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WATER QUALITY SERIES
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ABSTRACT

The effectiveness of a sloped rock-grass filtration system in treating municipal wastewater was tested on a 24 m by 36 m (80 ft by 120 ft) slope on a 2.5 percent grade sown with a mixture of rye grass, fescue grass, and bluegrass. The field was divided into six plots, each approximately 3.5 m wide and 36 m long. Three of the plots (slope-rock) were constructed with 9 m of gravel, 7.6 cm deep, on the upper reaches of the slope. Raw (screened, degritted) municipal wastewater from Hyrum, Utah, was applied to the slope-rock sections at application rates of 13 and 20 cm/wk the first year of operation (June through October 1979) and 23, 41, and 51 cm/wk the second year (June through October 1980). The other three plots were constructed as conventional overland flow slopes. Wastewater was applied at rates of 13 and 20 cm/wk the first year and 23, 41, and 57 cm/wk the second year.

The gravel layer increased infiltration and, therefore, decreased the amount of wastewater effluent recovered. The gravel also increased wastewater detention times on the treatment slopes. In general, the slope-rock sections achieved higher mass removal associated with greater water losses. However, the gravel layer had no statistically significant effect, at the 95 percent confidence level, on the concentration of pollutants.

On a concentration basis, BOD_5 removal for the test sections were 87 to 93 percent. BOD_5 effluent averages ranged from 6 to 12 mg/l. Mean effluent suspended solids ranged from 5 to 9 mg/l. Even at the highest hydraulic loading rate (57 cm/wk), effluent quality met the 1985 State of Utah effluent limits.

Total phosphorus reductions were only 20 to 33 percent. Orthophosphate concentrations increased on all slopes.

Ammonia removals were a function of flow rate. Greater ammonia removals, 69 to 93 percent, were achieved at the lower loadings (13 cm/wk, 20 cm/wk and 23 cm/wk). The highest loading (57 cm/wk) exhibited 33 to 43 percent removal. Nitrification of ammonia occurred on all the slopes.

Fecal coliforms were reduced by as much as 99 percent on some of the slopes, but effluent fecal coliforms were not reduced below 10^4 colonies/100 ml.

Harvesting temporarily decreased system performance. Effluent BOD_5 and suspended solids concentrations, however, still did not violate effluent discharge limits (i.e., 15 mg/l BOD_5, and 10 mg/l SS, 30 day average) for the State of Utah.

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INTRODUCTION AND OBJECTIVES

Introduction

Land application of wastewater has gained increasing popularity since the advent of Public Law 92-500 (1972) and the Clean Water Act of 1977. The Environmental Protection Agency's (EPA) policy on land application is that "regional administrators should preferentially consider land application as an alternative wastewater management technology" (Sills et al. 1978).

Overland flow, as one land application method, has received attention as a treatment technology capable of meeting federal and state wastewater discharge standards (Table 1). The City of Melbourne, Australia, has been using overland flow to treat domestic sewage since 1897 (Croxford 1978). Campbell's food processing company was the first to use overland flow on a large scale in the United States (Gilde et al. 1971).

In overland flow treatment, wastewater is applied at the high end of a vegetated slope and collected at the bottom. The process was originated for land application on soils of low permeability. Wastewater pollutants are physically filtered and/or absorbed by vegetation and soil, and biologically removed by a microbial film as the wastewater flows down the slope (EPA 1977). Short-circuiting or channelization is the most common problem (Kemp et al. 1978, Peters 1978, Hall et al. 1979, Ketchum et al. undated). Clump forming grass, poor site preparation and grading, nonuniform wastewater application, or improper field maintenance can contribute to the problem.

Channelization reduces soil-wastewater contact. Extensive short-circuiting erodes slopes thereby increasing operation and maintenance costs. Both processes can be reduced through More uniform distribution of the wastewater flow would improve the cost effectiveness of the system. Hydraulic short-circuiting could be inhibited by addition of a gravel distribution zone at the top of the slope and seeding the rest of the treatment surface with sod-forming grasses (Figure 1). The gravel layer also provides additional surface area for microbial degradation

Table 1. Summary of waste discharge requirements (Horrocks 1977 and State of Utah 1978).

Date for Compliance	Requirement	30 Day Limitation
Jun 30, 1977	State Interim Discharge Requirement	BOD_5 = 25 mg/l, 85% removal SS = 25 mg/l, 85% removal Fecal coliform = 200/100 ml
Jul 1, 1977	EPA Secondary Treatment	BOD_5 = 30 mg/l, 85% removal SS = 30 mg/l, 85% removal Fecal coliform = 200/100 ml
Jul 1, 1983	EPA Best Practicable Treatment	Nitrification[a]
Jun 30, 1985	Utah State Discharge Requirement	BOD_5 = 15 mg/l, 30 day average 20 mg/l, 7 day average SS = 10 mg/l, 30 day average 12 mg/l, 7 day average Fecal coliform = 20/100 ml

[a]Possible exclusion for wastes with a temperature less than 20°C.

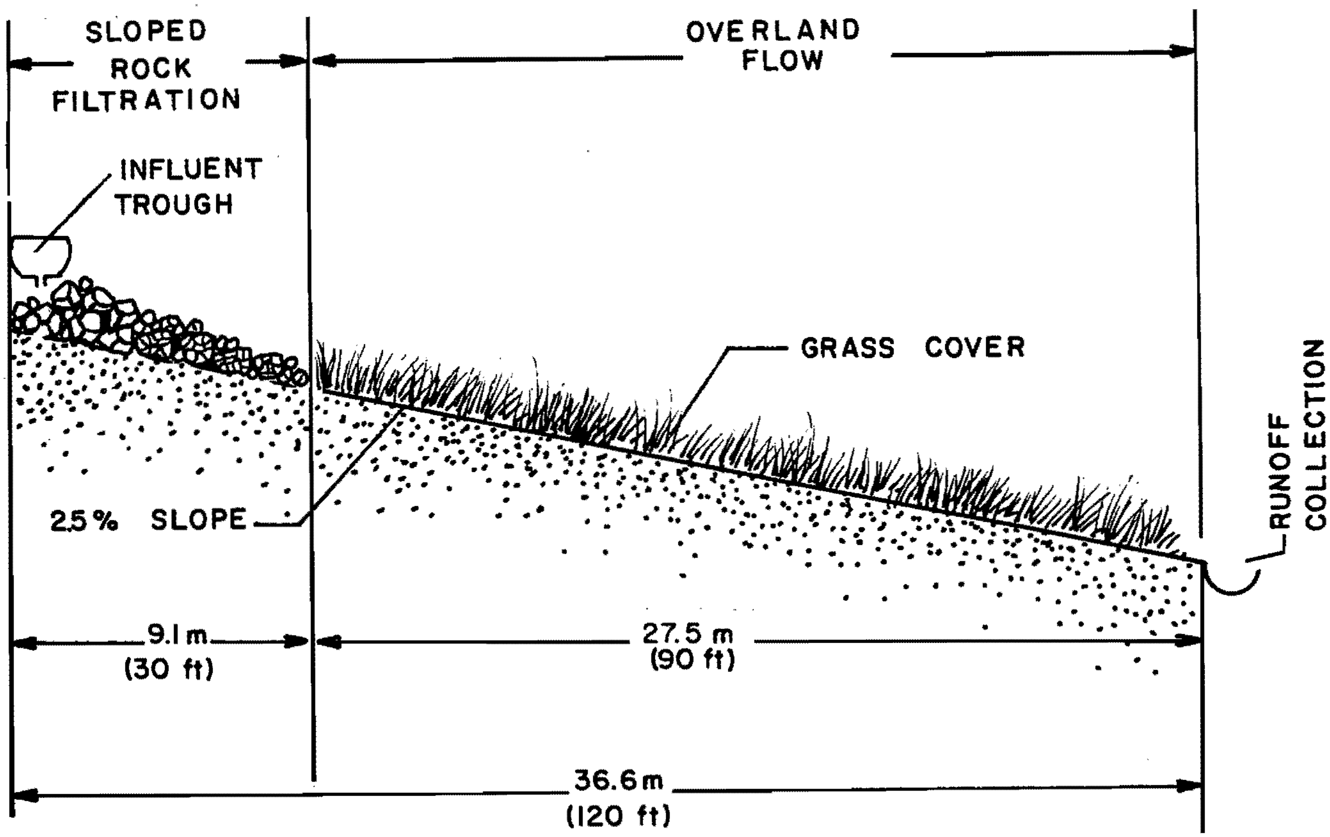


Figure 1. Schematic diagram of slope-rock filtration system.

of the wastewater. With improved performance, wastewater treatment flows could be increased, minimizing the land area required to meet discharge requirements of 15 mg/l for biochemical oxygen demand (BOD_5) and 10 mg/l suspended solids (SS).

Objectives

The overall objective of this study was to evaluate the effect of a gravel layer at the upper portion of the slope on the treatment performance of an overland flow system. Specific objectives were:

1. To evaluate the performance of slope-rock filtration in treating raw domestic wastewater at various hydraulic loading rates and to compare the results with the conventional overland flow system. Performance criteria were the removal efficiencies of inorganic and organic nitrogen, phosphorus, solids, and biochemical oxygen demand.

2. To compare the hydraulic performance of the slope-rock system to the hydraulic performance of the conventional overland flow system.

3. To compare the performance of the slope-rock and conventional overland flow systems at high loading rates (23, 41, and 57 cm/wk) to the results from lower loadings (13 and 20 cm/wk).

4. To evaluate the effects of grass harvesting on effluent quality.

LITERATURE REVIEW

Presently, there are two modes for nutrient (nitrogen and phosphorus) removal from wastewater: land application and advanced wastewater treatment (Table 2). Cost analyses show that, given suitable local conditions, land treatment systems can be more economical than conventional treatment (Pound et al. 1975). Land treatment system costs can be as much as 15 percent more than conventional systems and still be considered cost effective by the U.S. Environmental Protection Agency (Pound et al. 1978). Consequently, considerable interest has been generated on land treatment of municipal wastewater and developing effective technologies for the wide variety of local conditions encountered.

Land Application

Land application can be used to treat almost any type of waste which is amenable to biological treatment (Deemer 1978). Treatment efficiencies can be quite high for properly designed and operated systems. Design efficiencies for three basic land application methods shown in Figure 2 (slow-rate, rapid infiltration, and overland flow) are summarized in Table 3. Efficiencies vary with concentrations of constituents in the applied wastewater. The quality of the water after land treatment, however, appears to be somewhat more consistent. Effluent quality in terms of BOD_5, nitrogen, and phosphorus is nearly the same whether untreated, primary, or secondary effluent is applied at similar hydraulic loading rates (Pound et al. 1976). Expected values for renovated water quality by method of land application are compiled in Table 4. Typical system characteristics are shown in Table 5.

The advantages of land application include few pretreatment requirements, secondary and tertiary treatment combined with high removal efficiencies, no sludge handling costs, low capital costs (excluding land), and low operating costs, particularly from an energy use standpoint (Deemer 1978). Preventing wastes from entering surface waters, using the water and nutrients for crop production, renovating water for reuse,

Table 2. Design efficiencies and effluent qualities of conventional and advanced waste treatment processes (Loehr et al. 1979).

Treatment Process	Design Removal Efficiency (%)				Effluent Quality (mg/l)			
	BOD_5	SS	P	N	BOD_5	SS	P	N
Conventional and AW Treatments								
Preliminary Treatment	0	0	0	0	210	230	11	30
Primary Settling	20-40	50-65	-	-	140	110	-	-
Activated Sludge	75-95	-	-	-	20	25	-	-
Trickling Filter	75-90	-	-	-	30	35	-	-
Filtration	50	72	-	-	10	5	-	-
Activated Carbon	60	60	-	-	4	2	-	-
Two Stage Lime Treatment	-	-	50	-	-	-	0.5	-
Nitrification-Denitrification	-	-	-	90	-	-	-	3
Land Application Systems[a]								
Irrigation	98+	98+	80-99+	85+	4	5	2	6
Overland Flow	92+	92+	40-80	70-90	18	18	2-7	3-9
Infiltration/Percolation	85-99	98+	60-95	0-50	30	5	4	15-30

[a] Land Application Systems
Slow Rate
Overland Flow
Rapid Infiltration

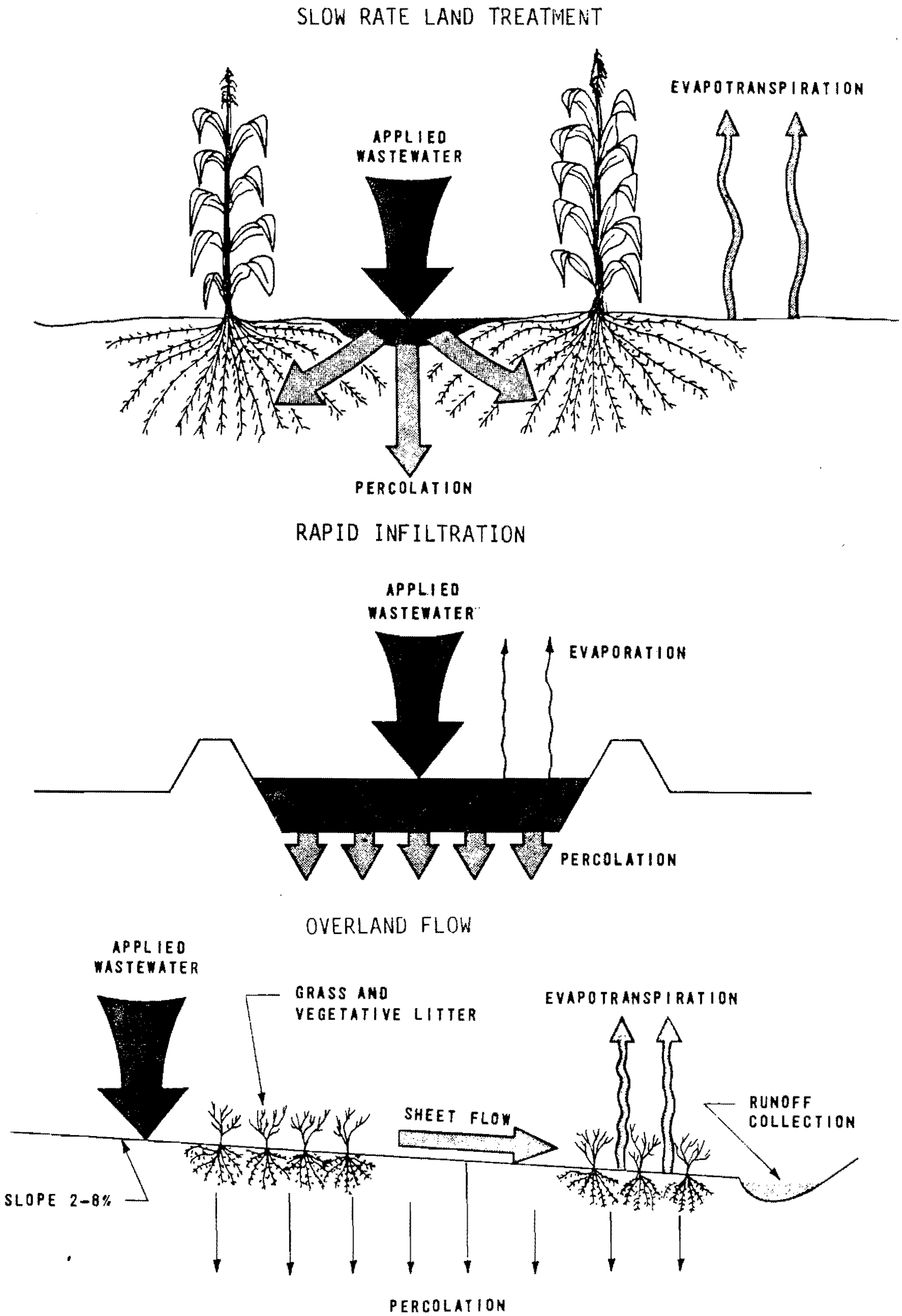


Figure 2. Schematic representation of the three major forms of land treatment (EPA 1977).

and minimizing environmental and health hazards are additional benefits (Bouwer et al. 1978).

