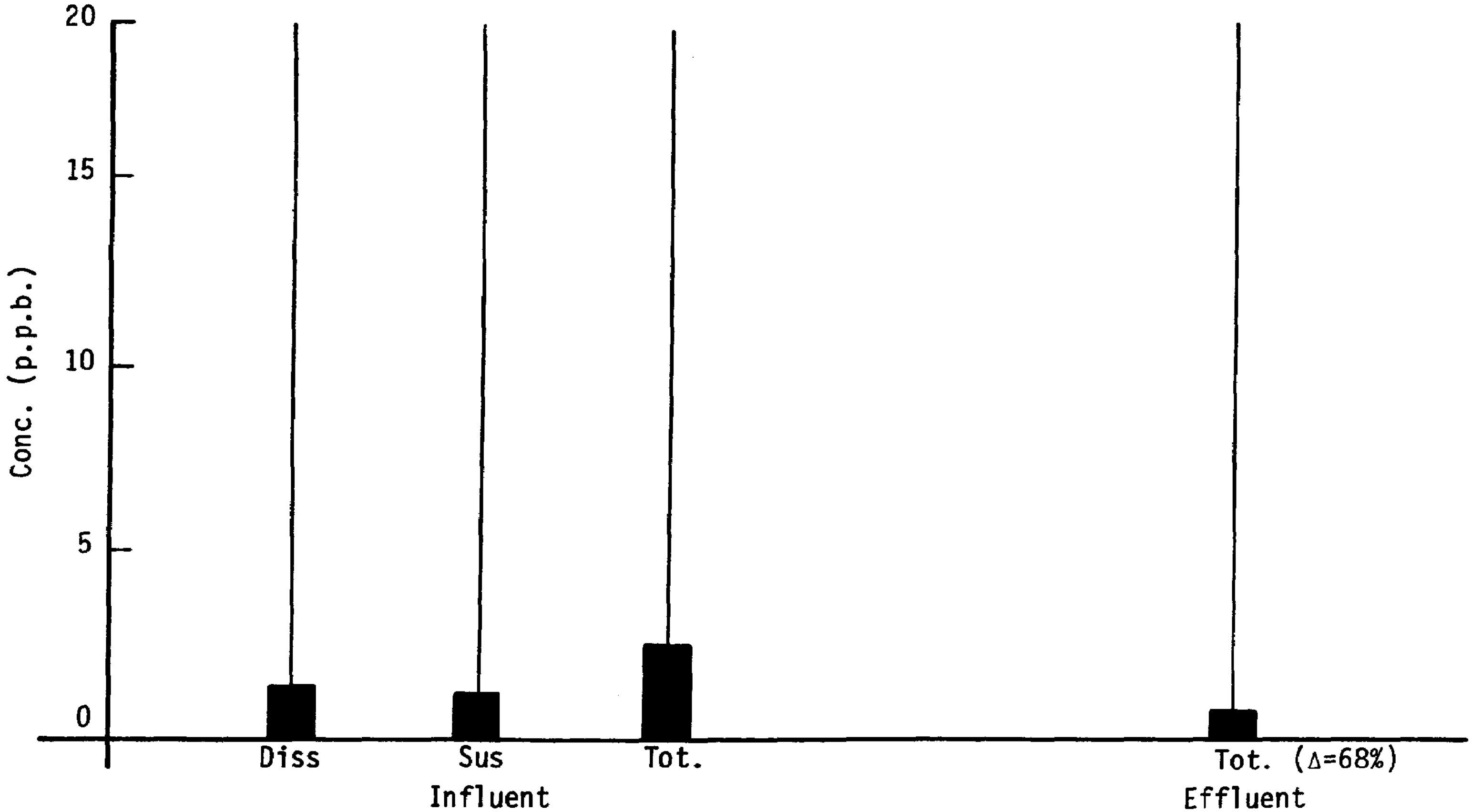
In a separate short term study phase, the removal of several important heavy metal wastewater constituents was assessed. The details of the research approach for this portion of the investigation have been presented previously in Section 5.

The results of this heavy metal investigation have been summarized in Figures 43 through 48. Each figure presents the concentrations of the dissolved, suspended, and total metals in the influent to the basins, and the total metal concentration in the effluent from the basins. The concentrations are averages of six determinations from each bed over a six week loading cycle. The data from all three of the basins were plotted on a single figure for each metal. The figures show the part per million concentrations as bar graphs with the means, standard deviations, and ranges tabulated below. The percent removal of each metal during infiltration-percolation was calculated and presented next to the mean value of the metal concentration in the effluent. The concentration of the metals are given in parts per billion (p.p.b.).

These data show that all of the metals studied were removed to some extent from the wastewater during the infiltration-percolation process. The levels of removal ranged from high efficiencies of greater than 80% for lead and zinc, to low levels of less than 50% for nickel and chromium.

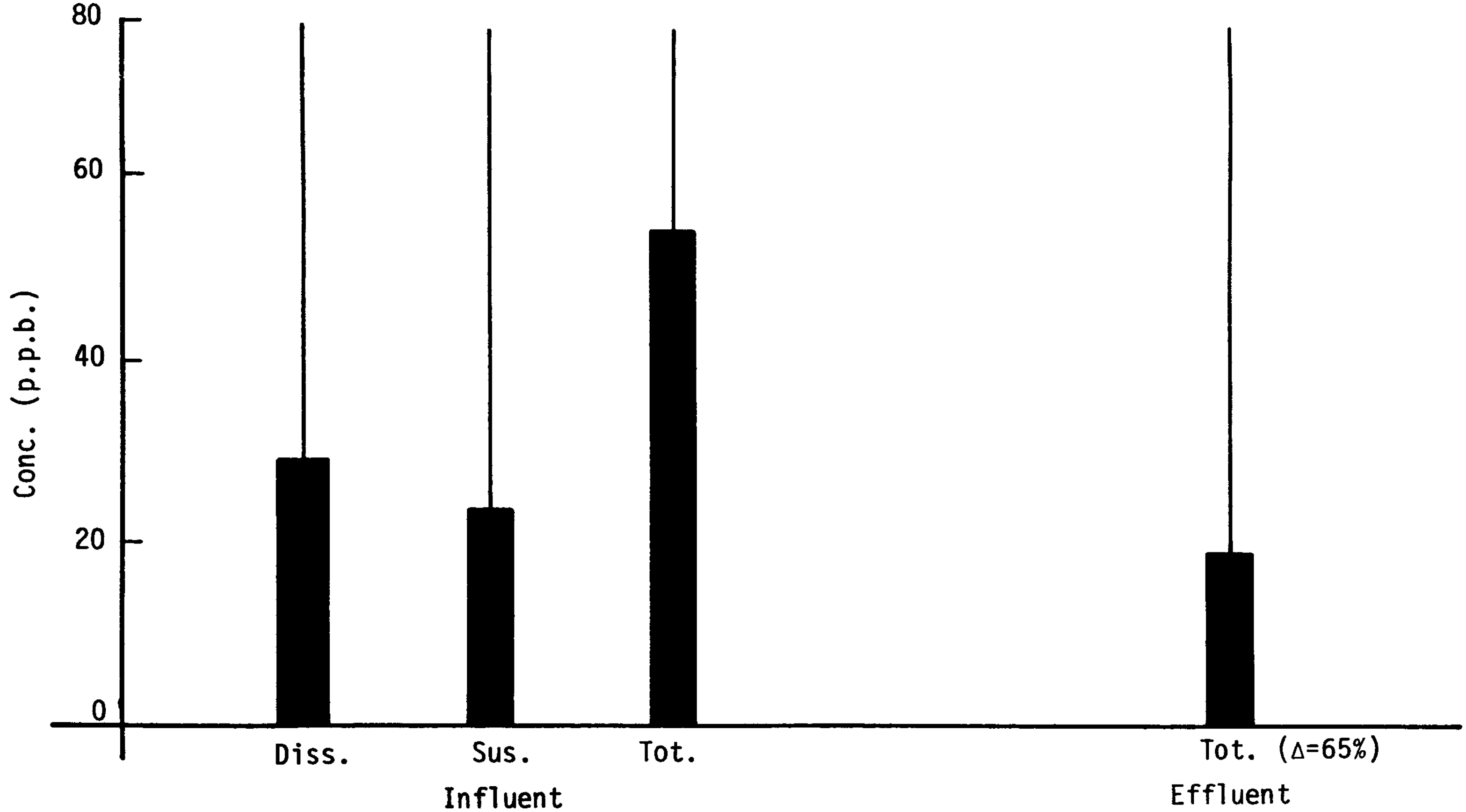
As seen in the previous figures, the metals which occurred in the Boulder wastewater were rather evenly distributed between the dissolved and suspended fractions. In fact, in several cases the measured mean concentrations of metals in the dissolved fraction exceeded those in the suspended fraction. This was not expected since it has been reported in the literature that heavy metals are usually strongly associated with the suspended fraction. However, in most cases the results which have been reported were from studies with either sludges or raw wastewater having much higher suspended solids concentrations.

The metals associated with the suspended material undoubtedly accounted for much of the observed removal. Since most of the suspended solids were removed from the wastewater during infiltration-percolation, it appears that a large portion of the metal removal observed in this study was a result of filtration of the solids. The remaining metal removal was likely due to a combination of exchange and precipitation reactions.