The disadvantages of land treatment are land costs and climatic restrictions, particularly for cold weather operations (Deemer 1978). However, work at the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL) at Hanover, New Hampshire, has shown that land application is viable at temperatures of 0°C (Iskandar 1978, Jenkins et al. 1978). Public health considerations relate to the risk of transmission of pathogenic bacteria and viruses to higher biological forms either by aerosols or direct exposure, contamination of groundwater, crops consumption of irrigated water with wastewater effluents, chemical accumulation of heavy metals, and propagation of insects that could be vectors in disease transmission (Powell 1976).

Although land treatment systems differ greatly in some aspects, the underlying treatment mechanisms are similar, including biochemical immobilization, mineralization, nitrification,

Table 3. Treatment comparison of land application alternatives (EPA 1975).

	Type of Approach		
	Slow Rate	Overland Flow	Rapid Infiltration
Objective			
Recovery of renovated water[a]	0-70%	50-80%	Up to 97%
Treatment beyond secondary			
1. BOD_5 and suspended solids removal	98+%	92+%	85-99%
2. Nitrogen removal	85+%[b]	79-90%	0-50%
3. Phosphorus removal	80-99%	40-80%	60-95%
Grow crops for sale	excellent	fair	poor
Direct recycle to land	complete	partial	complete
Recharge groundwater	0-70%	0-10%	Up to 97%

[a]Percentage of applied water recovered depends upon recovery technique and climate.

[b]Depends upon crop uptake.

Table 4. Expected quality of treated water from land application processes, mg/l (EPA 1977).

Constituent	Slow Rate[a]		Rapid Infiltration[b]		Overland Flow[c]	
	Average	Maximum	Average	Maximum	Average	Maximum
Biochemical Oxygen Demand (BOD_5)	<2	<5	2	<5	10	<15
Suspended Solids (SS)	<1	<5	2	<5	10	<20
Ammonia Nitrogen (NH_3-N)	<0.5	<2	0.5	<2	0.8	<2
Total Nitrogen (TN)	3	<8	10	<20	3	<5
Total Phosphorus (TP)	<0.1	<0.3	1	<5	4	<6

[a]Percolation of primary or secondary effluent through 5 ft (1.5 m) of soil.

[b]Percolation of primary or secondary effluent through 15 ft (4.5 m) of soil.

[c]Runoff of comminuted municipal wastewater over about 150 ft (45 m) of slope.

and denitrification, chemical exchange, and physical filtering by the soil and plants, as well as plant uptake (Hoeppel et al. 1974).

Although industry has been the dominant user of overland flow wastewater treatment systems in the United States, the use of overland flow for treatment of domestic wastewater is developing.

Overland Flow

Unlike other land application processes, where wastes are removed by vertical movement of water through the soil, overland flow is specifically

Table 5. Assumed model land disposal conditions and characteristics for treating chlorinated industrial-domestic secondary effluent (Spyridakis and Welch 1976).

Item	Spray Irrigation	Overland Runoff	Rapid Infiltration
Soil type	medium texture soils, silt loam	clay, clay loam	loamy sand to sandy loam
Permeability of the most impermeable subsoil horizon to 60 in. (1.5 m)	moderately rapid to moderate; >0.6 in. (1.5 cm)/hr	very low; <0.2 in. (0.5 cm)/hr	rapid
Infiltration	moderately rapid to moderate	slow-very slow; <0.2 in. (0.5 cm)/hr	rapid to very rapid; ≥1 in. (5 cm)/hr
Soil drainage	moderately well drained	poorly drained	excessively drained
Soil depth	uniformly >5 feet (1.5 m)	>2 feet (0.6 m)	>10 feet (3 m)
Effective travel distance	preferred 10 ft (3 m) or more	surface travel of 150 feet (46 m)	subsurface >200 ft (60 m)
Slope	up to 8 ft (2.4 m)	5 ft (1.5 m)	-
Waterholding capacity to 60 in. (1.5 m)[a]	>6 in. (15 cm)	-	-
Depth to groundwater	>5 ft (1.5 m) to seasonal high groundwater table	>3 ft (0.9 m)	15-20 ft (4.5-6 m) or more
Vegetation	grass, year-round	permanent growing grass	vegetated
Climate	similar to that of Great Lakes	Great Lakes	Great Lakes
Application amount and rate	2 in. (5 cm)/wk once a week @ 0.2 in. (0.5 cm) to 0.25 in. (0.64 cm)/hr	2 in. (5 cm)/wk 4 times a week at 0.5 in. (1.3 cm)/day	60 in. (150 cm)/wk
Land required for 1-mgd (378,500 m^3/day) disposal	129 acres (52 ha)	129 acres (52 ha)	4.3 acres (1.7 ha)
Resting period between applications	3-6 days	3 days or more	14 days wet - 14 days dry
Duration	9 mo. Mar.-Nov.	9 mo. Mar.-Nov.	year-round
Application technique	spray	spray	surface

[a]Water-holding capacity in inches from the soil column is the depth of the layer of water that would be formed if all water in the soil that can be used by plants were concentrated at the soil surface.

adapted for use on relatively impermeable soils. Overland flow treatment combines physical, biological, and chemical processes. Settling and filtration by grasses reduce the suspended solids in amounts determined by the distribution method and type of grass cover. Degradation of organic matter and nitrogen removal is accomplished primarily by bacterial or other biological growth on the soil surface. In addition, vegetative growth provides a mechanism for nutrient uptake and removal. Performance is affected by temperature and flow variations. Phosphorus is adsorbed on soil until the adsorption capacity is reached. Adsorption capacity is determined by soil type (Hinrichs et al. 1980). Phosphorus removal is limited by minimum soil-water interaction. A primary advantage of overland flow is that it can be used to treat raw domestic wastewater (Thomas et al. 1976, EPA 1977). Unlike the other land application methods, however, overland flow has an effluent to be reused.

Overland flow treatment in the United States has been used primarily for treatment of high strength industrial wastewater such as that from canneries. The Campbell soup plant in Paris, Texas, has been operating an overland flow treatment system for over 10 years. The plant achieves high reductions in nitrogen (90 percent), phosphorus (58 percent), and biochemical oxygen demand (99 percent) (Gilde et al. 1971, Hinrichs et al. 1980). At present, there are only a few municipal wastewater overland flow treatment systems. Full-scale projects are at Easley, South Carolina; Pauls Valley, Oklahoma; and Carbondale, Illinois (EPA 1977, Crites 1978, Aly et al. 1979, Hinrichs et al. 1980). An overland flow treatment system has recently been designed for Newman, Calif. (Tucker and Vivado 1980).

In Europe and Australia, overland flow has been used extensively. Wastewater from the City of Melbourne, Australia, is effectively and economically treated by overland flow. The Werribee sewage farm established in 1897 is one of the world's largest land treatment installations (Croxford 1978). Seabrook (1975) reported that the Werribee system attained a 96 percent BOD_5 removal, 60 percent total nitrogen decrease, and 35 percent total phosphorus removal.

Research has been conducted by the EPA at Ada, Oklahoma, and the Army Corps of Engineers at Vicksburg, Mississippi, on the treatment of municipal wastewaters by overland flow. Overland flow research has also been conducted at Utica, Mississippi, the Cold Regions Research and Engineering Laboratories in New Hampshire, University of California at Davis, Southern Illinois University at Carbondale (Aly et al. 1979), Pennsylvania State University at University Park (Husted 1974), and Utah State University at Logan (Kemp et al. 1978).

The major tasks involved in operating an overland flow system include:

1. Maintaining the proper application rate and application frequency in a hydraulic loading cycle.

2. Managing the soil and cover crop.

3. Monitoring the performance of the system (Pound and Crites 1973).

<u>Hydraulics</u>

Hydraulic loading rates have been increased as more experience with the process has been gained. Ada, Oklahoma, began treating raw, comminuted wastewater in 1971 at rates of 7.4, 8.6, and 9.8 cm/wk. Hydraulic loading rates employed in recent projects are summarized in Table 6 (Crites 1978). Recommended loading rates are shown in Table 7.

<u>Application schedule</u>

The application schedule should be controlled so as not to overstress the system and thus bring about anaerobic conditions. Resting periods between applications are necessary for soil draining and aeration to provide oxygen for BOD_5 removal and nitrification of ammonia (Bouwer 1973, Pound and Crites 1973, Aly et al. 1979, Loehr et al. 1979). It is also undesirable to keep the soil continuously saturated with water. Adsorption of phosphate upon iron oxides depends on the presence of oxidized or ferric forms of iron. Complete saturation of the soil with water seriously limits the oxygen supply, after which reducing conditions may rapidly develop. The resulting production of ferrous iron will destroy the absorbing surfaces and even release phosphate previously adsorbed upon oxide

Table 6. Selected hydraulic loading rates for overland flow research projects (Crites 1978, Hinrichs et al. 1980).

	Type of Effluent Applied	Hydraulic Loading Rates, cm/wk	Degree of Slope, %	Slope Length, m
Ada, Oklahoma	Raw Comminuted	10–20	2–4	36
Ada, Oklahoma	Trickling Filter	25–40	2–4	36
Pauls Valley, Oklahoma	Oxidation Pond	25	2–3	45
Utica, Mississippi	Oxidation Pond	6–12	2–8	45
Hanover, New Hampshire	Primary and Secondary	5	5	21
Davis, California	Oxidation Pond	20	2	30

Table 7. Recommended hydraulic loading rates for overland flow (Deemer 1979).

Pretreatment Level	Loading Rate, cm/wk		
	Deemer (1979)	EPA (1977)	Aly et al. (1979)
Raw	6.3 to 15		
Primary	10.0 to 20	6.4 to 15	
Secondary	20.0 to 40	15 to 40	
For Nitrogen Removal			10 to 20

particles (Taylor and Kunishi 1974). Furthermore, under anaerobic conditions, ammonia nitrogen can also be released (Viets 1974).

The resting period should be long enough to allow the soil surface layer to reaerate, yet short enough to keep the microorganisms active. Drying can reduce the microbial population and thus treatment efficiency. Recovery, however, is usually quite rapid (Pound and Crites 1973).

The optimum cycle depends on climate and BOD_5 loading (Crites 1978, Kemp et al. 1978, McPherson 1978). Generally, application periods of 6 to 8 hours are alternated with resting periods of 16 to 18 hours. After 4 to 6 days, the resting period is extended to 1 to 3 days. At Melbourne, Australia, however, land application of wastewater has been continuous rather than intermittent.

Field dimensions

The length and slope of a field should fall within certain ranges for best performance (Aly et al. 1979). Plot lengths of 30 to 60 m with 2-8 percent slopes have been used successfully (Powell 1976, EPA 1977, Palazzo 1977, Aly et al. 1979). Length-to-width ratios of 1:4 to 1:10 have been used. High hydraulic loadings per unit width of plot can result in grass kill and channeling (Ketchum et al. undated).

Soils

Overland flow has been specifically adapted to soils of low permeability (<0.5 cm/hr) (Spyridakis and Welch 1976, EPA 1977) where the subsoil, usually clay loam, acts as a barrier to down ward migration of applied wastewater. Wastewater permeates only the top few centimeters (Wright and Rovey 1979).

The soil biomass is versatile and effective in decomposing natural and synthetic organic compounds (Bouwer and Chaney 1974). The biological community (bacteria, actinomycetes, fungi, algae and soil micro- and macro-animals) recycles wastes by decomposing organic compounds; inhibiting propagation of pathogenic microorganisms; contributing nutrients to the nitrogen, phosphorus, and sulfur cycles; and influencing the solubility and mobility of inorganic ions (Miller 1973).

When overland flow is used to treat raw wastewater, the organic content of the soil is enhanced. Organic matter improves soil structure, water holding capacity, cation exchange capacity, adsorption of heavy metals, and denitrification.

Cover crop

A cover crop, usually grass, protects the soil from erosion and provides surface area for biological activity and a mechanism for nutrient uptake and removal (Pound et al. 1976). Vigorous, water tolerant grasses which form dense sods are ideal for overland flow systems (Carlson et al. 1976, Marten et al. 1978, Wolcott and Cook 1978). Effective cover crops include bluegrass, trefoil, and Italian rye grass. Clump forming grasses, which do not completely cover the top soil, such as reed canary or tall fescue are less desirable, unless mixed with a sod forming grass, because of the increased potential for erosion (Pound and Crites 1973, Kemp et al. 1978, Greene et al. 1980, Hinrichs et al. 1980). Nonetheless, clump forming grasses have been used.

Overland Flow Performance

A summary of data for selected overland flow systems is found in Table 8. Overland flow performance results from systems reviewed by Hinrichs et al. (1980) are presented in Table 9. Overland flow systems are most effective in reducing biological oxygen demand, suspended solids, and nitrogen.

Table 8. Literature data summary for overland flow systems (Overcash 1978).

Input to OLF System	Length of OLF System m	Application Rate cm/wk	Effluent															Total Coliform	Fecal Coliform
			H_2O	BOD_5	COD	TSS	TN	NH_3-N	O-N	NO_3-N	TP	TS	TVS	Na	Al	Ca	Cl		
			Percent reduction, with effluent concentration from OLF system, mg/l, in parenthesis																
Raw or minimally treated domestic waste	365	13(cool period)	-	96 (25)	-	95 (25)	45	-	-	-	35	-	-	-	-	-	-	-	-
	36	7.4-9.8 (warm per.)	50	93 (11)	77 (73)	95 (8)	89 (2.6)	94 (1)	86 (0.8)	- (0.4)	60 (4)	20 (826)	53 (140)	-	-	-	-	-	-
	36	7.4-9.8 (cool per.)	50	92 (12)	83 (53)	92 (12)	77 (5.4)	97 (0.5)	67 (1.9)	- (2.8)	56 (4.4)	31 (720)	42 (170)	-	-	-	-	-	-
	36	9.8 (alum addition)	50	96 (7)	86 (41)	93 (16)	88 (2.6)	94 (0.8)	86 (1.2)	- (0.6)	84 (1.6)	27 (810)	61 (120)	-	60 (0.28)	-	-	97.3 (0.2×10^6)	97.5 (0.025×10^6)
	36	9.8 (no alum addition)	50	94 (9)	84 (54)	93 (16)	86 (2.9)	95 (0.8)	82 (1.6)	- (0.5)	62 (3.7)	31 (770)	62 (120)	-	80 (0.15)	-	-	95.9 (0.3×10^6)	91 (0.09×10^6)
Secondary treated domestic waste	62	330 (warm period)	-	62 (12)	-	75 (8)	-	0 (16)	-	- (0.2)	-	-	-	-	-	-	-	-	-
	39	400	-	59 (9.7)	-	73 (15)	-	20 (8.4)	-	10 (21.9)	-	-	-	-	-	-	-	-	-
	46	6-25 (based on area of first terrace)	45-55	(0.5-7)	-	-	15 (15)	0 (12)	-	10 (5-6)	15 (5)	-	-	0 (21)	-	0 (38)	0	-	-
	6	5	-	0 (15)	-	-	-	100 (0)	45 (2.2)	99 (0.1)	40 (8)	-	-	-	-	-	-	-	-

Table 9. Existing overland flow system descriptions and data--summer/winter (Hinrichs et al. 1980).

Location	Type of Facility	Type of Wastewater	Preapplication Treatment	Runoff % of Applied	Slope %	Slope Length m	Wetted Area ha	Ave. Flow Rate m^3/d	Hydraulic Loading Rate cm/wk	Organic Loading Rate kg BOD/ha-d	Nitrogen Loading Rate kg N/ha-d	Ave. Percent Removal BOD	SS	N	P
Davis, CA	Research Pilot Studies	Domestic Sewage	Oxidation Pond	87	2	30	0.05	15	20	16	[d]	70	69	[d]	[d]
Hunt Wesson[a] Davis, CA	Full Scale	Food Processing	Screening	21	2.5	30	97	12,000	9	166	8.1	97	99	84	-
Ada, OK[b]	Research	Domestic Sewage	Screening	47	2	36	2.4	510	10-20	61/68	13/12	98	98	90	50
			Primary	50	2	36	0.8	260	15-20	14/9	5/6	98	98	90	50
			Oxidation Pond	50	2	36	0.8	260	25-40	7.4/4.3	4.6/4.1	98	98	90	50
Utica, MS[a]	Research	Domestic Sewage	Oxidation Pond	-	2-8	46	0.50	46	6.5	2.2	2.0	55	57	90	50
								130	18	6.2	5.6	[d]	[d]	75	30
Carbondale, IL	Full Scale	Domestic Sewage	Oxidation Pond	83	7-12	60	0.06	38	44	26	13	76	[d]	64	64
Hanover, NH	Research	Domestic Sewage	Primary	25	5	30	0.03	2.1	5	6.0	2.6	91/58	97/84	94/25	89/30
			Secondary	80						3.7	2.3	95/80	96/88	87/32	80/30
Easley, SC	Full Scale	Domestic Sewage	Screening	70	6	55	0.53	91	12	32	3.7[c]	84	76	77	45
			Oxidation Pond	70		47	1.4	290	15	15	0.7[c]	59	48	74	52
Paris, TX	Full Scale	Food Processing	Screening, Grease Removal	87	2-8	60-75	285	17.2	4.2	42	0.44	99	93	90	58

[a]Nitrogen added to promote grass or for research purposes.

[b]For spray application.

[c]NH_4 only.

[d]Not reported.

Biochemical oxygen demand and suspended solids

The predominant removal mechanisms for BOD_5 and suspended solids are physical filtration, sedimentation, and bio-oxidation (Thomas et al. 1974, Powell 1976). Secondary treatment discharge requirements (Table 1) for both BOD_5 and suspended solids effluent concentrations have been consistently met by the overland flow systems. Most of the overland flow treatment systems presented in the literature were able to meet the 1985 Utah State Polished Secondary Discharge Requirement of 15 mg/l BOD_5 and 10 mg/l SS over a 30 day average (Tables 8 and 9).

Contradictions, however, are found in the literature on BOD_5 removal efficiency. Thomas (1978) reported that loading rate and influent wastewater quality had no apparent effect on effluent BOD_5 concentration. Hinrichs et al. (1980), however, presented data from overland flow systems (Figure 3) which indicated that the percent reduction of BOD_5 was a function of hydraulic loading rate and degree of pretreatment. Thomas (1978) reported the results from one overland flow

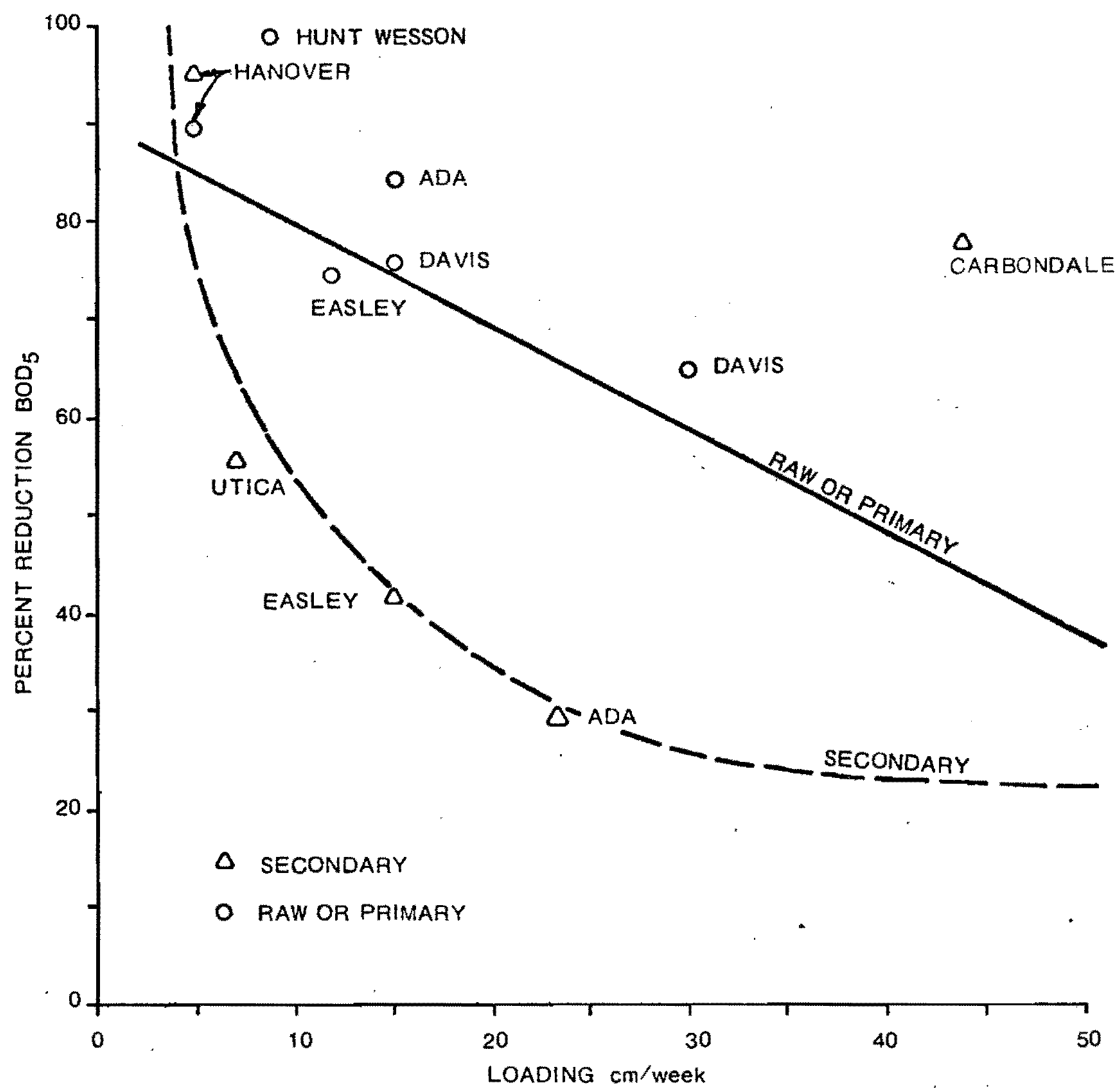


Figure 3. Hydraulic loading effects on BOD_5 reductions for overland flow systems (Hinrichs et al. 1980).

system operating under the same local, climatic, and design conditions, whereas, Hinrichs et al. (1980) were combining results from many systems operating under many different local, climatic, and design conditions. EPA (1977) concludes that overall BOD_5 removal efficiency is dependent on soil type, grass cover, temperature, and application schedule, as well as hydraulic loading rate.

Low soil temperatures have been found to affect BOD_5 removal. Jenkins et al. (1978) and Martel et al. (1980b) reported that during winter operation a municipal wastewater overland flow treatment system exceeded the runoff BOD_5 concentrations of 30 mg/l required by the secondary discharge standard for municipal wastewater treatment. However, Gilde et al. (1971) reported that organic removals were not seriously affected by operation at soil surface temperatures near freezing when treating cannery wastes. Biological studies at the cannery site in winter revealed that, although the respiration of microorganisms slowed down, their mass activity remained constant due to a tenfold increase in microbial population (Gilde et al. 1971). In general, information on climatic effects is very limited (Hinrichs et al. 1980).

System age has also been shown to affect BOD_5 removal efficiencies. The site at Ada, Oklahoma, required approximately 2 to 4 months to stabilize (Thomas et al. 1974). Nitrogen in the plot runoff declined to 5 mg/l in 60 days, while suspended solids declined to less than 25 mg/l after 120 days of operation. Other systems have taken less time. Kemp et al. (1978) collected data from an overland flow system which did not change over the time of operation.

Phosphorus

Total phosphorus reductions for overland flow systems range from 40 to 80 percent. Mechanisms for phosphorus removal include plant uptake and reaction with soil constituents through sorption and precipitation. Relatively large quantities of inorganic phosphorus can be sorbed in soils containing $CaCO_3$ (lime) and/or oxides of Fe and Al (Sommers et al. 1977). Since most of the wastewater flows over the soil surface and not through the soil matrix, overland flow has the poorest phosphorus removal capabilities among the land application alternatives (Ellis 1978, Lee and Peters 1978, Loehr et al. 1979). Carlson et al. (1974) reported that the phosphorus concentration in the runoff from the Vicksburg, Mississippi, overland flow system was 40 to 60 percent of that applied. The subsurface flow contained only a trace of phosphorus (0.18 mg/l).

Phosphorus removal can be increased to about 90 percent by adding 1.5 to 2.0 mg/l aluminum for each mg of phosphorus (Thomas et al. 1976, Lee and Peters 1978). Another alternative for increasing phosphorus removal is the use of soil infiltration following overland flow (Thomas et al. 1976).

Plant uptake. Grasses typically remove 10 percent of the phosphorus applied to the slopes, but 50 to 60 percent removal has been reported through plant harvesting (Carlson et al. 1974, EPA 1977, Palazzo et al. 1980). Thomas et al. (1976) noticed a change in concentration of phosphorus in the runoff from summer to winter amounting to the mass of phosphorus removed by plant harvesting.

Soil interactions. Phosphate removal by soils results from a combination of adsorption of phosphate and precipitation of compounds of phosphorus with $CaCO_3$ (lime) and/or oxides of Fe and Al (Shewman and Peterson 1973, Beek et al. 1977, Sommers et al. 1977). Data cited in the literature indicate that the phosphate retained in the soils was associated with hydrous forms of iron and aluminum in acidic soils (Taylor and Kunishi 1974, EPA 1977, Sommers et al. 1977, Hook and Burton 1979). Above pH 8, phosphate is removed by precipitation as calcium phosphates because of the high Ca content of most wastewaters (Taylor and Kunishi 1974, Holford and Patrick 1978). The soil properties most likely correlated with adsorption of phosphorus would be surface area, pH, cation exchange capacity (CEC), and percent clay (Shewman and Peterson 1973), as well as the presence of iron, aluminum, or lime.

Soils, however, have a finite capacity for removing phosphates from wastewaters (Novak and Adriano 1975, Beek et al. 1977, Sommers et al. 1977). Only a small proportion of the total aluminum and iron is present in reactive form (hydrous oxides) at neutral pH. Flooding soils usually shift the pH of the soil solution into the neutral range so that flooding decreases phosphorus removal (Beek et al. 1977, Holford and Patrick 1978). Waterlogged soils can cause a change in redox potential and pH. Upon reduction, iron (III)-phosphate-hydroxo complexes present in the soil are converted to more soluble iron (II) and phosphate. The pH controls the dissolution and subsequent reprecipitation of the reduced compounds. Under aerobic conditions, iron will be oxidized and adsorb phosphate (Holford and Patrick 1978, Lijklema 1980).

Sawney and Hill (1975) reported that soils treated with phosphorus solution regained the capacity to remove phosphorus after drying and wetting cycles. Consequently, resting periods for soil draining and aeration are desirable for phosphorus removal. The mechanisms of regeneration of a soil's phosphorus removal capabilities are not well defined. It appears that drying and wetting of the soil brings Al, Fe, or Ca, and fresh mineral surfaces in equilibrium with the soil solution, thereby creating new sites for phosphorus sorption. Factors responsible for the differences in the regeneration capacity of different soils have not been determined (Sawney and Hill 1975).

Nitrogen

Nitrogen removal mechanisms in overland flow treatment systems include: nitrification-denitrification, crop uptake, volatilization of NH_3, incorporation into microbial tissue, or removal by the soil mechanisms. Soil mechanisms for removal of nitrogen include: adsorption of NH_4^+ on the soil cation exchange sites, fixation by clay minerals, or adsorption by organic matter (Carlson et al. 1974, Thomas et al. 1974, Lance 1975, EPA 1977, Gilbert et al. 1979). Nitrogen uptake by plants harvested and removed from the slopes and biological denitrification are the most important processes for the removal of nitrogen by overland flow systems (Carlson et al. 1974, Thomas et al. 1976, Brar et al. 1978, Loehr et al. 1979).

Nitrate-nitrogen is readily taken up by most crops as a nitrogen source (Hoeppel et al. 1974) and is highly mobile and generally moves with the percolation of soil water (Lance 1975). Ammonium-nitrogen, stored primarily as a result of adsorption by the soil, can also be incorporated into microbial cell tissue or fixed in a stable form by expanding clay minerals if trapped between layers of minerals such as montmorillonite. Some nitrogen immobilized by microbial cells are stable, but some can be released by microbial decay (Viets 1974, Lance 1975). Nitrogen can also be released as ammonia from herbage (Scott and Fulton 1978).

Nitrification-denitrification. Nitrification-denitrification reduces applied nitrogen because of the close proximity of an oxidizing zone in the flowing water to the reducing zone below the soil surface.

In overland flow treatment, a thin film of wastewater passing over the surface of relatively impermeable soils serves as a barrier to oxygen movement below the soil surface. Organic nitrogen is applied to the soil surface and biologically converted to inorganic nitrogen. Nitrifying bacteria aerobically oxidize inorganic ammonium to nitrite and ultimately to nitrate. Denitrifying bacteria, existing in an anaerobic zone below the soil surface, convert nitrates into nitrogen gas. Energy is supplied by soluble organic matter (Carlson et al. 1974, Hoeppel et al. 1974, EPA 1977, Khalid et al. 1978, Aly et al. 1979).

Some denitrification can occur in a "reducing" microenvironment of an otherwise well-drained aerobic soil. For example, NH_3-N can be oxidized to NO_3-N, which then diffuses into a microzone near plant roots or near pieces of decomposing plant or animal residue. The extent of denitrification by this mechanism is limited to less than 30 percent of the total nitrate, whereas all of the nitrate may be denitrified in a waterlogged soil (Mitchell 1974, Lance 1975).

The effectiveness of this biological nitrogen control process depends on the ability of nitrifying organisms to oxidize ammonia to nitrate. The interaction of factors such as pH, temperature, moisture, dissolved oxygen (DO), microbial numbers, acclimation, and inhibiting compounds influences the rate at which nitrification takes place (Anthonisen et al. 1976). A schematic diagram showing the nitrification-denitrification process is shown in Figure 4 (Khalid et al. 1978).

Plant uptake of nitrogen. Plant uptake accounts for 10 to 30 percent of the nitrogen removed by overland flow systems (Lance 1975, Carlson et al. 1976, Khalid et al. 1978). Nitrate-nitrogen is assimilated by most crops as a nitrogen source (Hoeppel et al. 1974).

Ammonia volatilization. Nitrogen removal by ammonia volatilization is only significant under highly alkaline conditions. Volatilization losses are small for overland flow systems treating raw or primary treated domestic wastewater because the pH of wastewaters is usually between 7 and 8 (Lance 1975). At approximately neutral pH and 20°C, only 3 to 10 percent of the ammonia nitrogen will exist as volatilizable (free) ammonia (Mills et al. 1974, Lance 1975, Anthonisen et al. 1976, George and Adams 1980). Kemp et al. (1978), treating lagoon effluent on an overland flow slope, found 10 percent of the influent ammonia was stripped during

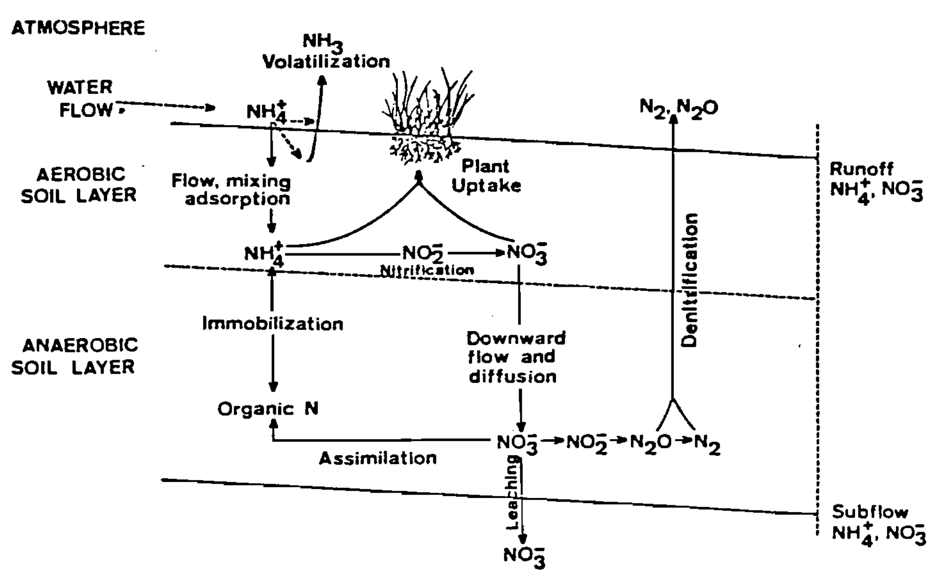


Figure 4. A schematic diagram of various N removal processes occurring in overland flow treatment of wastewater (Khalid et al. 1978).

sprinkler application. There is more potential for ammonia volatilization using spray application than by flooding.