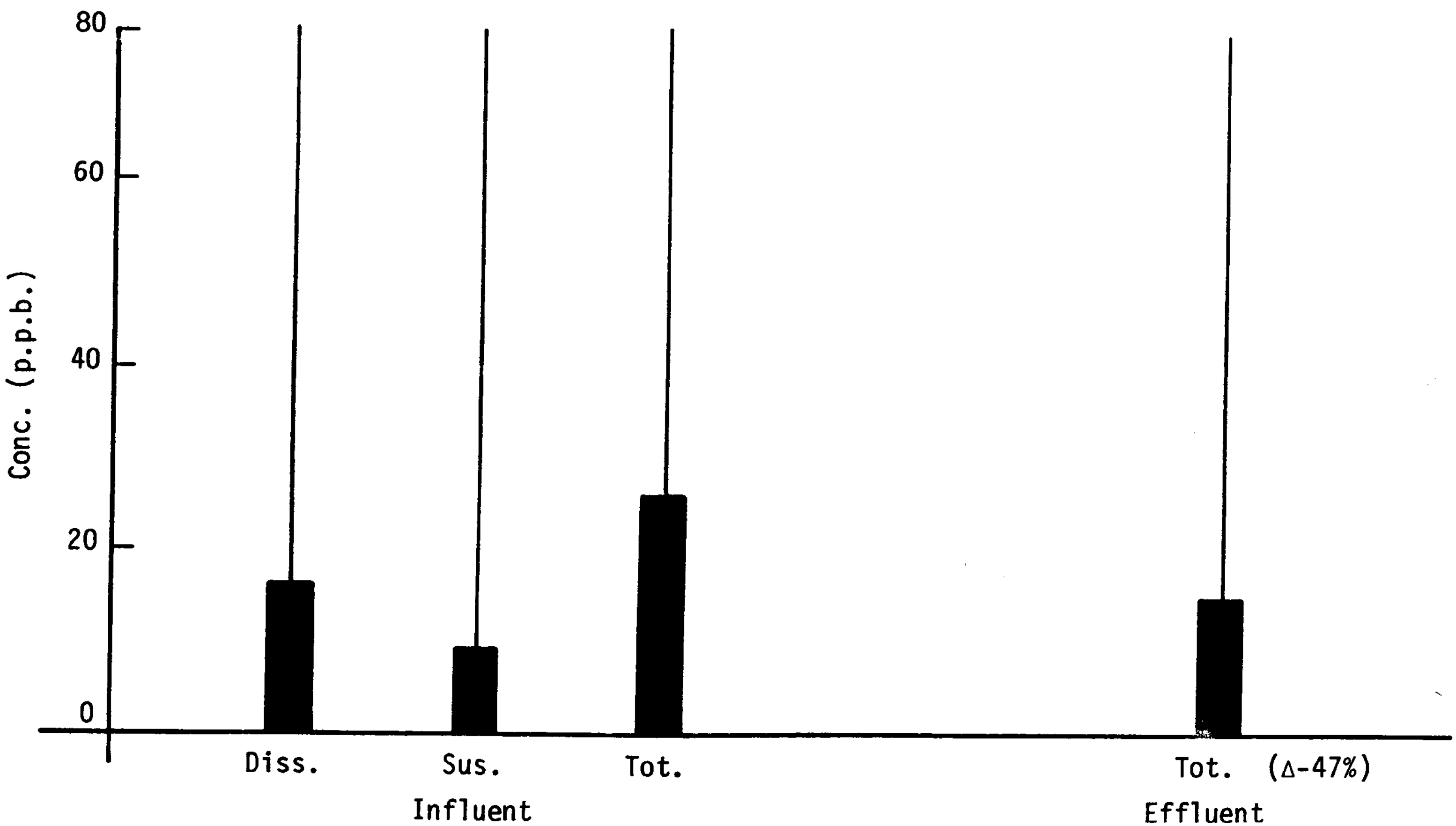
	Influent Diss	Influent Sus	Influent Tot.	Effluent Tot.
Mean	1.26	1.21	2.47	0.78
Stnd.Dev.	1.09	2.91	2.65	1.56
Range	4.50	11.00	11.00	6.00

Figure 43. Wastewater cadmium concentrations.



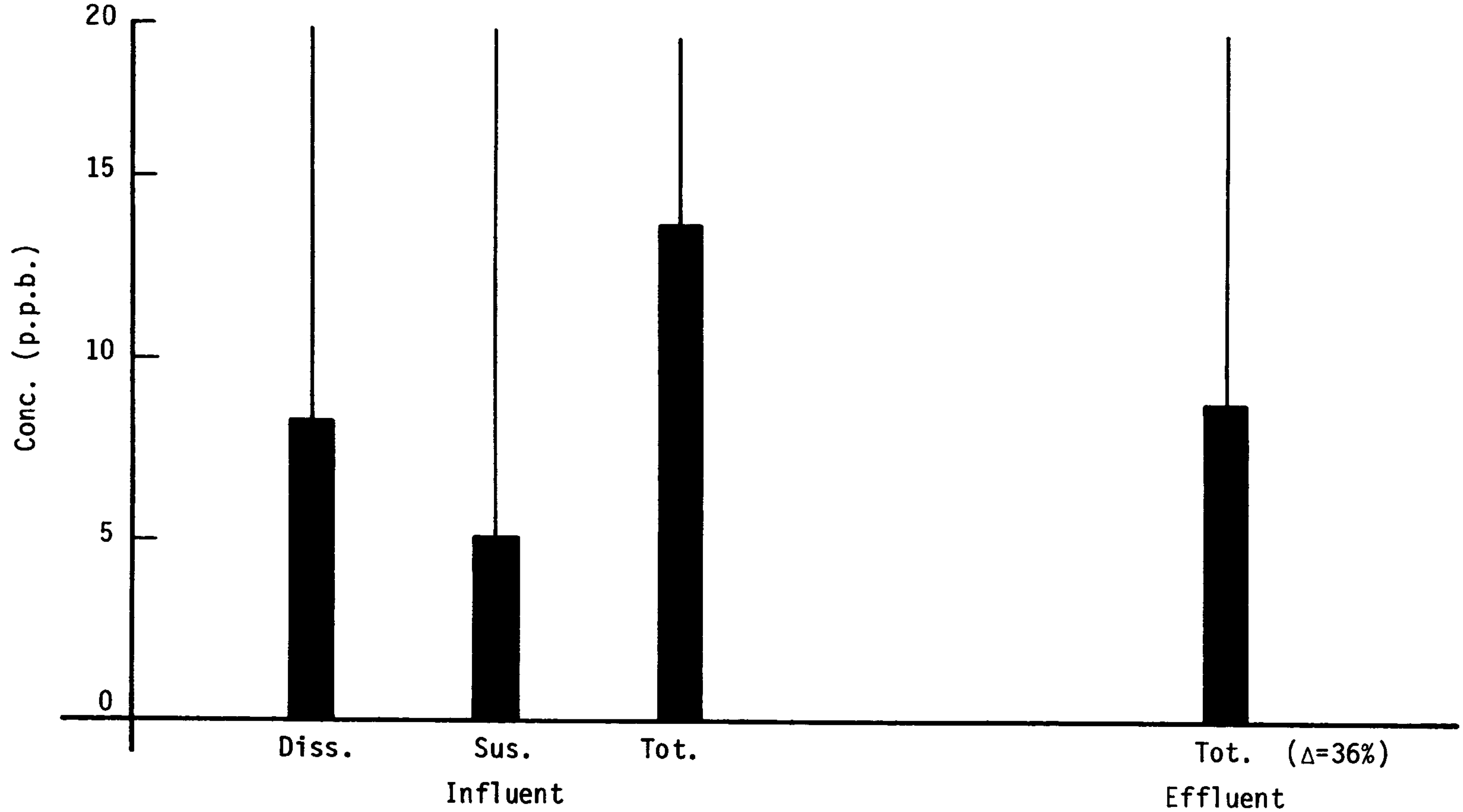
	Diss.	Sus.	Tot.	Tot.
Mean	19.75	24.27	54.02	18.75
Stnd.Dev.	13.83	9.67	16.05	8.72
Range	50.00	37.00	52.00	26.00

Figure 44. Wastewater copper concentrations.



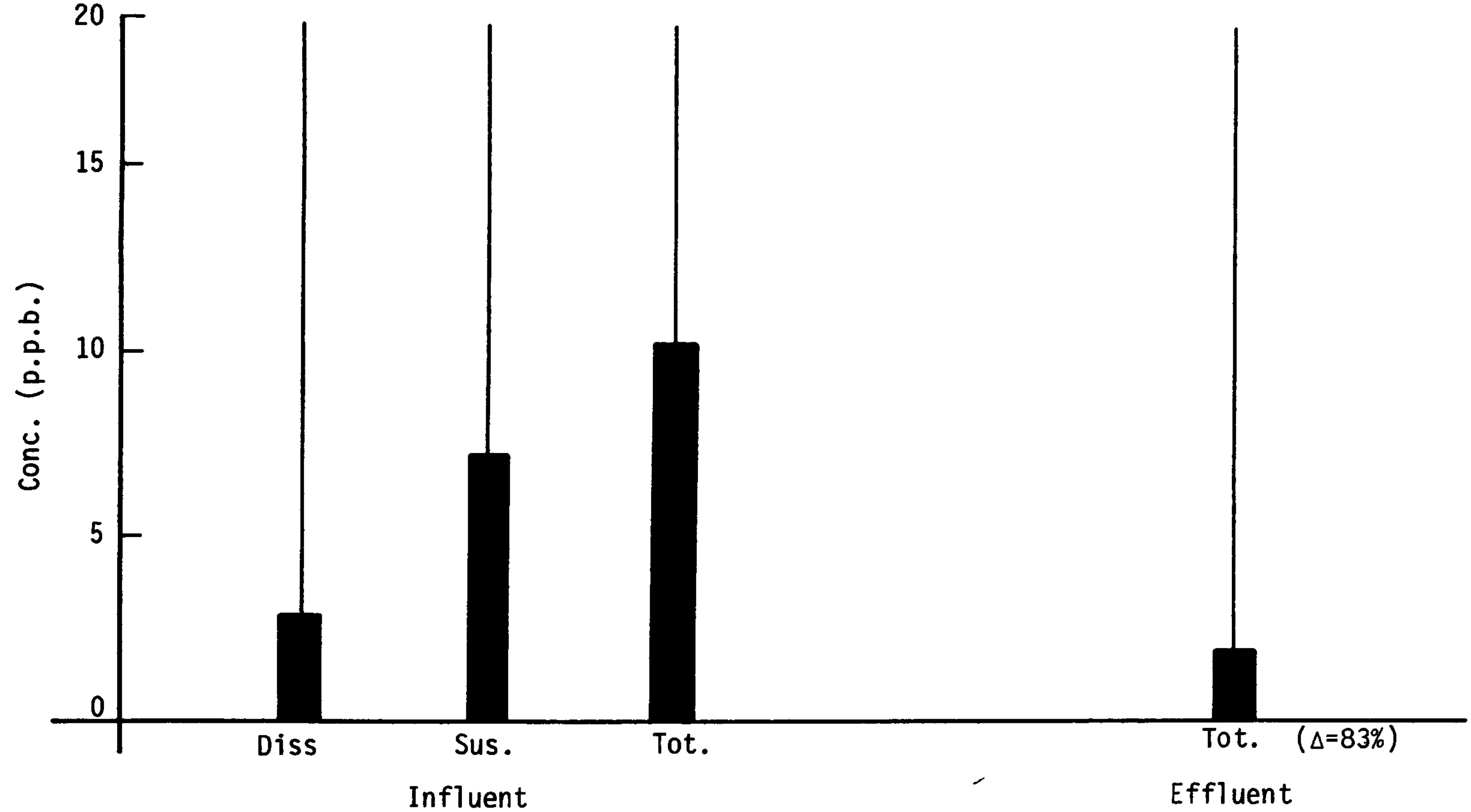
	Diss.	Sus.	Tot.	Tot.
Mean	16.93	9.00	25.93	14.60
Stnd.Dev.	14.11	7.38	14.10	9.97
Range	49.00	22.00	41.00	38.00

Figure 45. Wastewater chromium concentrations.



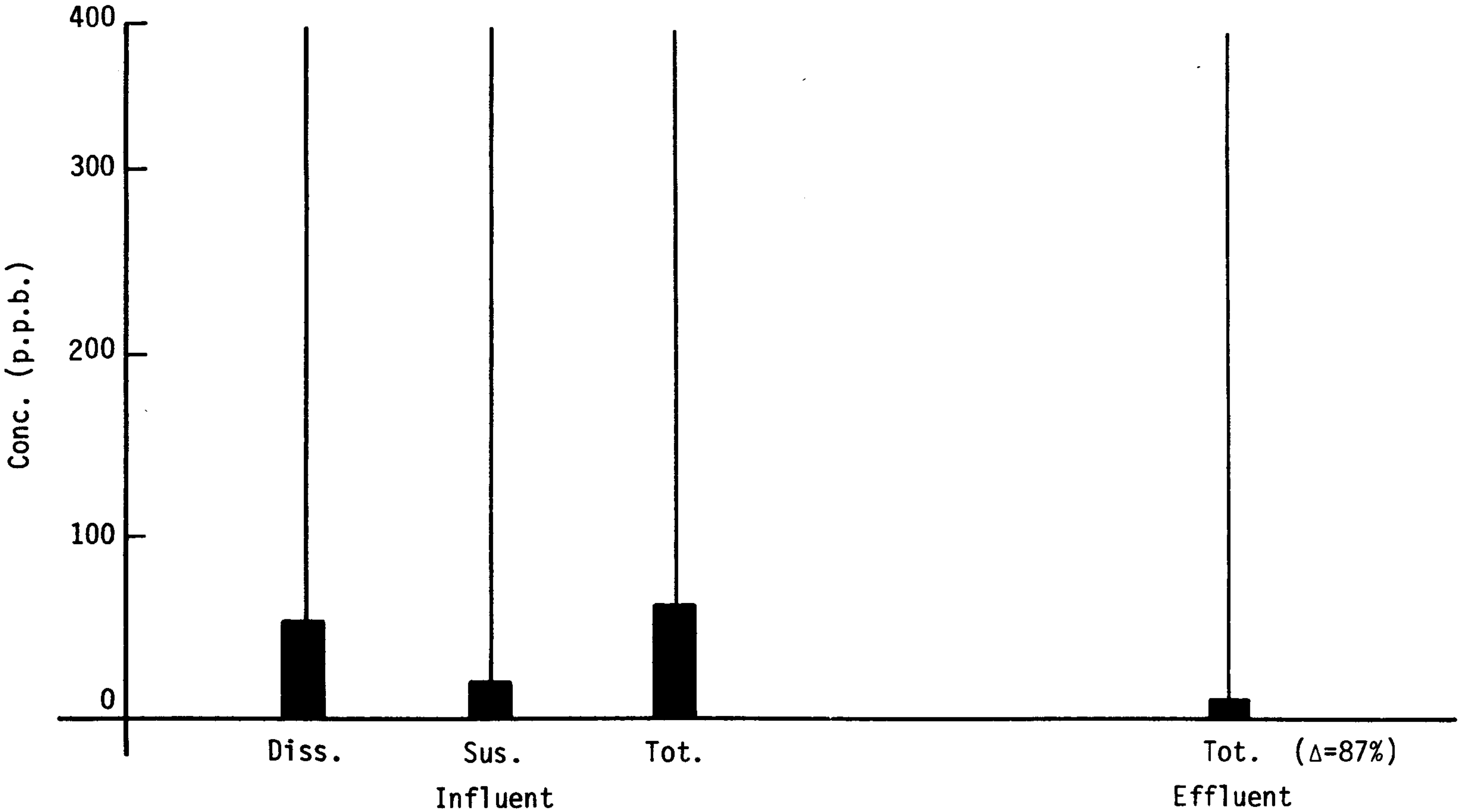
	Diss.	Sus.	Tot.	Tot.
Mean	8.59	5.27	13.86	8.94
Stnd.Dev.	5.58	4.36	8.31	4.55
Range	17.00	14.00	31.00	12.00

Figure 46. Wastewater nickel concentrations.



	Influent Diss	Influent Sus.	Influent Tot.	Effluent Tot.
Mean	2.88	7.27	10.15	1.75
Stnd.Dev.	2.55	3.41	3.75	2.54
Range	7.00	11.00	13.00	9.00

Figure 47. Wastewater lead concentrations.



	Influent Diss.	Influent Sus.	Influent Tot.	Effluent Tot.
Mean	49.17	11.11	60.28	7.94
Stnd.Dev.	41.06	11.67	35.76	15.92
Range	150.00	25.00	120.00	40.00

Figure 48. Wastewater zinc concentrations.

SECTION 8

COLUMN STUDIES

COLUMN OPERATION

In order to assess the patterns of constituent removal with soil depth, a small column was constructed to simulate the soil profile of Basin 1. The laboratory column consisted of a 3.7 meter (12 foot) section of 0.20 meter (8 inch) PVC pipe. It was capped on one end, and a 0.15 meter (6 inch) layer of gravel was placed in the bottom to serve as an underdrain system. A 2.1 meter (7 foot) layer of sand was placed directly above the gravel, and a 0.20 meter (8 inch) silt-loam layer was added on top of the sand. Sampling ports were placed at depths of 0.20 meters (8 inches), 0.56 meters (1 foot 10 inches), 1.2 meters (3 feet 10 inches), 1.8 meters (5 feet 10 inches) from the top, and at the bottom of the column.

The soil used for the laboratory column study was taken from throughout the depth of Basin 1. The top silt-loam layer was cohesive and facilitated collection of an undisturbed sample, while the sand was too loose to permit collection of undisturbed samples with depth. As a result, composite samples were collected at several basin depths and introduced into the column in the same order to simulate the gradation of soils found in the field.

The sample collection apparatus at each port consisted of a 65 mm funnel attached to a section of rubber tubing at the discharge end. To prevent clogging of the funnel, a small piece of fine screen was placed within the mouth of each funnel.

Since the amount of sample that could be collected at each sampling was quite small, it was necessary to make a series of column runs to adequately evaluate the soil column performance. This series consisted of several column loading cycles for assessing the organic removal behavior of the soil column. These were followed by two column runs for evaluating the removal of phosphorus in the soil. The following two loadings provided the data for evaluating the flow, nitrate, nitrite, and ammonium profiles. The next loading was tested for calcium, hardness, alkalinity, and chlorides, and a run for coliform and phosphorus analyses completed the series.

In each series of tests, 0.30-0.33 meters (12-13 inches) of secondary effluent was applied to the top of the column. During the wastewater application, a portable gravel layer was set on top of the soil while the wastewater was applied slowly through small holes in a plastic bag to prevent disturbance of the top loam layer.

COLUMN PERFORMANCE

The removal of COD, phosphorus, nitrogen, dissolved solids, and coliforms by the infiltration-percolation soil column was evaluated under a number of different loading cycles. The results of these evaluations have been summarized graphically in this section.

Chemical Oxygen Demand

The first column run was made to assess the removal of COD by the pilot soil column. Samples for testing the levels of COD removal were collected every 6-12 hours from each of the sample ports. A graphical representation of the results of this run has been presented in Figure 49. This figure indicates the COD concentration in the percolate water as a function of depth and time after loading.

In analyzing the data on this figure, it is apparent that one of the most significant characteristics of the figure was the large increase in the COD concentration during passage of the water through the top 0.23 meters (9 inches) of soil. In fact, the COD of water removed from the 0.23 meter (9 inch) sampling port never dropped below the soluble COD of the applied wastewater. Since the COD associated with suspended solids is typically removed in the top few inches of soil by filtering and straining, it was expected that the COD in the column would decrease very rapidly to a level below the soluble COD concentration of the influent (23). However, in this run there was no net removal of COD until the water had percolated through approximately 0.46 meters (18 inches) of soil. An increase of COD within the top few centimeters (cm) of soil has not been reported by previous investigators. However, it has been noted that the suspended solids are rapidly removed in the top few centimeters of soil by filtering and straining (23). As oxygen enters the soil during a subsequent drying period, the retained organics are usually decomposed (24). Since the oxygen diffusion rates are typically high at shallow depths, the trapped COD would be expected to decompose quickly (23). However, in this study the moisture content of the top soil layer was high during the entire loading schedule. This high moisture content may have slowed the rates of oxygen transport and thus affected the rates of organic decomposition (23, 25). This factor may have contributed to incomplete oxidation of the organics, resulting in some solubilization. Subsequent loadings of the secondary effluent could have flushed this soluble COD out of the top few centimeters of soil. In time, as the soluble organics were flushed from the soil, the COD concentration in the collected water would decrease. This pattern was observed to occur as can be seen by the curves developed at the later sampling times.