Immobilization and mineralization. Approximately 5-10 units of nitrogen are incorporated into cell tissue for each 100 units of carbon incorporated. Nitrogen, immobilized by incorporation into microbial tissue, can be released (mineralized) by decay of microbial cells. Some immobilized nitrogen incorporated into humus is stable (Mitchell 1974, Viets 1974, Lance 1975).

Soil removal mechanisms. Khalid et al. (1978) reported that the soil accounts for 35 percent of the nitrogen removed by overland flow treatment. Nitrogen may be immobilized in the soil by adsorption of NH_4^+ on the soil cation exchange sites, fixation by clay minerals, and adsorption by organic matter (Carlson et al. 1974, Thomas et al. 1974, Lance 1975, EPA 1977, Gilbert et al. 1979).

Ammonium-nitrogen (NH_4^+) removal by soil adsorption is temporary. NH_4^+ is oxidized under aerobic conditions to nitrate (nitrification). Thus soil adsorption of NH_4^+ is an important temporary storage mechanism for final removal by nitrification-denitrification.

On the other hand, ammonia-nitrogen can react with soil organic matter to form complexes that resist leaching and decomposition. Fixation of NH_4^+ by clay minerals is also a permanent removal mechanism. Incorporation into microbes can be either temporary (released by decay) or permanent NH_4^+-N removal mechanisms (Lance 1975). Nitrate-nitrogen forms soluble salts so there is essentially no reaction between it and the soil (Thomas 1972).

Lance (1975) summarized the effects of soil moisture content with regards to denitrification. At moisture contents well below field capacity, nitrogen loss is insignificant. Bremmer and Shaw (1958) observed little nitrogen loss below 60 percent of the soil water-holding capacity even if other conditions for denitrification were favorable.

The second general environment is a well-aerated soil with reduced microzones where denitrification occurs. These zones increase in number as moisture content increases and are more numerous in fine-textured soils than in sandy soils. The amount of nitrogen lost is difficult to estimate. The only data available are from experiments where fertilizers were applied to soils. Broadbent and Clark (1965) estimated these losses at 10 to 15 percent of the total amount of fertilizer applied. More nitrogen may be lost when sewage is applied.

When the moisture content is increased above field capacity, a third environment is established where denitrification is quite rapid. Meek and Mackenzie (1969) found that production of N_2 in incubated soil samples did not change when the soil water varied from 34 percent to 41 percent, but increased sharply when the moisture content reached 44.5 percent. The soil contained 48 percent moisture at saturation and 32 percent at field capacity. Stefanson (1973) reported that changes of 2 to 6 percent in water content above field capacity sometimes doubled the amount of denitrification. Increases in soil moisture content above field capacity result in saturation of large pores and a predominantly reduced environment. Under such conditions, the denitrification rate would usually be governed by the amount of organic carbon available if the nitrogen were in the nitrate form.

Overland flow systems effectively remove nitrogen in combinations of these three environments but the third is the most effective. Typical nitrogen removals are about 60 to 90 percent (Tables 8 and 9). Melbourne, Australia, however, achieved only a 30 percent removal of nitrogen (Scott and Fulton 1978). The system at Hanover, New Hampshire, obtained a nitrogen removal of 94 percent (Table 9).

Hoeppel et al. (1974), Thomas (1978), and Hinrichs et al. (1980) have shown that increasing the flow rate decreases ammonia-nitrogen removal. Hinrichs et al. (1980) summarized the nitrogen removal efficiencies of the various overland flow systems in the United States in a graph (Figure 5) that supports the hypotheses that nitrogen removal decreases with increased hydraulic loadings. Therefore, retention time is important for nitrogen removal. Nitrogen removal is kinetically limited at higher loadings (EPA 1977).

As with other biological systems, cold temperatures reduce nitrogen removal efficiencies. During winter operation, nitrogen removal dropped from 87 and 94 percent to 25 and 32 percent at Hanover, New Hampshire (Table 9). Optimum temperatures for nitrification are 24° to 35°C. Optimum temperatures are much higher for denitrification (60° to 65°C). Minimum temperatures for nitrification and denitrification are 5° and 2°C, respectively (Mitchell 1974, EPA 1977).

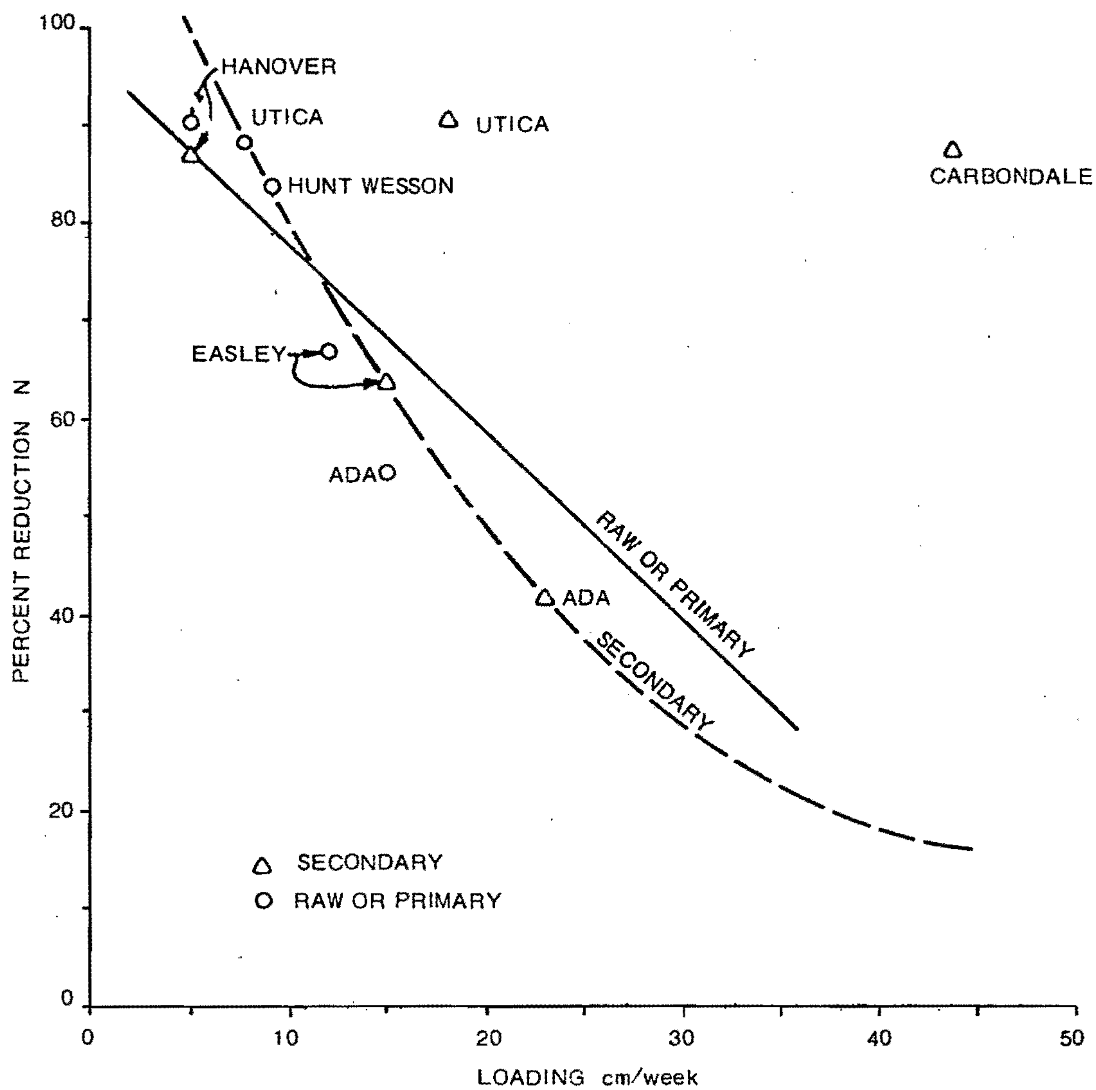


Figure 5. Hydraulic loading effects on nitrogen reductions for overland flow systems (Hinrichs et al. 1980).

Environmental Hazards of Overland Flow

Public health concerns related to land application and specifically overland flow are 1) pathogenic bacteria and viruses in municipal wastewater and their possible transmissions to higher biological forms including man, 2) heavy metals and chemicals that can accumulate and jeopardize both crops and animals, and 3) the propagation of insects that could be factors in disease transmission (Pound and Crites 1973, Powell 1976, Wasbottom 1978). Since overland flow or spray runoff systems are used only on grasslands, no crops used for human consumption are contaminated. Since very little water enters the slowly permeable soils, contamination of groundwater by these systems is generally not a problem. Experience at Werribee farm, Australia, indicates that forage may be safely used for cattle (Lance 1978).

Pathogenic bacteria and viruses

In overland flow systems, bacteria and viruses are removed primarily by entrapment and settling of suspended solids harboring the microorganisms, and adsorption by clay constituents. Detention times in overland flow systems normally are too short for substantial inactivation (Bouwer and Chaney 1974, Loehr et al. 1979). The pathogens are eventually destroyed by the natural environmental conditions which favor native soil organisms such as predation, toxicity of the soil, photochemical reactions, desiccation, and thermal inactivation (Schaub et al. 1978, Loehr et al. 1979). Laboratory and field studies have shown that viruses and bacterial pathogens may persist as long as several months on vegetable crops and in soils. In addition, some parasitic eggs and cysts have been shown to last in soils up to 3 years (Larkin 1978). Wastewater pathogens do not generally survive as long on vegetation as they do in soil because they are more exposed to adverse environmental conditions (EPA 1977, Loehr et al. 1979). Bacteria can be expected to survive on crops about 30 to 40 days while virus survival ranges from 14 to 36 days (EPA 1977).

Thomas et al. (1976) reported a reduction in fecal coliform numbers after treating raw comminuted sewage by overland flow. However, the final number of fecal coliforms was greater than 10^4/100 ml. An overland flow system treating raw (screened) domestic wastewater at St. Pauls Valley, Oklahoma, reduced influent coliform numbers by 80 to 90 percent. The overland flow system at Utica, Mississippi, reduced coliform numbers during October through March. Significantly more coliforms than those in the applied wastewater were found in runoff waters during the summer months (Peters and Lee 1978).

Schaub et al. (1978) using labeled bacteriophage, f_2, found that 30 to 60 percent of the viruses were removed by an overland flow system. The reduction of enteric virus was even greater due to poor adsorption characteristics of f_2 virus. In timed studies, it was determined that tracer virus advanced to the bottom of the slopes at the same rate as wastewater. Soil sampling revealed that some f_2 virus was associated with the wastewater saturated topsoil (Schaub et al. 1978).

Another pathway for infection is via aerosols (EPA 1977, Hall et al. 1979) emitted by spray applicators. At St. Pauls, Oklahoma, spray boom application resulted in detectable bacterial aerosols as far as 60 m (Hall et al. 1979).

No disease transmission has been documented from any properly operated land treatment systems in the U.S. (EPA 1977, Loehr et al. 1979). However, to date, no epidemiological studies have been conducted. Grazing land has been irrigated with untreated wastewater on a large scale in Europe and Australia. There seems to be little health threat to farm animals under normal conditions. It also appears that cattle which are fed effluent irrigated silage show no ill effects, and the milk or meat from them does not appear to be infected (Loehr et al. 1979).

Heavy metals and chemicals

Potentially toxic elements to plants or animals are copper (Cu) and cadmium (Cd) (primary food chain hazards), zinc (Zn), nickel (Ni), lead (Pb), boron (B), molybdenum (Mb), and cobalt (Co) (Chaney 1974, EPA 1977, Sidle et al. 1977, Epstein and Chaney 1978, Loehr et al. 1979). Concentrations of these metals in domestic wastewaters not combined with industrial discharges are normally low. Other pollutants of major interest include organic chemicals (chlorinated hydrocarbons and other pesticides) (Bouwer and Chaney 1974, Jones and Lee 1977) and high nitrate concentrations (Loehr et al. 1979). At this time, there is essentially no information on the fate of many of the organic compounds found in domestic wastewaters when disposed of by land treatment (Jones and Lee 1977).

Factors that govern the availability of heavy metals to plants and the uptake and accumulation of heavy metals in plants are:

Soil factors

1. Soil pH: Toxic metals are more available to plants below pH 6.5.

2. Organic matter: Organic matter can chelate and complex heavy metals so that they are less available to plants.

3. Soil phosphorus: Phosphorus interacts with certain metal cations to decrease their availability to plants.

4. Cation exchange capacity (CEC): This factor is important in the binding of metal cations. Soils with a high CEC are safer for disposal of sludges.

5. Moisture, temperature, and aeration: These can affect plant growth and uptake of metals.

Plant factors

1. Plant species and varieties: Vegetable crops are more sensitive than grasses to heavy metals.

2. Organs of the plant: Grain and fruit accumulate lower amounts of heavy metals than leafy tissues.

3. Plant age and seasonal effects: The older leaves of plants will contain higher amounts of metals.

Other factors

1. Reversion: With time, metals may revert to unavailable forms in soil.

2. Metals: Zn, Cu, Ni, and other metals differ in their relative toxicities to plants and in their reactivity in soils (Sidle et al. 1977, Croxford 1978, Epstein and Chaney 1978).

The high removal of heavy metals (from 72 to over 90 percent) from wastewater by overland flow was found to occur in the organic mat, with little movement deeper into the soil profile. The greatest accumulation of heavy metals occurred nearest the point of wastewater application. The grass nearest the point of wastewater application also contained the greatest concentrations of heavy metals (Hoeppel et al. 1974, Carlson et al. 1976).

A study of Werribee Sewage Farm has revealed that substantial amounts of metals have accumulated during 70 years of application of raw and settled sewage (Bouwer and Chaney 1974). Investigations into the liver and kidney levels of metals in the farm cattle grazed on the wastewater irrigated pastures indicate no increase with age. The levels, while generally higher than those of farm cattle grazed solely on nonirrigated pastures, were marginally lower than a random sample from nonsewage farm cattle (Croxford 1978).

Dowdy et al. (1978) reported no accumulation of trace metals in reed canary grass, quackgrass, tall fescue, or orchard grass as a result of effluent applications. Copper values for grass tissues were well below average due to enhanced phosphorus levels (Bouwer and Chaney 1974, Dowdy et al. 1978).

Insects and rodents. Gilde et al. (1971) and Kemp et al. (1978) reported infestations of mosquitoes and other pests (i.e., flies, snails, and worms), although they were not believed to be hatched on site. Experience at Paris, Texas, indicated that application of insecticides to control snails and army worms had no perceptible effect on the microbial population in the soil or on the performance of the system. Control of mosquitoes is needed to protect both the animal and human population from the nuisance and disease threat associated with large numbers of mosquitoes (Loehr et al. 1979).

Control of hazards. The disease threat to animals from eating wastewater irrigated crops and fodder can be minimized by preventing grazing on pastures for at least 2 weeks after wastewater application is ceased and by drying and storing forage crops before feeding them to animals (EPA 1977, Loehr et al. 1979).

To reduce aerosol infection, buffer zones and vegetative screening should be incorporated into the design. In addition, application techniques which minimize aerosolization (i.e., low pressures) should be used, such as large droplet irrigation equipment or surface flooding. If spray irrigation is employed, it should be stopped during high winds (Wasbottom 1978). Loehr et al. (1979) suggest that raw wastewater should not be used with spray application.

Soils at the overland flow site should be monitored for potentially toxic elements (i.e., organics, metals) before operation begins to determine naturally occurring amounts (Loehr et al. 1979). Wastes containing high levels of toxic elements should be pretreated or excluded from the overland flow process to extend the lifetime of the site (Chaney 1974, EPA 1977).

Preventative measures, such as controlling standing water conditions (Loehr et al. 1979) by providing adequate drainage and drying (resting) periods, should be used to hamper insect propagation. Research concerning mosquito control on overland flow systems at the University of Notre Dame indicated that 24 hours of dry conditions each week would effectively control mosquito populations on the plot (Ketchum et al. undated). Furthermore, a saturation period in excess of 2 or 3 days would control fly populations.

Harvesting

Vegetation is critical to efficient overland flow treatment. Vegetation provides soil erosion protection, filtration of wastewater, an environment for beneficial bacteria growth, a mechanism for nutrient assimilation, and potential revenue to help defray operating costs (Hinrichs et al. 1980). It is critical to maintain the vegetation in a healthy, productive, and renovative state. This involves regular cutting and harvesting of grass crops (State of Maryland 1978) to a) remove the nutrients and minerals taken up by the plants (Pound and Crites 1973, Clapp et al. 1978, Hall et al. 1979, Hook and Burton 1979), b) renew the plant's capacity to accomplish this uptake, c) extend the life of the renovation site, and d) realize any cash value from the crop.

Vegetation is an effective nutrient sink when the crop is harvested and removed. For example, reed canary grass can take up more than 160 kg of nitrogen per acre annually. Phosphorus is not removed as readily as nitrogen. An acre of reed canary grass removes only 15 kg of phosphorus per year, and an acre of bluegrass removes 5 kg of phosphorus annually. Quantities of nitrogen and phosphorus removed vary with vegetation type by a factor of about 10 (Loehr et al. 1979). Figure 6 illustrates removal rates for various types of vegetation. Different land application systems result in different levels of nutrient uptake.

Nitrogen and phosphorus are retained a) in a standing crop, b) in the detritus, and c) in residual humus. Decay of natural plant residues releases stored nutrients back into the environment. Nitrogen and phosphorus retained by the vegetation may become sources of soluble nitrate and phosphate in the future (Wolcott and Cook 1978). In overland flow systems, both crop uptake and denitrification play major roles in nitrogen removal. Since wastewaters only have contact with the soil surface, phosphorus removal may depend heavily on vegetative uptake and subsequent harvesting (Loehr et al. 1979). Phosphorus retained in the overland flow system gradually reverts to a soluble, stable form (orthophosphate) which can then be leached.

Frequency and time of harvest are important factors in maintaining a plant's capacity to remove nutrients and sustain lush growth (Smith et al. 1973, Jung et al. 1974, Hook and Burton 1979). Hook and Burton (1979) reported that grass plots that were never cut had a rapid growth early in the growing season, but after mid-July, productivity slowed, biomass declined, and plants decomposed. Frequent mowing, however, maintained a good stand of bluegrass. Tall fescue and Kentucky bluegrass failed to persist well when cut only twice annually. When cut four times annually, excellent persistence was reported (Smith et al. 1973, Jung et al. 1974, Marten et al. 1978).

Timing of harvest is important in maximizing the sale value of a crop. A grass crop should be harvested when it has the highest nutritive value, in order to maximize crop value and nutrient removal capacity (Pound and Crites 1973). Experimental work at Paris, Texas, indicated that it is possible to predict the time of year and stage of growth when hay has the highest value (Gilde et al. 1971). Consultation with local farm advisors or agricultural extension service representatives can be helpful in planning a harvesting schedule (Pound and Crites 1973, Loehr et al. 1979).

Prices and crop yields vary geographically and should be determined from local sources (Loehr et al. 1979). The Werribee farm in Australia recovers about 12 percent of its costs from the sale of livestock raised and fattened on irrigated pasture land (Hart 1974). Returns from the sale of hay at Paris, Texas, contributed 8 percent of the operation and maintenance costs. The forages grown in an overland flow system are not usually readily marketable; however, nutrient contents are increased to a point where they are comparable to higher quality varieties (Gilde et al. 1971). Hanover, New Hampshire, forage was tested for feed analysis during the last cutting in 1979. The forage was considered excellent quality for grass hay and contained an average of 20 percent crude protein and 69 percent total digestible nutrients. Grass testing 15 percent or above in crude protein and 65 percent or above for total digestible nutrients is considered excellent (Palazzo et al. 1980). The use of forages irrigated with municipal wastewater for dairy and beef cattle is not, however, widely practiced in the northeastern and northcentral United States. Therefore sale of harvested grass may not be possible (Hook and Burton 1979).

Cutting the crop is also beneficial because it eliminates the possibility of tall grass interfering with system management. Excessively tall grass interferes with wastewater distribution and does not allow the operator to spot trouble signs such as broken pipes. Equally important, high grass can

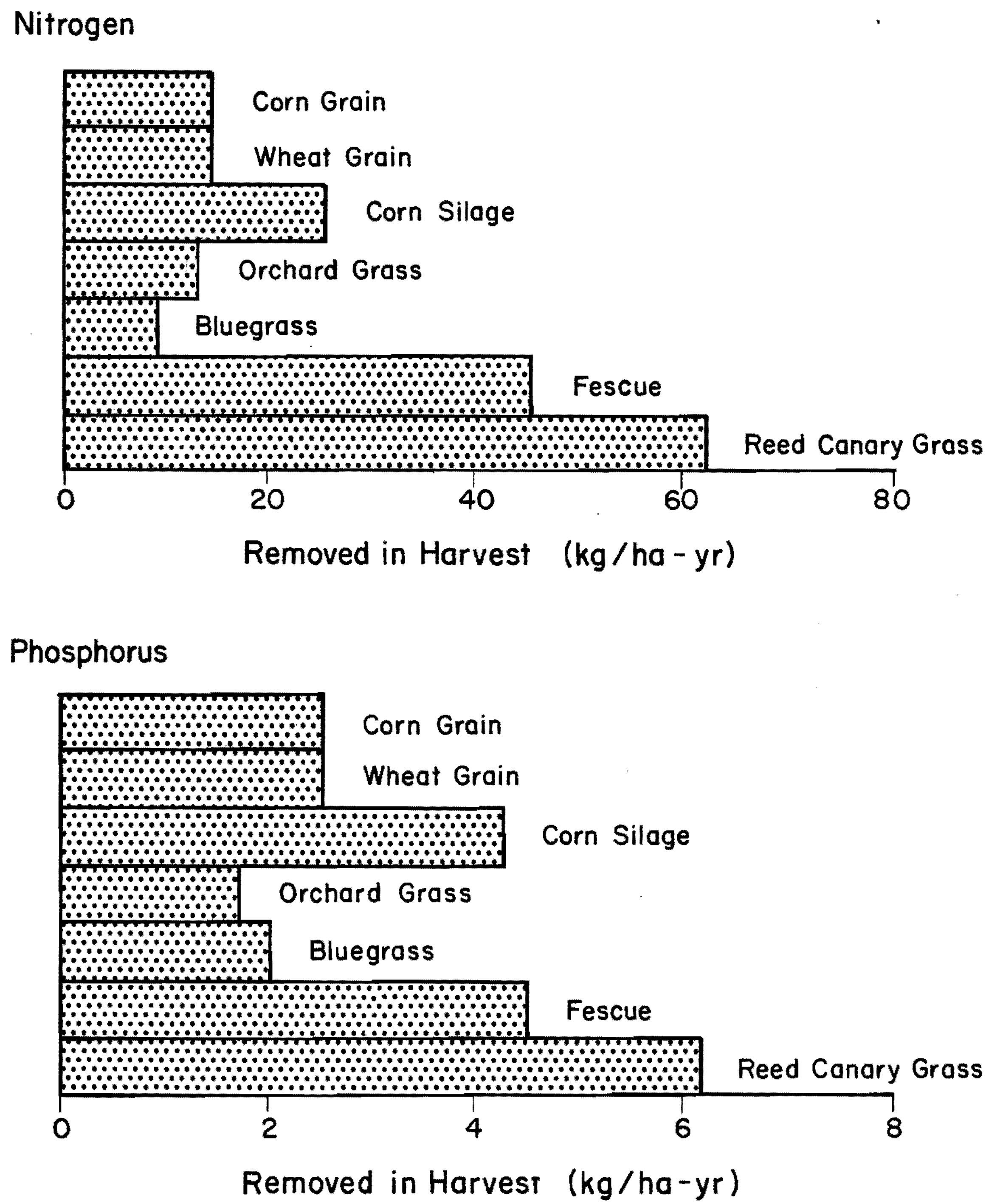


Figure 6. The relative amounts of nitrogen and phosphorus removal in crop harvest vary with the type of crop and the proportion of the tissue removed (Loehr et al. 1979).

inhibit the terrace surface from drying during the off cycle (Pound and Crites 1973, Aly et al. 1979). Moreover, tall grass is subject to wind damage and may shade and kill shorter grass below, making cutting and removal operations difficult (Hall et al. 1979, Hinrichs et al. 1980, Ketchum et al. undated). If grasses are killed, they may be replaced by weeds which will remove less nitrogen and phosphorus or reduce the marketable value of the crop, or erosion and channeling could result. The eradication of weedy plants or erosion problems necessitates system shutdown time and costly reseeding (Hook and Burton 1979, Palazzo et al. 1980, Ketchum et al. undated). Grass should not be allowed to grow higher than about 1.0 m (3 ft), but should be cut when 20 to 30 cm (8 to 12 in) high to approximately 8 to 10 cm (3 to 4 in) (Aly et al. 1979, Hall et al. 1979).

Three croppings per growing season were common practice at overland flow sites in Vicksburg, Mississippi (Lee and Peters 1978), Paris, Texas (Center for the Study of Federalism 1972), and Pennsylvania State, Pennsylvania (Hall et al. 1979). The Ada, Oklahoma, overland flow fields were harvested based on the height of stand (30 to 40 cm). Infrequent cutting hindered efficient operation of mowing and baling equipment (Hook and Kardos 1977). At the Werribee farm in Australia, crop removal is carried out by grazing animals at the end of their winter application season (McPherson 1978).

During harvesting, alternate disposal areas must be available to accept wastewater while the field dries. Water applications must be discontinued in advance of harvesting so soils can drain and carry harvest equipment without serious impairment to soil structure (approximately 5 to 10 days) (Hall et al. 1979). Use of equipment with high flotation tires is helpful in preventing damage. Also, by operating the equipment perpendicular to the slope of the plot, any damage that is caused would not result in channeling of wastewater (Pound and Crites 1973, Sopper 1973, Wolcott and Cook 1978, Aly et al. 1979, Hall et al. 1979, Martel et al. 1980a).

To minimize disruption of wastewater application schedules, harvesting must be carried out expeditiously. Adequate power, labor, and equipment must be provided, allowing for inevitable delays due to weather (Wolcott and Cook 1978). Harvesting can be performed either on a contract basis or by purchasing the equipment. At Paris, Texas, grass is harvested by contract with local farmers. In the past, grass was cut and removed from the field while it was still green. It was then chopped and pelletized for cattle feed. Future plans are for the grass to be cut, windrowed, and allowed to dry on site. Once dry, the grass will be baled (Hinrichs et al. 1980). The type of equipment used at Ada, Oklahoma, that produced the best results was a sickle-type cutter and baling machine (Hall et al. 1979). Utica researchers, however, recommended occasional mulching in some areas to help maintain a dense vegetative mat (Hinrichs et al. 1980).

While regular harvesting facilitates nutrient removal, it may also cause degradation of effluent quality. Hoeppel et al. (1974) found that nitrate removal efficiency appeared to decrease when the grass was cut to a height of a few centimeters, and blue-green algae were allowed to form a crust over the surface of the soil. Removal of ammonium and organic nitrogen also decreased. Law et al. (1970), however, observed that high nitrate in overland flow effluent occurred when system operation was resumed following any shutdown. In addition, drying the slopes reduced the microbial population, and thus resulted in reduced treatment efficiency (Pound and Crites 1973).

Alternate disposal areas must be available to accept wastewater during the period allowed for field conditions to dry. Either storage or additional wetted area will be required. Assuming three harvests per year and a dry period of one week per harvest, the wetted area must be increased by 11 percent (3 wk/28 wk growing season) (Martel et al. 1980a).

Site Description

The test site was adjacent to the Hyrum, Utah, Sewage Treatment Plant (Figure 7). Prior to land application, the raw domestic sewage was passed through bar-racks, grit removal, and coarse screens. It was then pumped through approximately 61 m (200 ft) of PVC piping to the top of six sloping plots having the dimensions shown in Figure 8. Half of the six plots were constructed as conventional grassed overland flow plots, and the other three had 9 m (30 ft) long by 4.5 cm (3 in) deep gravel layers at the upper (influent) ends. The first 1.8 m (6 ft) of this gravel layer was composed of 4 cm (1.5 in) gravel, and the remaining 7.3 m (24 ft) was covered with 1 cm (3/8 in) gravel.

The foundation soil was of high clay content and graded at a 2.5 percent slope. A 15 cm (6 in) topsoil layer covered the clay. The cover crop consisted of a mixture of rye grass, bluegrass, and fescue grass established the previous year (1979). The plots were separated by dirt berms.

Modified rain gutters (Figure 9) distributed the flow across the top of each plot thereby minimizing aerosols. Sheet metal barriers at the bottom of each slope collected the effluent into V-shaped sampling troughs. The effluent was then channeled to a sump and subsequently pumped to the headworks of the Hyrum plant (Figure 7).

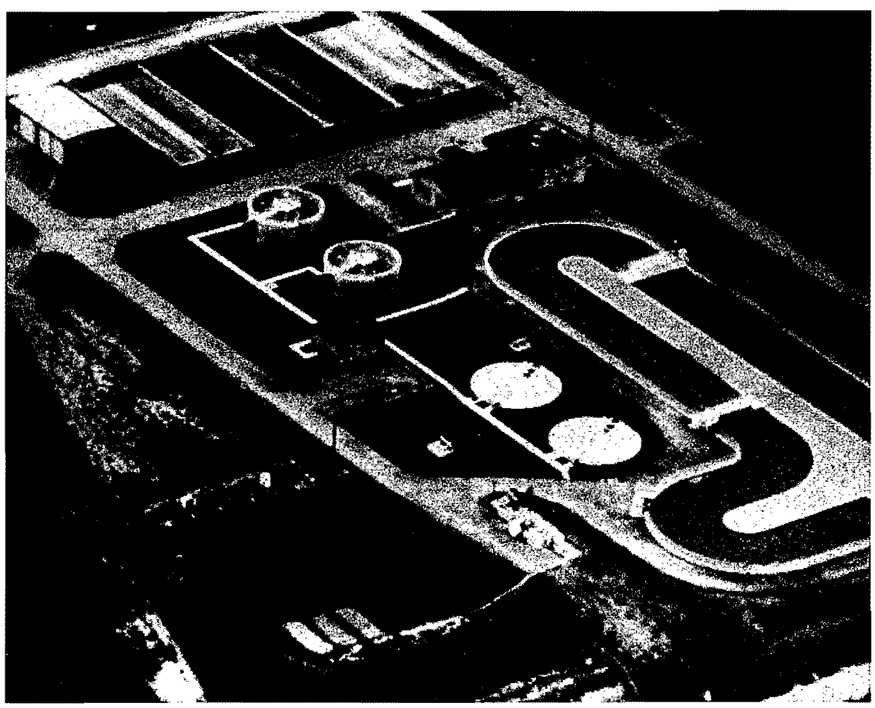


Figure 7. Slope rock and overland flow test plots located adjacent (lower left) to Hyrum City Sewage Treatment Plant.