The COD increase in the upper levels of the column had no effect on the effluent quality. The high initial COD levels in the samples from the top sampling port were quickly reduced by the underlying sand layer in the column. The COD concentration decreased with depth to a constant COD concentration of approximately 20 mg/ℓ at the bottom of the column. The effluent level of 20 mg/ℓ of COD represented 50-75% removal of the applied COD.

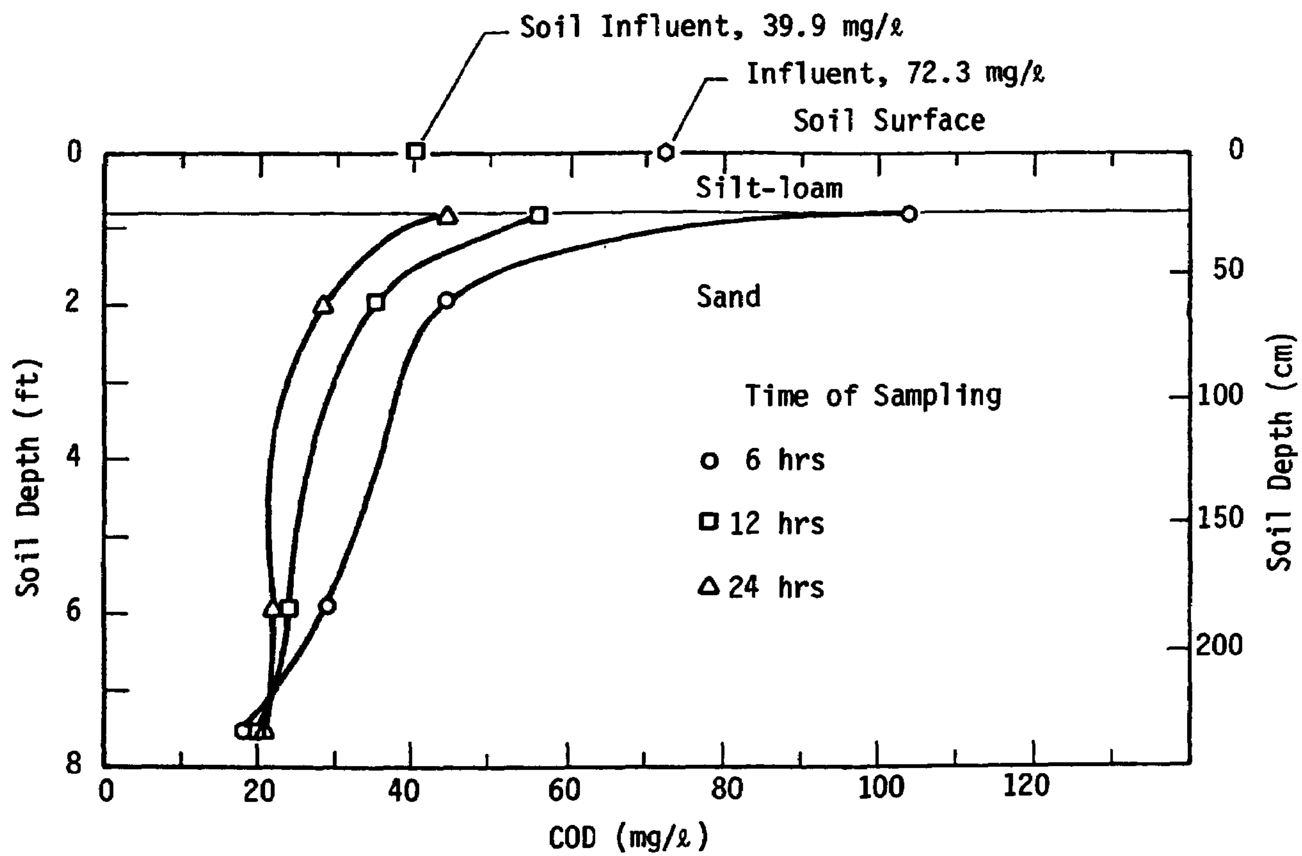


Figure 49. COD Concentration variation with column depth and time.

The results are consistent with those of previous studies and the actual field operations in this study (19, 20, 26).

Phosphorus

The behavior of phosphorus in the soil column was evaluated through three separate loadings of secondary effluent. Two of the phosphorus runs, Nos. 6 and 7, were made early in the loading history of the column, and the third was made after many column loading cycles. This was done to gain some insight regarding the phosphorus reduction capacity with repeated loading of a soil matrix. A graphical summarization of the phosphorus depth relationship has been presented in Figure 50.

The three curves plotted in Figure 50 represent the phosphorus concentration of the soil solution as a function of the soil depth for each of three separate loadings. In all three of these curves the phosphorus concentration of the applied wastewater declined sharply in the top silt loam layer. This highly efficient removal suggests that the silt loam provided numerous sites for phosphorus adsorption. The rate of phosphorus removal with increased depth declined significantly when the water reached the sand layer. This decline in the removal rate could be due to many factors, although the most probable cause is related to the characteristics of the sand media. Because of their relatively large particle size, sandy soils provide relatively few adsorption sites for phosphorus removal (27). As a result, the effectiveness of sand for phosphorus removal is not as great as that of finer textured soils.

While the results of the column study seemed reasonable, the effluents collected in the field operation had phosphorus concentrations which were significantly higher than those determined in the laboratory column study. Since the soil used in the column was taken directly from Basin 1, it was thought that changes in the soil matrix must have occurred. The soil samples taken for the column study were obtained in the middle of January, 1977, while the column construction was not completed until the end of June, 1977. This resting period may have provided time for nearly all of the adsorbed phosphorus to become mineralized (28). The result of this phosphorus precipitation was the freeing of adsorption sites for the subsequent fixation of phosphorus within the column.

Nitrogen

In a separate loading cycle, secondary effluent was applied to the soil matrix for analysis of the behavior of the nitrogen species. The data obtained from this run consisted of time profiles at each sampling port of the ammonium, nitrite, and nitrate concentrations, and the flow quantities.

The ammonium level of the applied wastewater was very low in this phase of the study, containing only 5.44 mg/ℓ. As can be seen in Table 9, the ammonium concentration decreased rapidly to less than 1.0 mg/ℓ as the water moved through the silt loam layer. The concentration decreased further when the water entered the sand region. Sampling Port No. 3 yielded ammonium

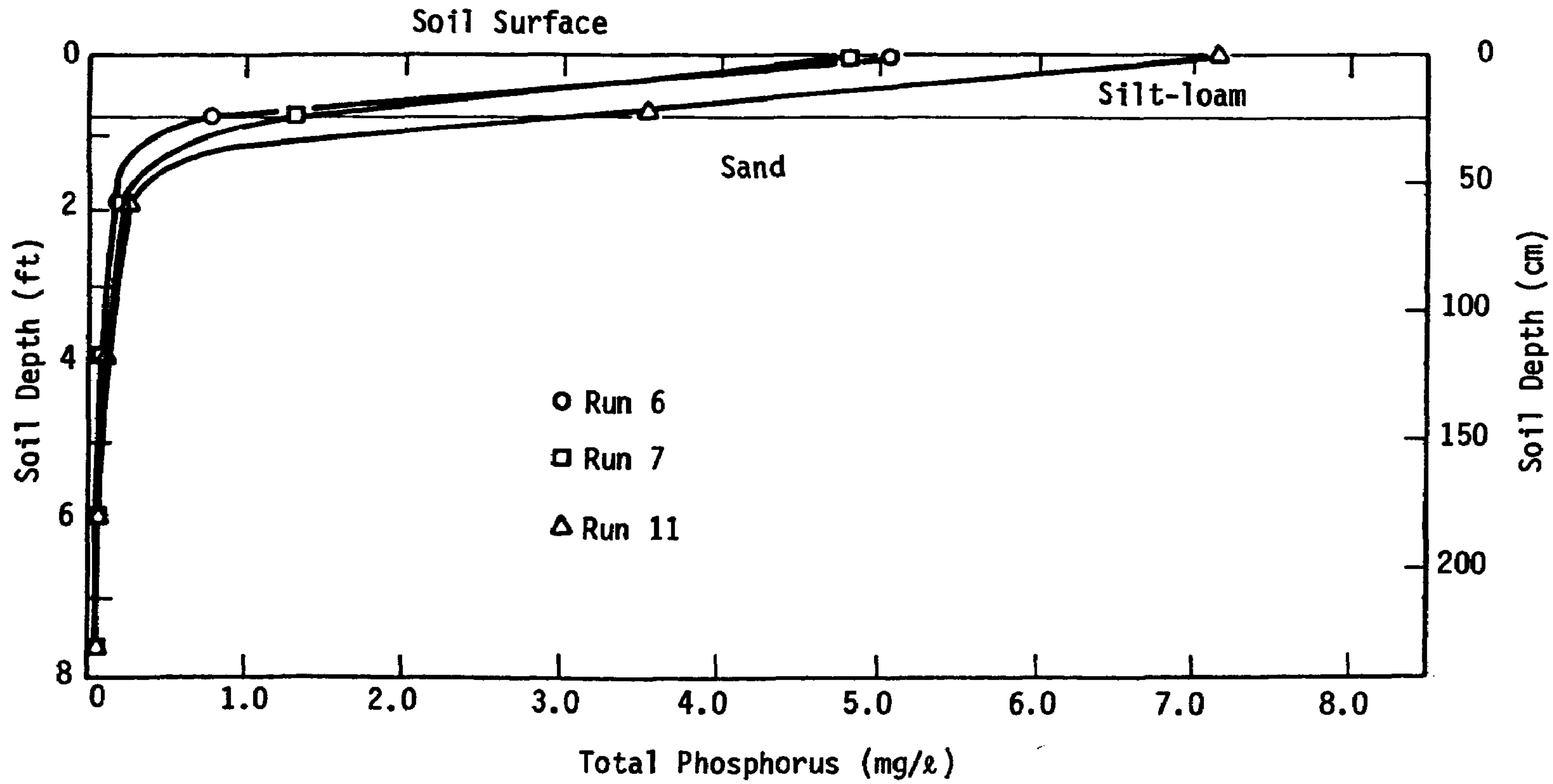


Figure 50. Wastewater phosphorus concentration as a function of soil depth.