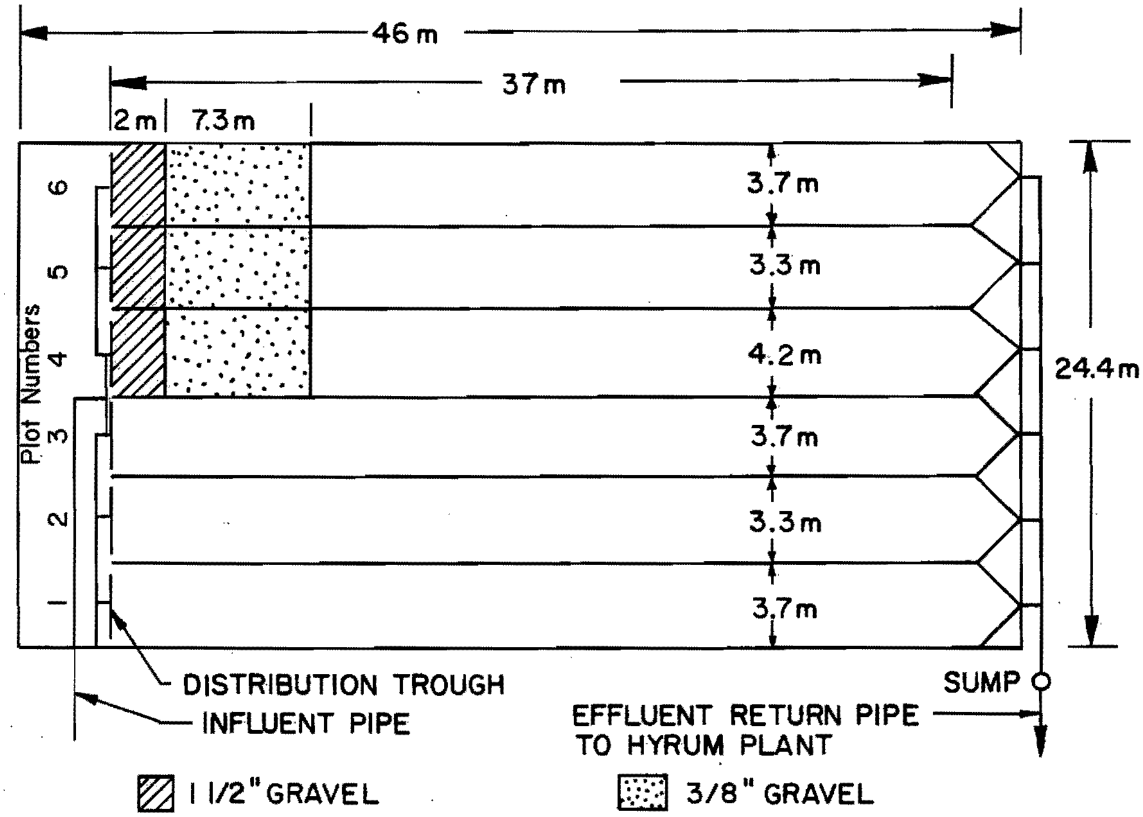


Figure 8. Schematic plant view of experimental facility.

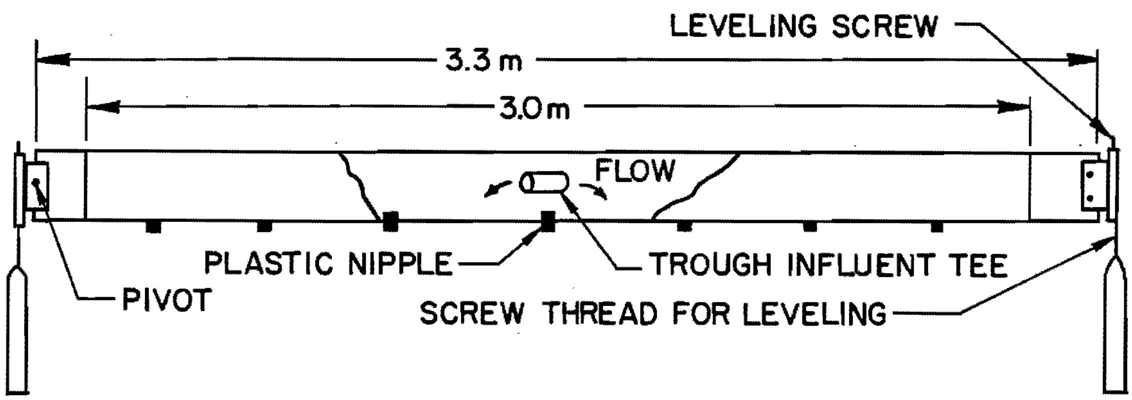


Figure 9. Cutaway view of rain-gutter distribution trough.

Operation and Data Collection

Wastewater application began in June of 1979 and 1980, and continued until October of 1979 and 1980. Wastewater was applied Monday through Friday, from 8 a.m. until 4 p.m. Drying and soil reaeration occurred on weekends and 16 hours per weekday. Hydraulic loading rates designed for 13, 20, 41, and 58 cm/wk (5, 8, 16, and 23 in/wk) were applied to both the conventional and modified overland flow slopes by pumping wastewater directly from the headworks with a Kenco Model 58N submersible pump. Variations in the geometry of each plot altered the actual wastewater application rates. The 20 cm/wk application was actually 23 cm/wk for both the slope-rock and overland flow plots for the second year (1980) of operation. Similarly, the slope-rock section loaded at the highest hydraulic loading (58 cm/wk) was only 51 cm/wk.

Water sampling and analyses

Influent and effluent samples were collected each Thursday during the application period with Isco composite samplers (Model 1580). The influent sampler started at 8 a.m. and sampled every 20 minutes for 8 hours adjacent to the Kenco influent pump. The effluent samplers were manually activated when the runoff reached the V-shaped discharge troughs from which the samples were obtained. The effluent samplers extracted samples every 20 minutes until 4 p.m., at which time the samples were taken to the Utah Water Research Laboratory for analyses. Samples were transported and stored at 4°C to retard degradation.

Intermediate samples were taken 9 m (30 ft) below the influent troughs of each test section as overland flow was intercepted by glass jars placed in PVC pipe sleeves (Figure 10). Three sampling devices were installed at 0.3 m (1 ft), 1.8 m (6 ft), and 3.3 m (11 ft) across each plot. Samples were manually composited from these samplers at 2-hour intervals starting at 10 a.m. and refrigerated. Samples from the 9 m sampling point on each section were collected and composited the fourth day of each application period from September 6, 1979, to October 4, 1979. During the second year of the study, the intermediate samples were not collected until after August 25, 1980.

Influent and effluent samples were analyzed for the parameters listed in Table 10. The intermediate samples were analyzed for the same parameters except that volatile suspended solids and total phosphorus were not measured because extraneous solids (mostly soil) were introduced during sampling and interfered with the results. All the analyses except total phosphorus, nitrite and nitrate nitrogen, and total Kjeldahl nitrogen were analyzed within 4 hours after the completion of sampling. Total phosphorus samples were autoclaved and stored at 4°C and analyzed within 24 hours. Nitrite and nitrate nitrogen were preserved with chloroform and refrigerated at 4°C and analyzed within one week, except for the samples taken during the 15th week of the second year of operation. They were analyzed 2 to 3 weeks later due to autoanalyzer malfunctions. Standards were run after one, two, and three weeks of preservation. No substantial change was believed to have occurred at high nitrogen concentrations. Total Kjeldahl nitrogen samples were acidified and analyzed within one week except during the first year of operation when some samples were stored longer. Laboratory analyses were performed according to procedures in the 14th edition of Standard Methods (APHA 1975).

Hydraulics

A fluorescein dye study was used to determine the surface hydraulic characteristics of each plot. Concentrated dye was introduced at the influent troughs as a single pulse. Water samples were collected at the effluent flume, and analyzed with a Turner Model 430 spectrofluorometer for fluorescein. The spectrofluorometer was set for an

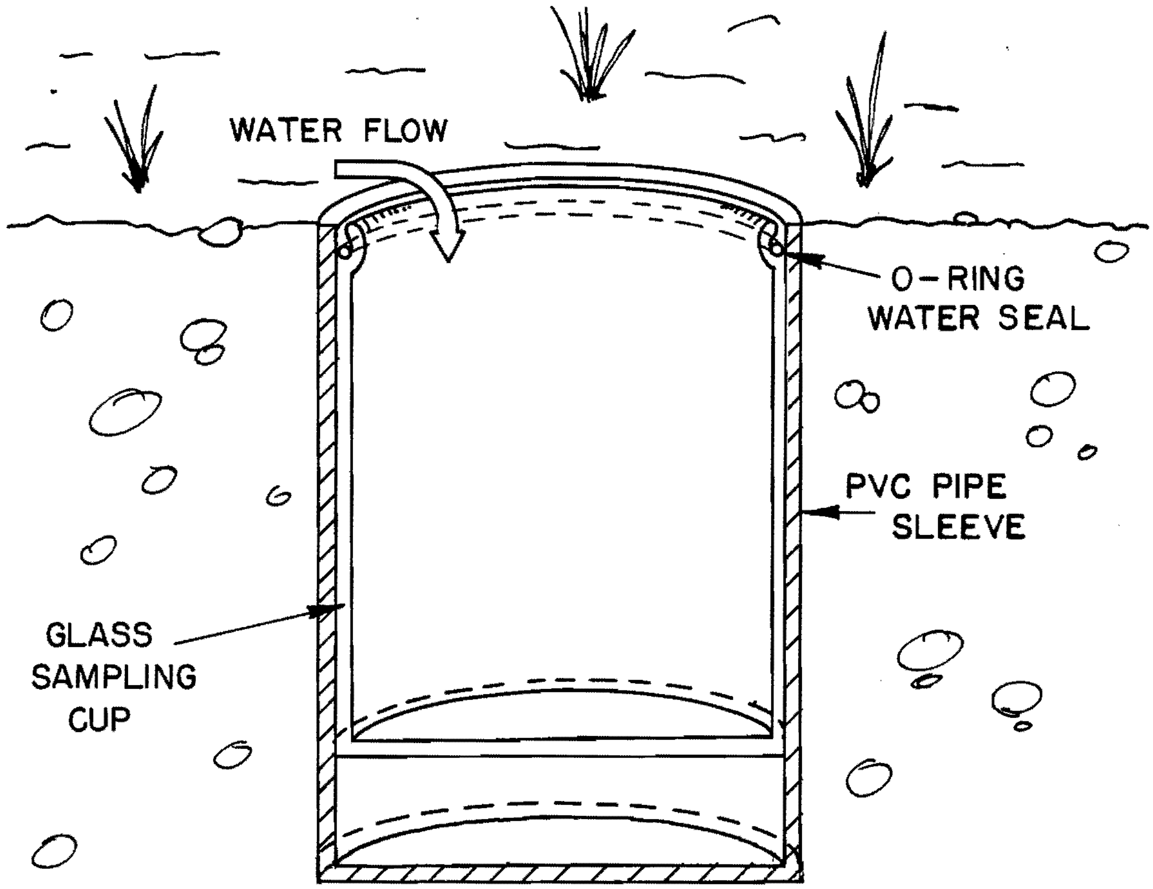


Figure 10. Sampling cut cross-section at 9 m from influent troughs.

Table 10. Procedures for analyses performed (APHA 1975).

Parameter	Test Method
Volatile Suspended Solids	Gravimetric
Suspended Solids	Gravimetric
Total Phosphorus	Persulfate digestion, ascorbic acid method
Orthophosphate Phosphorus	Ascorbic acid method
Ammonia	Phenate method
BOD	Membrane electrode method
Nitrate	Automated
Nitrite	Automated
Total Kjeldahl Nitrogen	Kjeldahl method

emission wavelength of 530 nm. A polarizing filter and a Turner 2A filter were also used.

Influent hydraulic flows were calibrated, using a shallow, plastic, 25-liter container. The flows were adjusted to reach the calibration marks in 45 seconds. The flows remained relatively constant when the influent pump intake screen was free of debris. The intake screen was manually cleaned every 2 hours to ensure correct hydraulic application. The influent flows were checked once a week, after the pump was cleaned, and showed minimal variation (± 0.5 ℓ/min).

Effluent runoff was measured using HS flumes developed by the Soil Conservation Service to measure flow rates ranging from 2.27 ℓ/s to 23.22 ℓ/s (0.08 cfs to 0.82 cfs) (Grant 1978). No attempt was made to obtain a complete water balance. The measurements served to approximate the fraction of the applied water that would be recovered as direct runoff for reuse or discharge and for estimating mass loading and removals.

Soil sampling and analyses

Soil samples, before and after the application season, were taken to evaluate the effects of wastewater application on the soil matrix. Soil samples were analyzed by the Utah State University Soil Laboratory for the parameters shown in Tables 11 and 12.

Harvesting

During the first year of operation (June 1979 to October 1979), the grass was harvested during the 2-day rest period between the fourth and fifth weeks of application and then again at the end of the growing season (October). A gasoline lawn mower was used to cut the grass. The clippings were then raked and removed. A substantial amount of residue, however, remained on the slopes.

The cover crop was harvested once in the spring before the operation began and twice during the second year study period (June 9, 1980, to October 23, 1980). The initial harvest helped to remove dead clumps of grass that would have inhibited uniform wastewater flow over the slopes. It took from April 29, 1980, to May 3, 1980, to complete the cropping with the use of gasoline lawn mowers and a hand driven sickle-bar mower. The harvest was complicated by rainfall and equipment breakdown. Additional time (2 to 3 days) was included for drying the clippings on the slope. Application of raw (screened, degritted) wastewater began June 9, 1980, approximately one month after the first cropping. Subsequent harvesting was conducted after a 5- to 6-day rest period with no wastewater application. The system was down a total of 9 days for drying the slopes, cutting, drying the clippings, and raking. The latter harvests were intended to maximize crop removal and minimize mulching.

Table 11. Summary of soil analyses for the first year of operation (Black 1965).

Analyses	Analyses
Total Percent Nitrogen	pH
Extractable Nitrate-Nitrogen	Sodium Adsorption Ratio (SAR)
Total Percent Phosphorus	Calcium Plus Magnesium
Extractable Phosphate-Phosphorus	Cation Exchange Capacity (CEC)
Sodium	Electroconductivity (EC_e)
Texture	Potassium
	Organic Carbon

Table 12. Soil analyses performed for the second year of operation (Black 1965).

Texture
CEC, meq/100 g
pH
EC_e
Sodium, meq/100 g[a]
Potassium, mg/l[a]
Chloride, meq/l
Phosphorus, mg/l
Nitrate, mg/l
Org. Carbon, percent
Bicarbonate, meq/l
Iron, mg/l
Zinc, mg/l
Copper, mg/l
Exch Sodium, meq/100 g
Exch Potassium, meq/100 g
Exc Calcium, meq/100 g
Exch Magnesium, meq/100 g

[a]H_2O soluble.

The second harvest was executed on July 25, 1980, with a tractor driven rotary blade. However, this method was also inadequate. The rotary blade mower compacted the grass beneath the tractor wheels and mulched the grass within each swath. This hampered raking as well as crop removal efforts. By the third harvest on September 18, the proper equipment was used. A tractor driven 5 foot sickle-bar blade located on the side of the tractor cut the grass in one piece which facilitated raking and maximized crop removal.

The effects of harvesting on system performance were ascertained by monitoring the variation in effluent quality during the subsequent application period. Daily composite samples were collected and analyzed for the parameters listed in Table 10. Daily composite samples were also obtained prior to harvesting and analyzed for the same parameters (Table 10).

Data Analysis

Comparisons between the performances of the sections were based on the data obtained from the Thursday samples. The daily data obtained after harvesting were omitted because they were not representative of equilibrium conditions. Statistical comparisons were performed using Duncan's multiple range test (Middlebrooks 1976).

RESULTS AND DISCUSSION

System Hydrology

Wastewater applied to overland flow slopes either infiltrates into the soil, becomes evapotranspiration, or flows into collection channels. Overland flow systems should have a uniform depth of flow. This requires smooth surfaces, uniform application of wastewater at the upper end, and a sod forming grass (Bouwer and Chaney 1974, Hinrichs et al. 1980). Unavoidable surface roughness or vegetation variations invariably cause some channeling and short circuiting (Kemp et al. 1978, Peters 1978, Hall et al. 1979). It was anticipated that the gravel layer would improve flow dispersion and minimize channeling and short-circuiting.

Short-circuiting, in effect, increases the loading rate. The design flow rate assumes complete flow coverage. If wastewater, however, is only flowing down half of the surface, because of short-circuiting, the actual hydraulic and organic loading rates on that half are twice the intended loading rates. Thus, increasing the flow coverage decreases the difference between the actual and design loading rates. Theoretically, a system receiving 10 cm/wk with 50 percent flow coverage could treat 20 cm/wk if 100 percent flow coverage could be attained. Careful construction and maintenance of slopes are therefore critical to optimum performance.

Fluorescein dye studies were conducted during both years of operation to characterize system hydraulics in terms of wastewater dispersion and detention times. The dye was applied as a single pulse input to the influent to the slopes on the last day of the weekly application cycle; thus the results apply to flow characteristics at the end of the cycle.

Dispersion

Results of the dye study for the first year of operation at the loading rates of 13 and 20 cm/wk are shown in Figure 11, and the results for the second year at loading rates of 23, 41, and 57 cm/wk are shown in Figure 12.

Shapes of representative dye curves to characterize the degree of dispersion are shown in Figure 13. Ideal plug flow is manifested by a vertical line with no residual dye before or after the peak, while broad curves indicate dispersion. It has been observed that a sharp peak of solute tends to spread out and disperse as it moves with the applied wastewater over and through the soil. This spreading is due to molecular diffusion and to the variation in flow velocity that exists from place to place on the overland flow slopes.

Except at the lowest loading rate (13 cm/wk during the first year), the slope-rock plots resulted in more dispersion at each loading rate than the overland flow plots, as indicated by broader, flatter dye peaks.

To quantify dispersion, the chemical engineering dispersion index, d, which is calculated from the variance of the dye dispersion curve was used. Ideal plug flow conditions are indicated when the value of d approaches zero (Marske and Boyle 1973). The dispersion indexes obtained for each dye curve are given in Table 13. These values indicate little difference in dispersion between treatment slopes.

Detention times

The peak and mean detention times were also calculated as a means of describing the hydrology of the systems (Table 14). The mean detention time is

Table 13. Dispersion indexes for the first year of operation.

Dispersion Index	20 cm/wk Overland Flow	20 cm/wk Slope-Rock	13 cm/wk Overland Flow	13 cm/wk Slope-Rock	23 cm/wk Slope-Rock	23 cm/wk Overland Flow	41 cm/wk Slope-Rock	41 cm/wk Overland Flow	51 cm/wk Slope-Rock	57 cm/wk Overland Flow
d	0.12	0.10	0.08	0.08	0.14	0.20	0.26	0.35	0.36	0.32

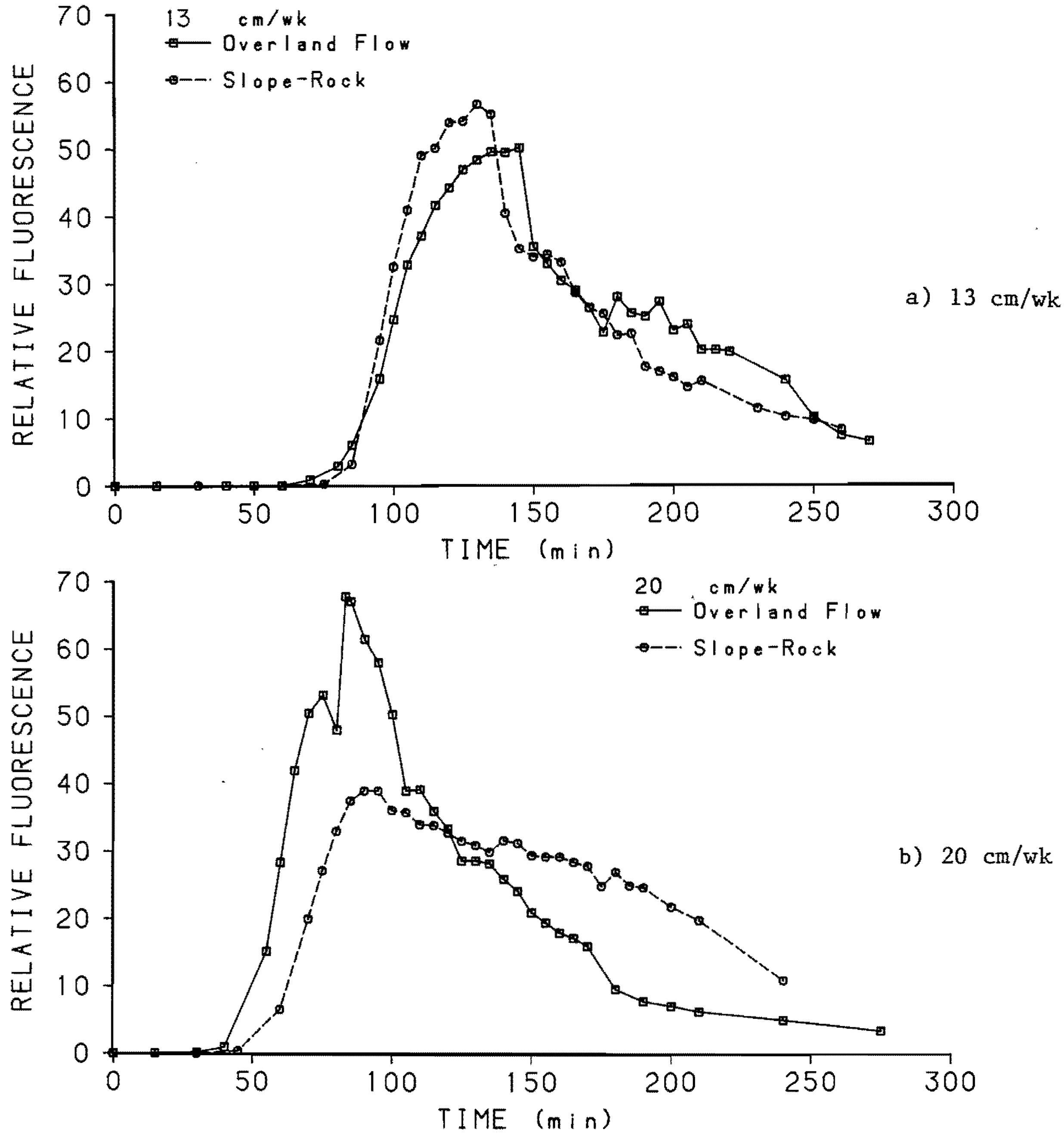
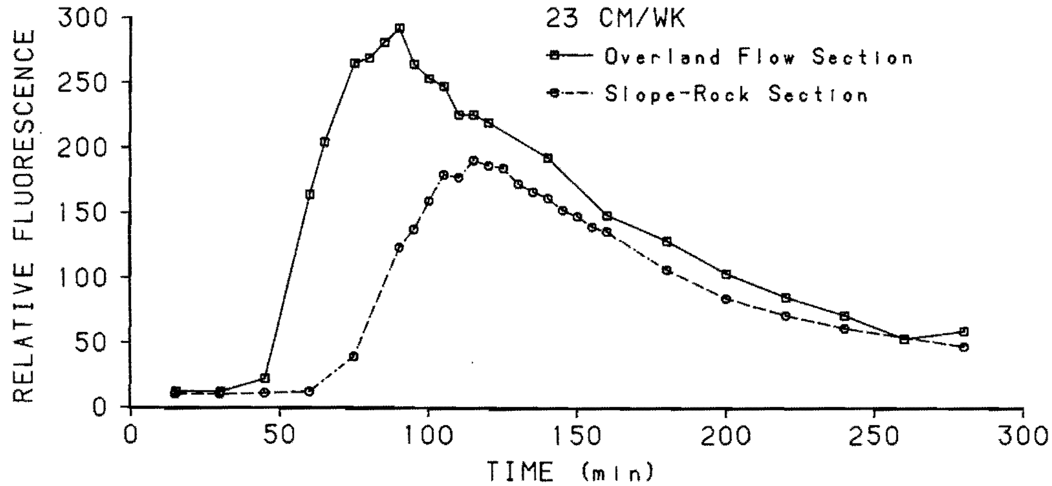
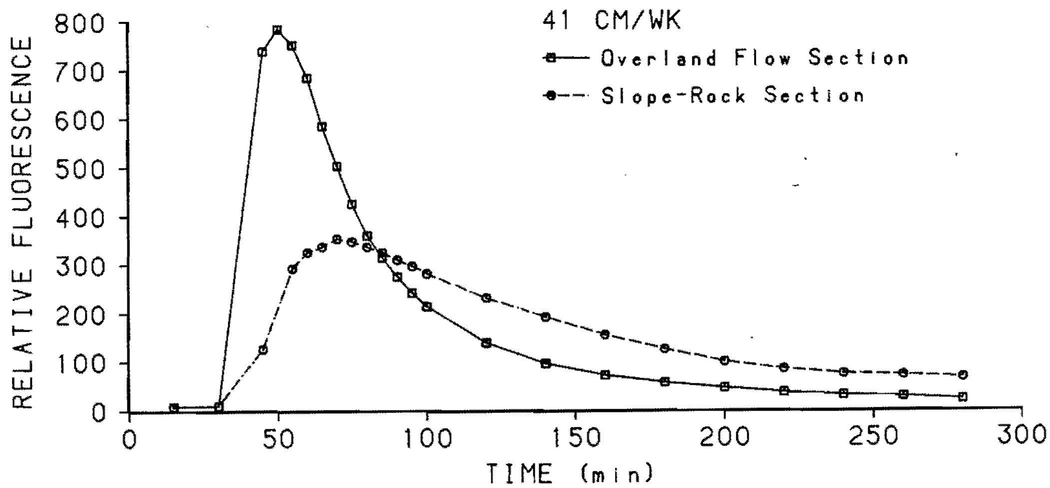


Figure 11. Dye curves for hydraulic application rates of a) 13 cm/wk and b) 20 cm/wk.





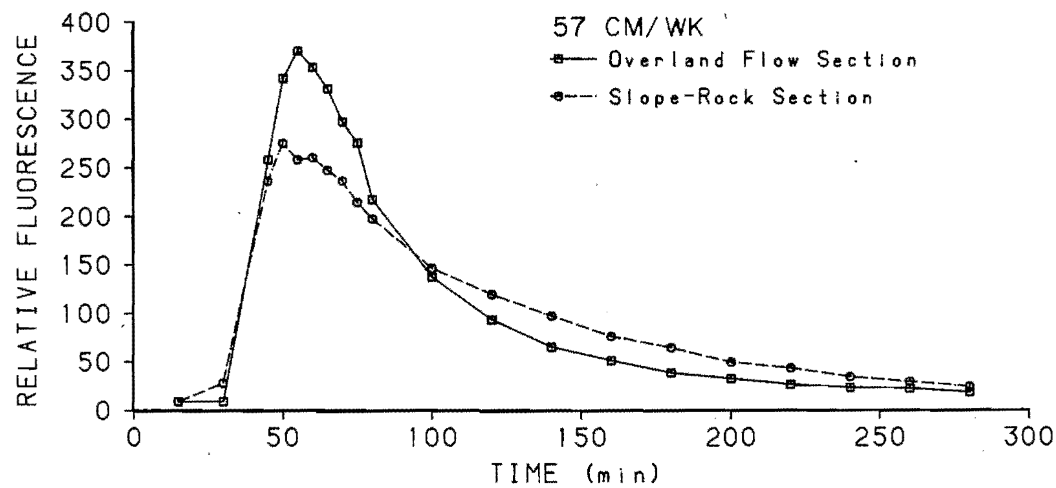


Figure 12. Results of the fluorescene dye study to determine wastewater detention times on the plots for hydraulic loading rates of (a) 23 cm/wk, (b) 41 cm/wk and (c) 57 cm/wk. OF = overland flow, SR = slope-rock.

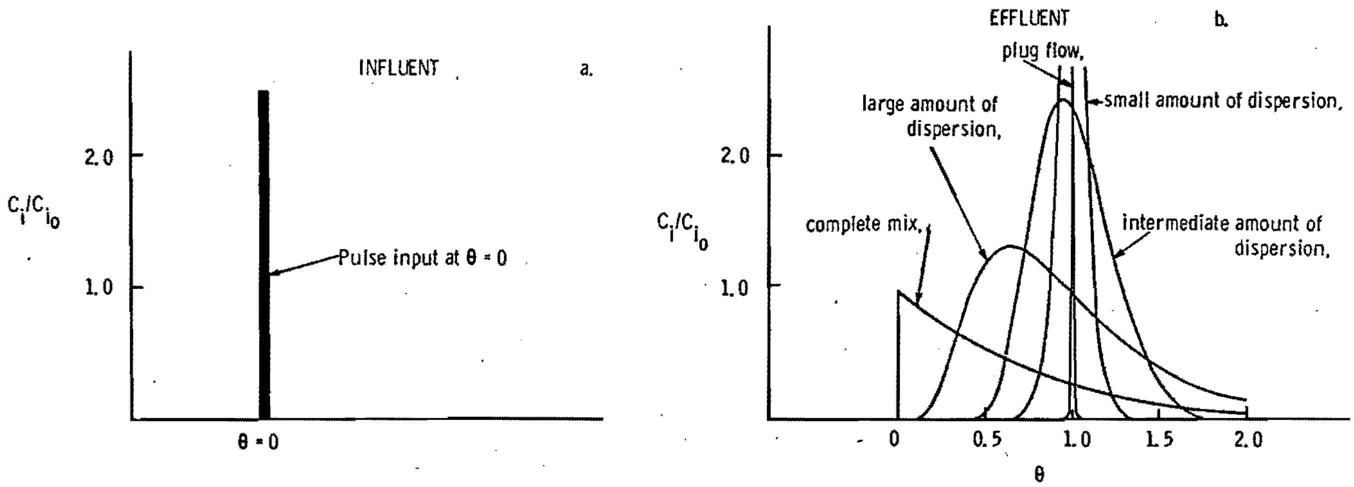


Figure 13. Effluent responses to a pulse input (Weber 1972). (= tV_x/L, where t = time, V_x = velocity, and L = length of reactor).

Table 14. Hydraulic detention times on the treatment slopes.

Year/ Loading Rate/ Treatment	Peak Detention Time (min)	Mean Detention Time (min)	Time Differential Between Peak and Mean Detention Times (min)
Year 1			
13 cm/wk			
Overland Flow	145	154	9
Slope-Rock	130	147	17
20 cm/wk			
Overland Flow	83	106	26
Slope-Rock	93	132	39
Year 2			
23 cm/wk			
Overland Flow	90	138	48
Slope-Rock	114	156	42
41 cm/wk			
Overland Flow	48	90	42
Slope-Rock	72	126	54
57 cm/wk			
Overland Flow	54	96	42
Slope-Rock	48	114	66

defined as the time to reach the centroid of the dye curve, while the peak detention time (or mode) is defined as the time to reach peak concentration (Marske and Boyle 1973). Except for the 13 cm/wk loading rate, the addition of the gravel layer increased the mean detention time at each loading rate. Insufficient flow to cause a significant head loss through the gravel layer may have accounted for the lack of difference seen at the 13 cm/wk loading rate.

The peak detention times were also longer on the slope rock plots compared to the corresponding overland flow plots, except for the 13 cm/wk and the 57 cm/wk plots. The wastewater at the 57 cm/wk flow rate bypassed the gravel layer to some extent because the depth of the wastewater exceeded the thickness of the gravel layer. Thus, head loss did not continue to increase with flow rate.