TABLE 9. SELECTED NITROGEN REMOVAL DATA (mg/ℓ)

Sample Port	Collection Time Hrs.	Min.	Nitrite	Nitrate	Ammonia
Influent			.48	1.63	5.44
Port 1	3	00	.46	16.59	.92
	5	30	.70	14.00	.77
	8	55	.14	8.75	-
	19	40	.02	3.08	.22
Port 3	3	00	.02	9.95	.23
	5	30	.01	12.67	.09
	8	55	.01	13.61	.08
	19	45	.02	14.48	.07
Port 5	3	00	.05	29.34	-
	5	30	.01	34.02	.08
	9	00	.01	30.42	.06
	19	50	.01	22.18	.07

levels in the percolate water of less than 0.10 mg/ℓ. This indicates that most of the ammonium ion was adsorbed or fixed in the upper one-half of the column depth, with little change in the ammonium levels noted below that point.

As seen in Table 9, the nitrite level of the influent wastewater was 0.48 mg/ℓ as N. At the first sampling level this concentration had increased slightly to levels ranging from 0.32-0.71 mg/ℓ as N. Below the first sampling port the nitrite concentration dropped to less than 0.05 mg/ℓ and remained at a very low level throughout the rest of the soil matrix. The increased levels of nitrite at the top sampling port would point to the initiation of nitrification in the applied wastewater, or incomplete nitrification during the previous drying period (25). As shown by the extremely low nitrite concentrations at Port No. 2, a majority of the nitrite produced in the silt loam layer was quickly converted into the nitrate form.

The nitrate form of nitrogen was present at significantly higher concentrations than the other nitrogen forms throughout the column depth. The time profiles of the nitrate concentrations in the water from the various ports are shown in Figure 51. As can be seen from this figure, the concentrations of nitrate in the column increased as the water moved through the soil. In addition, the shape of the nitrogen discharge curves from Port No. 1 and the effluent, displayed the characteristic first flush nitrate peaking which has been reported from several other studies of cyclically loaded soil systems (10, 29).

Flow measurement on the column effluent provided the data for the curve of discharge as a function of time in Figure 52. These data were used in conjunction with the effluent nitrogen data to provide the nitrogen mass discharge pattern shown in Figure 53. The pattern shown in these figures compared favorably with that of the actual field experience.

Dissolved Solids

In the portion of this column study dealing with the behavior of dissolved solids in soil systems, analyses were made for calcium, magnesium, total hardness, alkalinity, and chlorides. These determinations were made on the total water volume collected from each port during one loading of secondary effluent. The data describing the fate of these constituents have been summarized graphically in Figures 54 through 58. Each figure presents the variation in concentration of the dissolved material with soil depth.

The curves of alkalinity, calcium, magnesium, and total hardness displayed similar variations of concentration with depth. The concentration of each of these constituents increased substantially in the top portions of the soil. The rate of concentration increased, then gradually declined within the 0.61-1.2 meter (2-4 foot) level. At depths greater than 1.2 meters (4 feet) the concentrations of these materials remained essentially constant.

A slight difference in the concentration behavior of these four constituents was noted in the top silt loam layer. The concentrations of hardness and calcium increased significantly as the wastewater moved through the fine

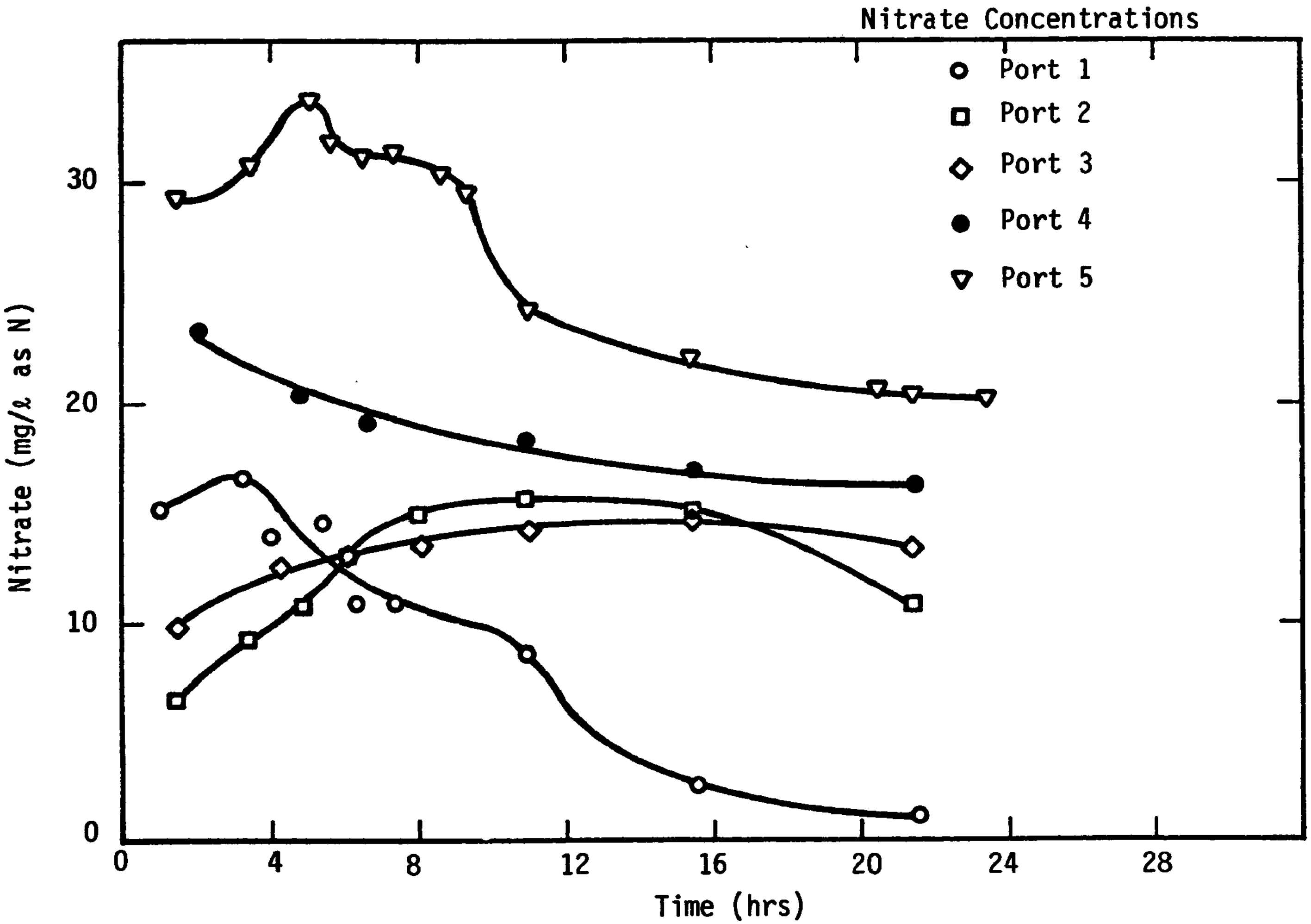


Figure 51. Nitrate concentration as a function of column depth and sampling time.

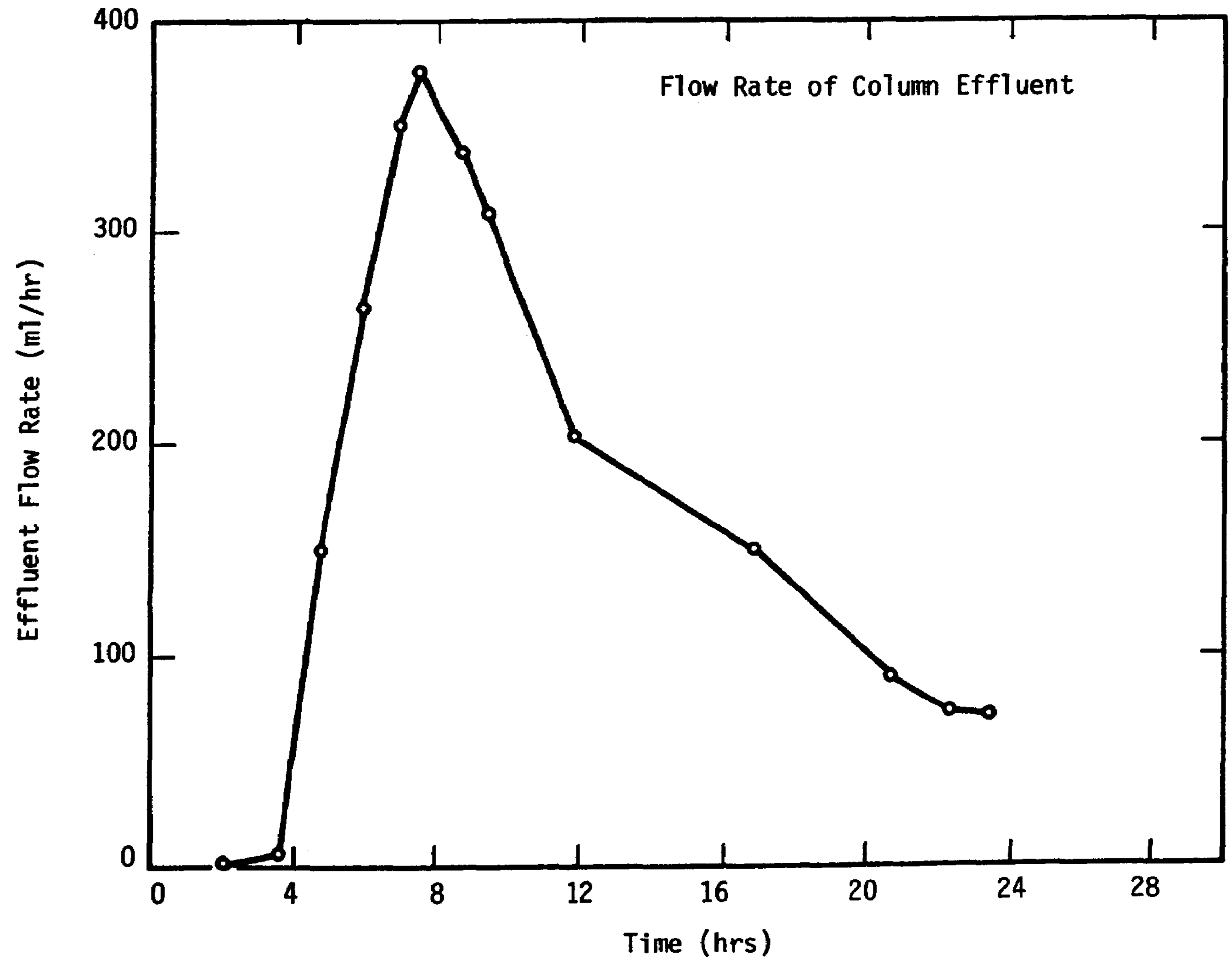


Figure 52. Column underdrain flow as a function of time after loading.

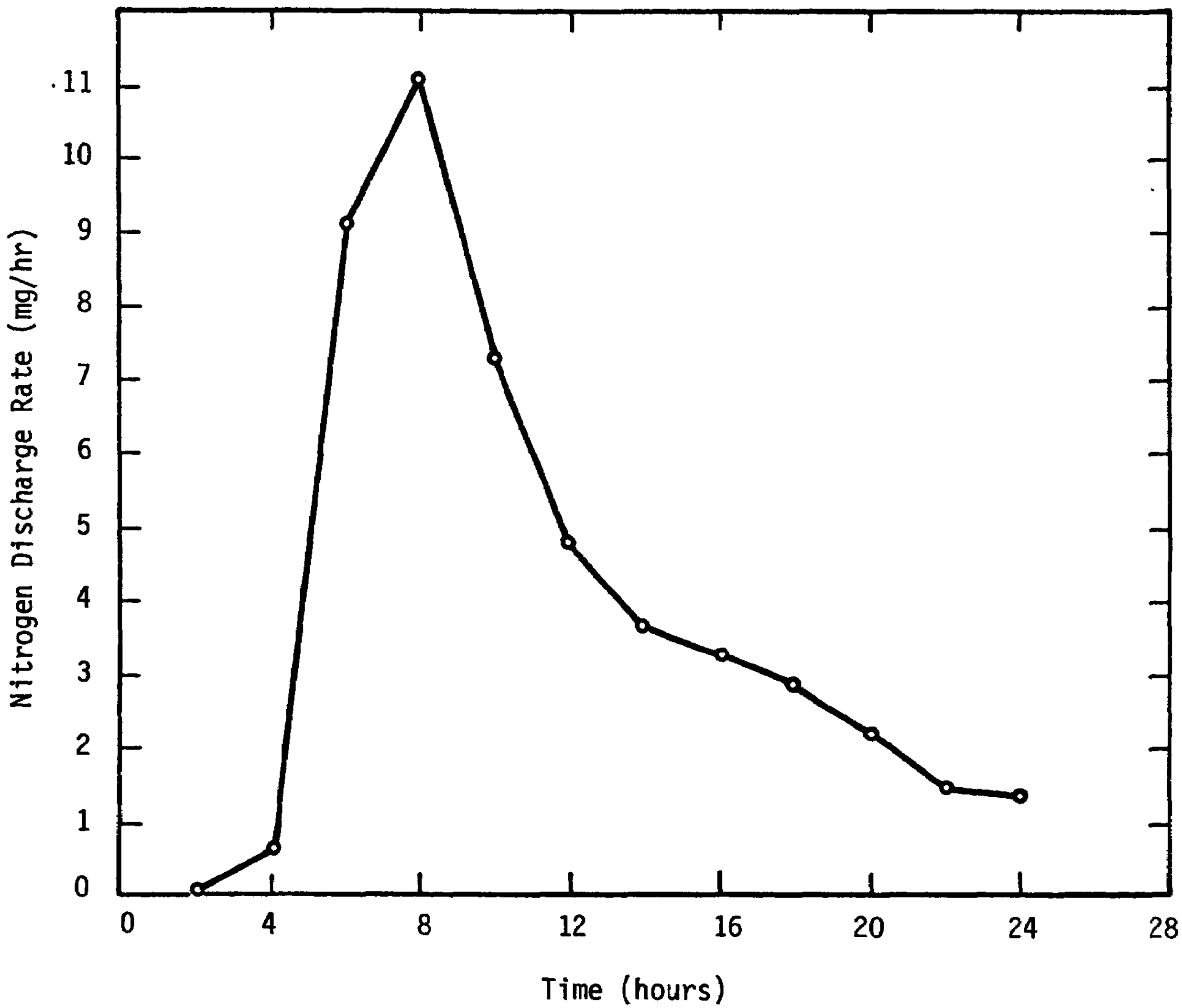


Figure 53. Mass flow of nitrogen from column as a function of time after loading.

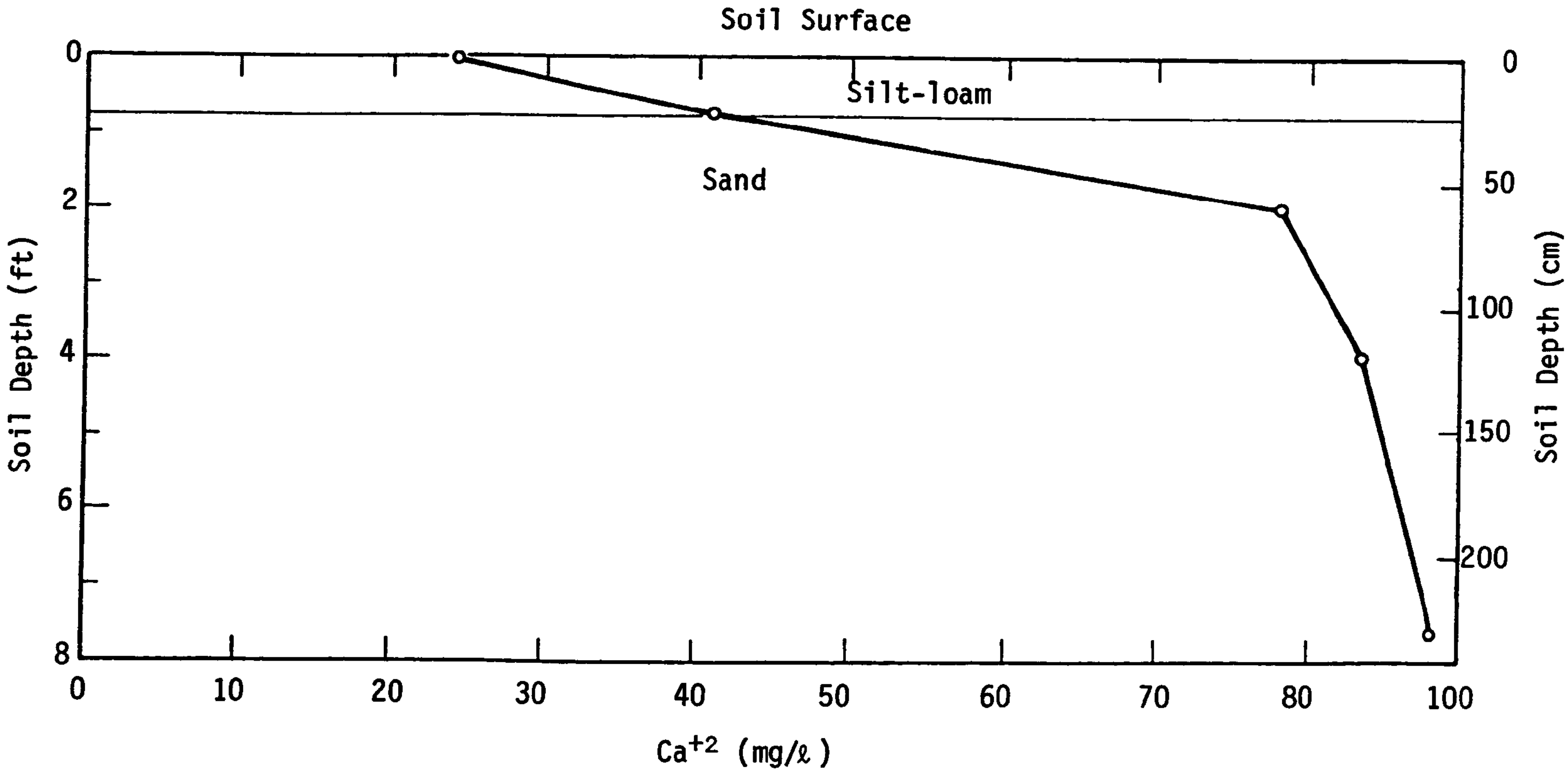


Figure 54. Wastewater calcium concentration as a function of column depth.

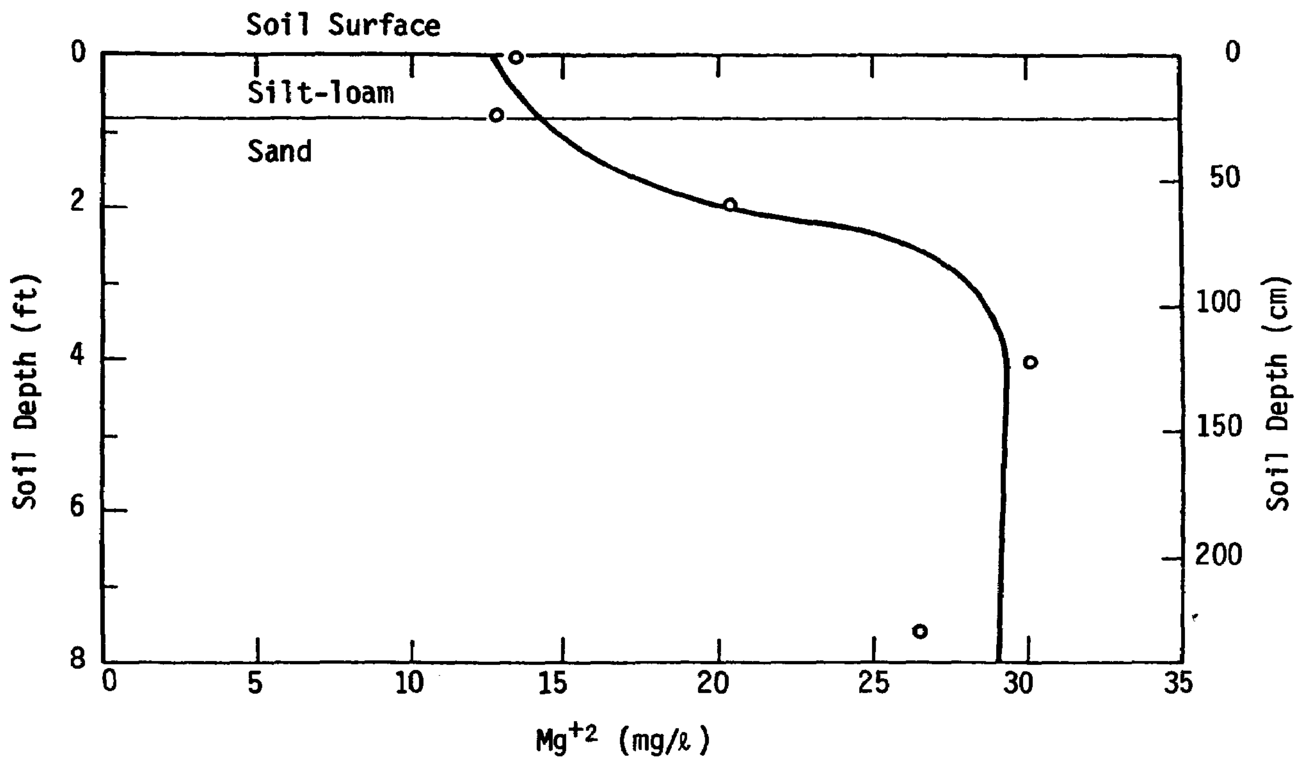


Figure 55. Wastewater magnesium concentration as a function of column depth.

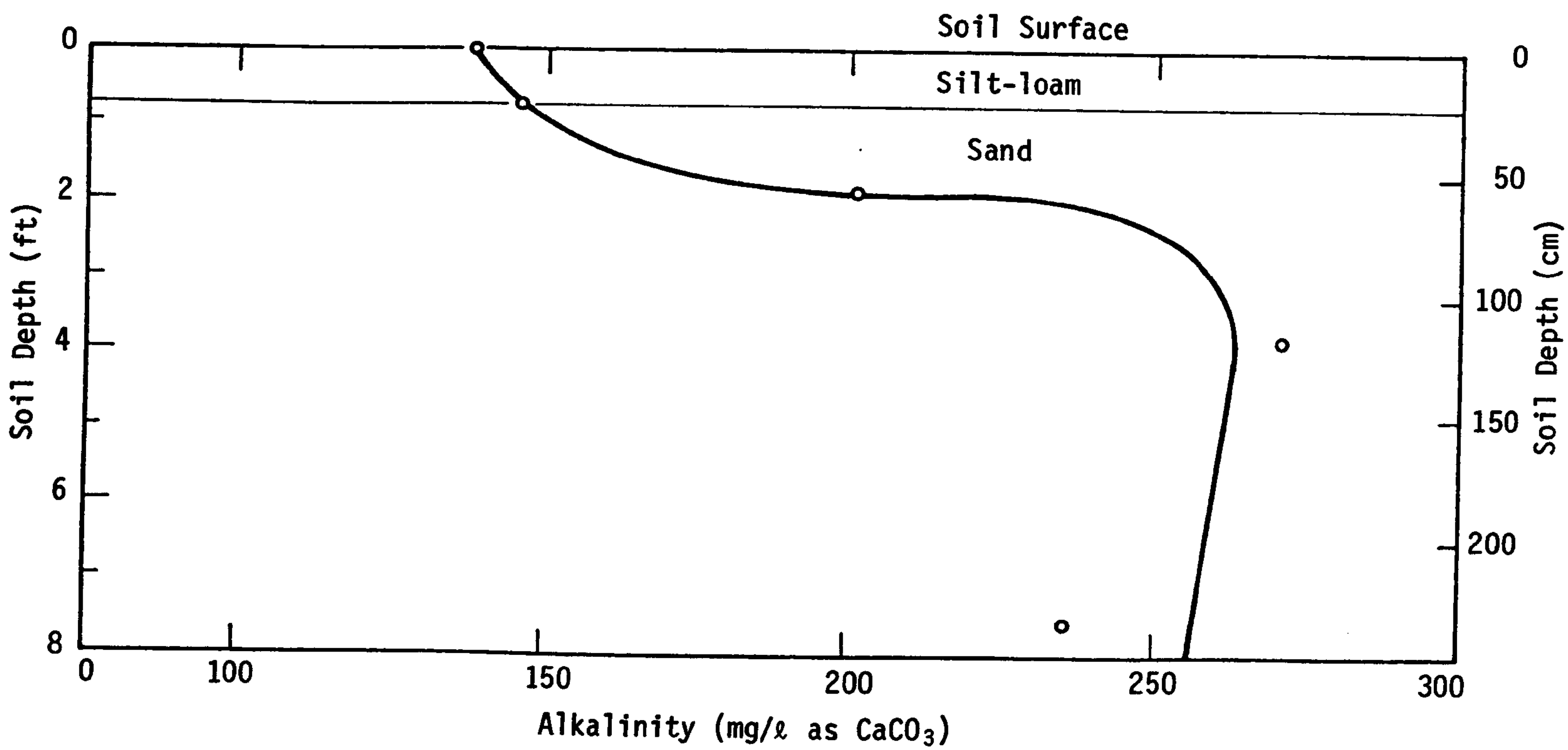


Figure 56. Wastewater alkalinity as a function of column depth.

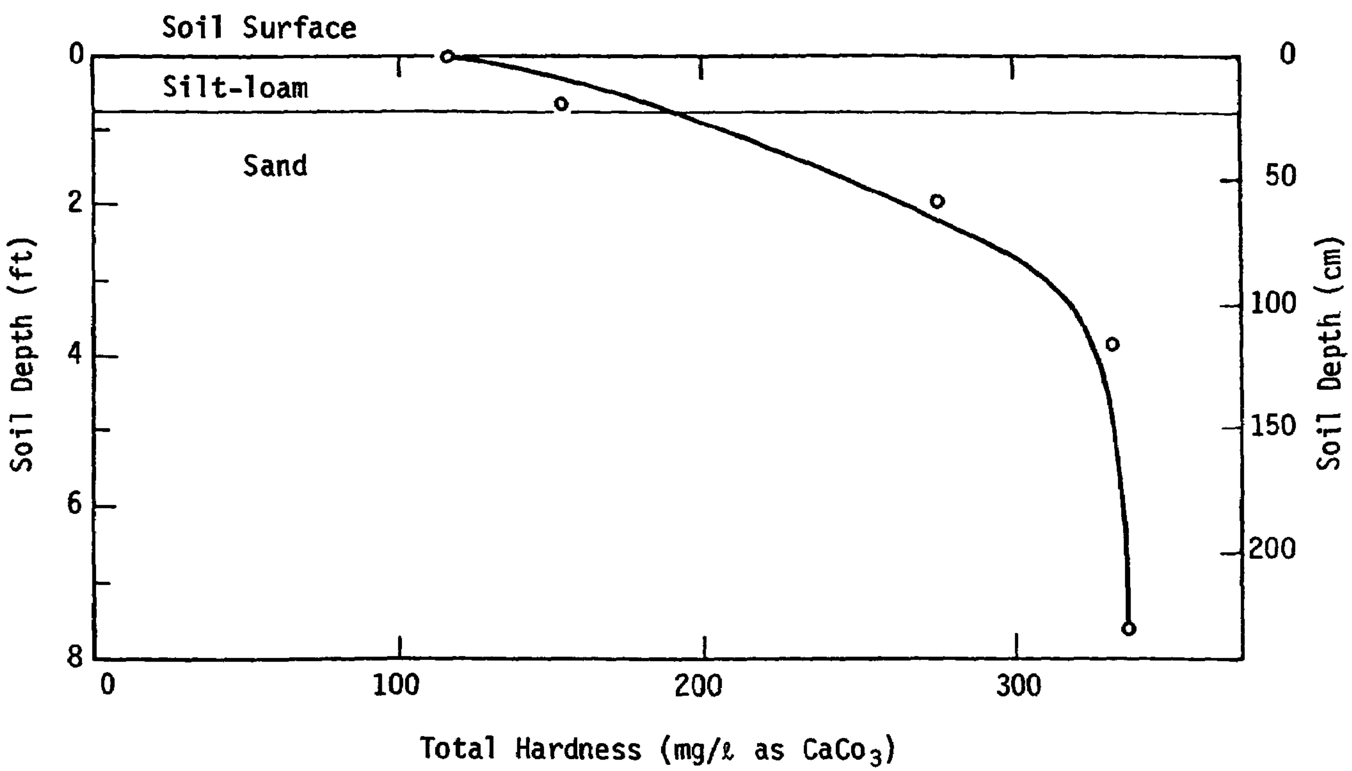


Figure 57. Wastewater hardness as a function of column depth.

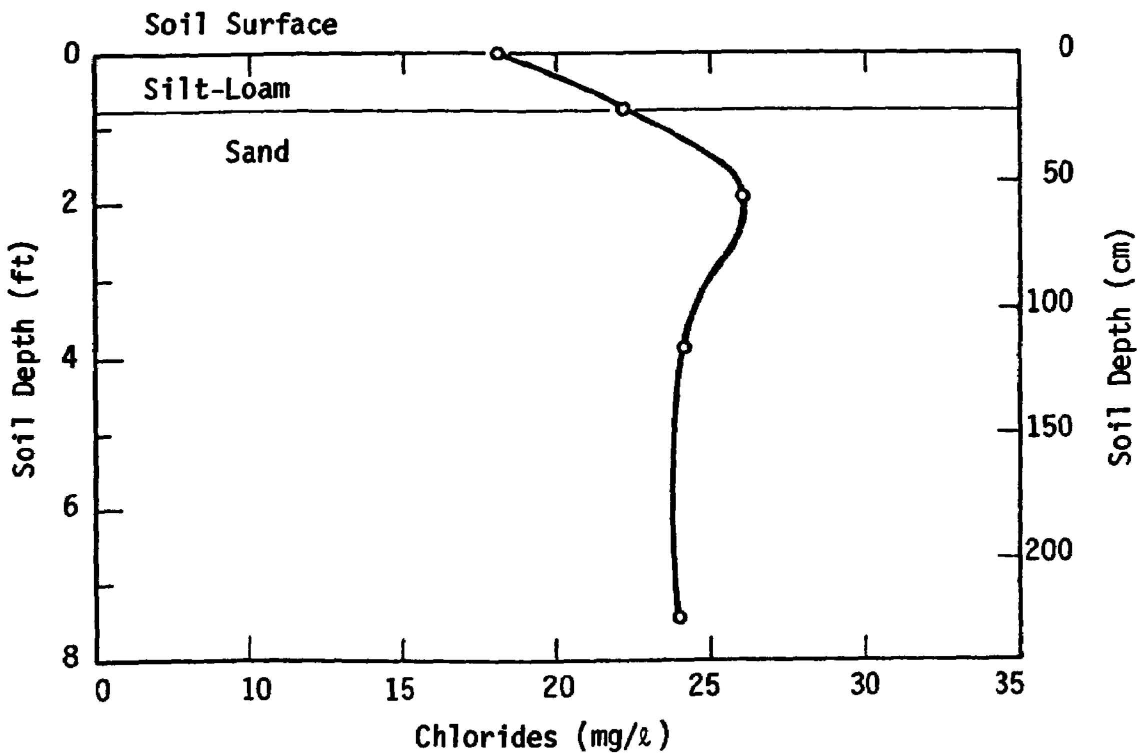


Figure 58. Wastewater chloride concentration as a function of column depth.

grain soil. However, the alkalinity and magnesium showed little change in concentration during percolation through the silt loam layer.

Testing for chlorides in the soil column effluents showed variations similar to those of the other dissolved constituents, although the changes in concentration were not as large. The chloride concentration increased rapidly in the top 0.61 meters (2 feet) of soil and remained essentially constant throughout the remaining column depth.

The observed variations in concentration of the dissolved materials within the soil matrix can be attributed to a number of reactions (30). The dissolved material in a soil solution may interact with the soil matrix by ion exchange, precipitation or dissolution with the solid phase, ingestion by microorganisms, incorporation into the soil organic matter, and reactions with the soil air. The large increases observed in the concentration of alkalinity, calcium, magnesium, and total hardness indicate that a dissolution process was likely occurring as a result of the contact between the soil and the applied wastewater. Chemical principles suggest that mineral dissolution would be expected when a soil solution contains dissolved salts at a concentration level which is below equilibrium with respect to any solid phase or mineral present. Thus, the composition of the renovated water ultimately would be controlled by the solubility of the various minerals within the soil matrix. However, for some precipitation and dissolution reactions, the rates are extremely slow and are controlled by kinetic and thermodynamic constraints (30). The materials tested in this study have been classified as being very reactive within the soil matrix (30). This would account for the very rapid change of concentration within the top centimeters of soil.

All of the materials tested did display the same characteristics in the lower regions of the column. In each case, the dissolved solids showed little change in concentration once the 1.22 meter (4 foot) column depth was reached. Thus, the reaction rate of the dissolution reaction had slowed and the soil solution was near equilibrium. This slowing of the reaction rate would suggest that the soils below the 1.22 meter (4 foot) level were not significantly involved in the dissolution reaction.

Coliforms

The concentrations of both fecal and total coliforms were determined during the study. A summary of these results has been presented in Table 10. As can be seen in this table, the column removed 95% of the total and fecal coliforms applied within the first 0.23 meters (9 inches) of the silt loam layer. As the wastewater moved further through the soil matrix, the coliform count was reduced somewhat more, but at a slower rate. These results were predicted since bacterial organisms are removed in soil by straining, die-off, sedimentation, entrapment, and adsorption (23, 24, 31). The finer grained silt loam layer would be expected to provide more locations for these processes to occur. The bottom layer was much coarser sand, and provided lower removals by the indicated processes. The results of this column study compare favorably with other data on the removal of bacteria in soils (32, 33).

TABLE 10. COLIFORM REMOVAL WITH DEPTH

	% Removal Total	% Removal Fecal
Port 1	95	95
Port 2	99.6	99.8
Port 3	--	--
Port 4	99.75	99.98
Port 5	99. 5	99.98

Heavy Metals

Figure 59 provides a summary of the behavior of heavy metals in the column as a function of column depth. It is clear from this figure that most of the heavy metal removal occurred in the top 25 cm (10 in) of the soil profile. It is not apparent from these data whether the decreased removal below 25 cm (10 in) was a mass action effect, which would be observed in a homogenous soil column, or whether the surface removal was simply due to the presence of the silty-loam soil in that region. However, it is likely that both mass action effects and soil type effects combined to yield the observed results, with soil type providing the predominant influence.

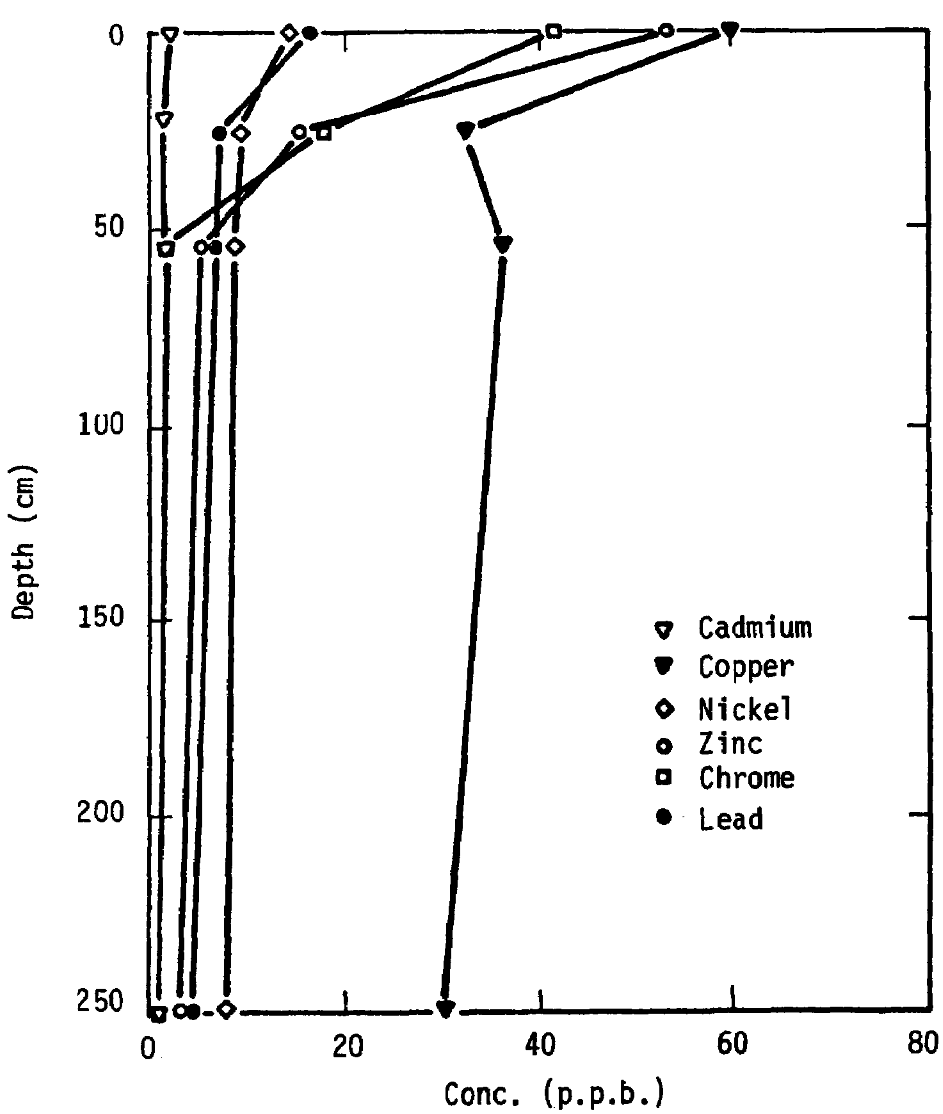


Figure 59. Wastewater heavy metal concentration variations with depth.

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16. ABSTRACT

Treatment performance of three rapid infiltration basins receiving secondary treated municipal wastewater is evaluated at the City of Boulder, Colorado, Sewage Treatment Plant. The prime objectives of the project are evaluation of the pre-treatment provided, the hydraulic load to the soil, duration of wetting and drying cycles, and climatic influences on system performance. Two of the basins have unscarified beds while one has a bed surface constructed in a ridge and furrow arrangement. Year-round operation allowed evaluation of applied effluent and infiltrated water collected from underdrains during extreme climatic conditions.

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