Figure 14 shows the contrast in flow across the gravel zone between a lower loading rate (23 cm/wk) and the highest loading rate (57 cm/wk). The 23 cm/wk hydraulic loading trickles through the gravel, where as the 57 cm/wk flow rate covered the rock layer, bypassing the rock layer to some extent.

Solids deposition also helped carry the water over the gravel layer at higher loadings. The solids had to be removed from the gravel at the beginning of each week. At the lower loading rates (13, 20, and 23 cm/wk), most of the suspended solids settled in the influent troughs, while at the higher loadings (41 and 57 cm/wk), the grass and gravel trapped most of the solids in the first 9 m, causing blocking of flows (Figure 15). Channels in the larger gravel zone were needed to reduce the head loss through the first 2 m.

The overland flow slopes also experienced larger head losses due to solids accumulation in the upper reaches of the slope at higher loading rates (41 and 57 cm/wk). The combination of tall grass and solids accumulation caused the wastewater to back up. This also caused the grass to fall down and mat, further impeding flow. The grass had to be physically parted to allow wastewater through, until the grass could be cut. Care should be taken to cut and remove clippings and dead grass mats at the

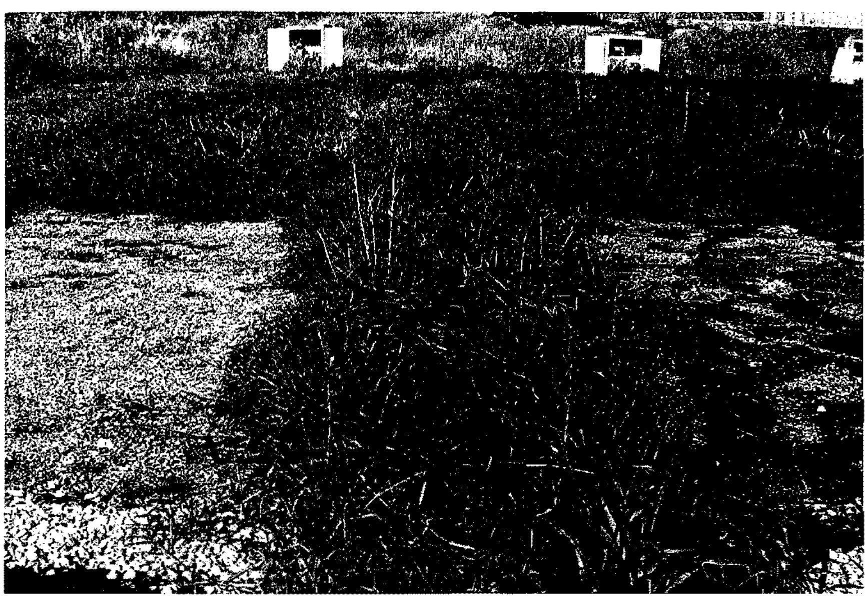


Figure 14. Flow patterns observed on the 23 cm/wk and 57 cm/wk hydraulically loaded slope-rock fields.

head of slopes to assure proper wastewater distribution.

Comparison of time differentials between peak and mean detention times for each pair of loading rates indicated greater dispersion of wastewater on the slope rock plots (Table 13). Except for the 23 cm/wk loading rate, the time differentials increased for the slope rock plots with increased loading rate. However, for the overland flow plots, the time differentials increased from 13 cm/wk to 20 cm/wk but remained relatively constant at the higher rates.

In general, increased hydraulic loading rate decreased detention time of wastewater on the plots. The longest peak detention times occurred on the plots loaded at 13 cm/wk, while the longest mean detention times occurred on the 13 cm/wk plots and the 23 cm/wk slope rock plot.

The shortest mean and peak detention times occurred on the 57 cm/wk plots and the 41 cm/wk overland flow slope. The shorter times on the 41 cm/wk plot were probably due to increased velocity resulting from channeling, caused by about 10 percent of the plot being a few centimeters higher than the rest of the plot. The application rate obtained from subtracting the area not covered by flow was approximately 46 cm/wk.

The five application rates treated on the overland flow plots and the slope rock plots were regressed against both mean and peak concentration times, with the results shown in Table 19. Correlations between hydraulic flow rate and peak detention time were better than correlations between flow rate and mean detention time (Table 15). Correlation coefficients generally did not improve with log-log transformations.

Runoff and hydrologic balance

As the application cycle progressed through the week, the lag time to runoff decreased (Table 16, data for second year). Surface soils were wetted and possible cracks in the clay subsurface were sealed by the wetting. By the third day, conditions stabilized, yielding a constant rate of discharge.

Figure 15. Solids deposition occurred both in the influent troughs and on the gravel layer.

The gravel layer reduced the amount of discharge water recovered form the plots and increased the percent of wastewater lost to evaporation, infiltration, and subflow (water passing along the clay-top soil boundary layers) according to the hydrologic balance presented in Table 17. Direct measurements of evapotranspiration, infiltration, or subflows were not made but estimated from the direct measurements of influent and effluent flows. Water storage and more uniform flow distribution in the gravel layer favored increased infiltration. Water ponding was often observed on the surface of the gravel layers at the higher flow rates. An increase in flow coverage created a greater water surface area for evaporation and ensured that adequate water was available for all the vegetative cover, thereby possibly increasing transpiration.

The amount of water recovered generally increased with increased flow rate. However, the lowest recovery was seen on the 41 cm/wk slope rock plot. This plot was on the periphery of the test area and suffered from uneven settling after construction. Clay berms with plastic linings were installed to reduce the water loss, but ponding on the side of the plot was still noticeable.

Table 15. Regression equations of detention times as a function of flow rates.

Treatments/Detention Time	Equations	Correlation Coefficient (r^2)
Overland Flow		
Mean (min)	-1.15 x flow (cm/wk) + 152.26	0.56
Peak (min)	-1.73 x flow (cm/wk) + 138.02	0.67
Log (mean)	-0.32 x log (flow) + 2.52	0.67
Slope-Rock		
Mean (min)	-0.91 x flow (cm/wk) + 161.27	0.69
Peak (min)	-1.73 x flow (cm/wk) + 144.27	0.86

Table 16. Lag times to collection of first discharge from the plots (year 2).

Loading Rate/ Treatment	Time Before Runoff Occurred (min) Day of Weekly Application Cycle				
	1	2	3	4	5
23 cm/wk					
Overland Flow	150	78	84	72	78
Slope-Rock	216	132	108	108	114
41 cm/wk					
Overland Flow	108	60	60	54	54
Slope-Rock	198	108	90	90	90
51 cm/wk					
Overland Flow	84	54	48	48	48
Slope-Rock	138	84	78	78	78

Table 17. Hydrologic balance of overland flow and slope-rock plots.

Year/Loading Rate/ Treatment	Flows				
	Applied (ℓ/min)	Recovered (ℓ/min)	Loss to Evapo-transpiration, Infiltration, and Subflow (ℓ/min)	Percent Recovered (%)	Percent Lost to Evapo-transpiration, Infiltration, and Subflow (%)
Year 1					
13 cm/wk					
Overland Flow	6.8	4.2	2.7	61	39
Slope-Rock	6.3	3.1	3.2	49	51
20 cm/wk					
Overland Flow	10.1	8.8	1.4	87	13
Slope-Rock	13.0	7.2	5.8	55	45
Year 2					
23 cm/wk					
Overland Flow	11.3	9.5	1.8	84	16
Slope-Rock	11.3	6.4	4.9	57	43
41 cm/wk					
Overland Flow	22.6	16.7	5.9	74	26
Slope-Rock	22.6	9.2	13.4	41	59
51 cm/wk					
Overland Flow	32.0	26.7	5.3	83	17
Slope-Rock	32.0	25.1	6.9	78	22

Two-year summary

The addition of the gravel layer increased detention time. A slower application rate also increased detention time. Short-circuiting of the wastewater on the slopes decreased detention time. Therefore, maintaining a level cross-section is necessary for maximum hydraulic detention time.

More runoff was recovered from the overland flow slopes than their counterpart slope-rock sections.

Soil Analyses

Soil analyses for the two years of operation are presented in Tables 18 and 19. Single soil samples were taken from the top, middle, and bottom of each test section. Samples were taken in the spring of each year and again in the fall to determine soil changes due to wastewater application. The analyses of the samples are listed by application rate and also by plot number (Figure 16) for comparison between years. Some of the variations observed were undoubtedly derived from the inherent variability among soil samples from the same source.

Both the topsoil and the impermeable subsoil contained clay. The cation exchange capacity (CEC) was

Table 18. Summary of initial and final soil data for the first year of operation.

	Initial Subsoil Sample	Initial Topsoil Samples (Spring)	Final Topsoil Samples (Fall)			
			20 cm/wk Overland Flow	20 cm/wk Slope-Rock	13 cm/wk Overland Flow	13 cm/wk Slope-Rock
Plot Number			3	4	2	5
Texture	Clay	Clay	Clay	Clay	Clay	Clay
pH	7.7	7.7	7.6	8.0	7.8	7.6
Electro-conductivity EC_e (mmhos/cm)	1.0	1.5	1.1	1.1	1.2	1.0
Extractable Phosphate (ppm)	19	14	22	37	14	31
Total Phosphorus (%)	0.12	0.09	0.13	0.14	0.13	0.15
Potassium (ppm)	287	340	214	>400	214	318
Total Nitrogen (%)	0.10	0.17	0.18	0.21	0.19	0.20
Nitrate-nitrogen (ppm)	25.0	8.6	23.0	4.6	13.0	15.0
Sodium (meq/l)	1.5	3.6	3.5	3.4	4.2	4.0
Calcium plus Magnesium (meq/l)	8.0	11.5	8.5	7.2	7.2	7.2
Sodium Adsorption Ratio (SAR)	0.8	2.0	1.7	1.8	2.2	1.6
CEC (meq/100 g)	18.9	25.1	23.4	27.0	29.0	27.0
Organic Carbon (%)	1.0	1.6	1.6	2.1	1.9	1.9

Table 19. Soil analyses results for second year of operation.

	15 cm Depth (6 in.)											
	23 cm/wk Slope-rock		23 cm/wk Overland Flow		41 cm/wk Slope-rock		41 cm/wk Overland Flow		51 cm/wk Slope-rock		57 cm/wk Overland Flow	
	Spring	Fall	Spring	Fall	Spring	Fall	Spring	Fall	Spring	Fall	Spring	Fall
Plot Number	5		2		6		1		4		3	
Texture	SiCL[a]		SiCL		SiCL		SiCL		SiCL		SiCL	
pH	7.7	7.7	7.6	7.8	7.6	7.9	7.6	7.7	7.6	7.8	7.6	7.8
EC_e (mmhos/cm)	0.5	0.7	0.6	0.9	0.5	0.7	0.5	0.8	0.5	0.8	0.5	0.7
CEC[b]meq/100 g	21.6	22.8	22.8	16.6	22.6	20.3	22.6	19.8	22.4	18.9	23.3	21.2
Saturation Percent	58	64.4	58	51.4	57	55.9	61	59.3	61	52.6	60	59.3
Iron mg/l[c]	12	18	16	22	11	21	12	18	13	25	12	18
Zinc mg/l[c]	1.2	0.9	1.3	1.4	1.3	1.3	1.7	1.4	1.5	1.2	1.4	1.3
Copper mg/l[c]	2.0	2.0	2.7	2.6	1.9	2.2	2.5	2.2	1.9	2.4	2.5	2.8
Phosphorus mg/l[d]	7.6	18.0	10	19	6.7	7.8	7.6	24	19	11	7.1	11
Nitrate-N mg/l	2.0	3.6	1.5	1.9	1.2	3.0	1.2	4.1	6.5	5.4	1.5	3.6
Sodium meq/100 g[e]	0.5	0.7	0.6	0.6	0.6	0.7	0.5	0.7	0.3	0.8	0.6	0.7
Potassium meq/100 g[e]	1.0	1.2	1.0	0.9	1.6	0.9	1.1	1.1	1.2	0.9	1.2	0.9
Calcium* meq/100 g[e]	46.9*	32*	40.6*	49.5*	40.9*	44.4*	39.8*	38.3*	41.7*	40.1*	42.2*	39.5*
Magnesium* meq/100 g[e]	8.8*	8.4*	8.7*	7.2*	8.4*	7.8*	8.7*	7.8*	8.3*	8.0*	8.9*	7.9*
% Lime ($CaCO_3$)	+	3.0	+	4.9	+	3.1	+	3.7	+	3.7	+	2.9
Sodium meq/100 g[f]	0.1	0.18	0.2	0.17	0.1	0.17	0.1	0.18	0.1	0.20	0.2	0.17
Potassium meq/100 g[f]	<0.1	0.02	<0.1	0.02	<0.1	0.01	<0.1	0.02	<0.1	0.01	<0.1	0.01
Calcium meq/100 g[f]	0.1	0.15	0.1	0.14	0.1	0.11	0.1	0.15	0.1	0.11	0.1	0.14
Magnesium meq/100 g[f]	0.1	0.11	0.1	0.10	0.1	0.08	0.1	0.10	0.1	0.08	0.1	0.08
% organic carbon	1.4	1.76	1.5	0.91	1.5	1.35	1.5	1.57	1.5	0.72	1.4	1.50
	46 cm Depth (18 in.)											
Texture	SiCL[a]		SiCL		SiCL		SiCL		SiCL		SiCL	
pH	7.6	7.8	7.6	7.8	7.8	8.1	7.6	7.8	7.6	7.9	7.7	7.9
EC_e (mmhos/cm)	1.0	0.8	0.8	0.7	0.7	0.9	0.9	0.8	0.6	0.6	0.7	0.8
CEC meq/100 g	17.3	16.5	20.0	17.7	15.0	17.2	21.0	18.5	17.1	16.8	17.7	16.3
Saturation Percent	53	49.9	57	50.9	48	54.5	55	54.4	49	52.8	55	48.4
Iron mg/l[c]	13	19	14	18	16	16	13	30	16	23	14	18
Zinc mg/l[c]	1.1	1.4	1.2	2.4	0.9	1.6	1.4	1.3	1.3	1.8	1.1	1.5
Copper mg/l[c]	1.8	2.8	2.0	2.7	1.6	2.9	1.9	2.4	1.9	3.4	2.0	2.2
Phosphorus mg/l[d]	10	13.0	11	13	10	9.0	11	13.0	12	10	9.4	14.0
Nitrate-N mg/l	2.0	1.2	6.2	9.1	0.7	0.9	1.5	5.5	3.5	2.1	2.1	3.5
Sodium meq/100 g[e]	0.5	0.5	0.3	0.4	0.5	0.1	0.4	0.5	0.4	0.6	0.4	0.5
Potassium meq/100 g[e]	0.8	0.9	9.6	0.9	0.6	<0.1	9.5	1.0	0.8	0.8	0.7	0.8
Calcium* meq/100 g[e]	50.8*	55.5*	43.3*	48.1*	49.5*	56.5*	38.6*	31.3	53.6*	44.4*	52.8*	49.4
Magnesium* meq/100 g[e]	7.7*	7.3*	7.8*	6.2*	6.5*	10.3*	8.3*	7.2	7.8*	7.4*	7.8*	7.2
% Lime ($CaCO_3$)	++	6.6	+	3.9	++	10.5	+	3.1	++	4.5	++	6.2
Sodium meq/100 g[f]	0.2	0.13	0.1	0.08	0.1	0.29	0.1	0.12	0.1	0.14	0.1	0.13
Potassium meq/100 g[f]	<0.1	0.01	<0.1	0.02	<0.1	0.01	<0.1	0.02	<0.1	0.01	<0.1	0.01
Calcium meq/100 g[f]	0.2	0.13	0.2	0.14	0.1	0.07	0.2	0.15	0.1	0.10	0.1	0.11
Magnesium meq/100 g[f]	0.1	0.10	0.1	0.10	0.1	0.09	0.1	0.10	0.1	0.07	0.1	0.08
% organic carbon	0.9	0.88	1.1	0.90	0.7	0.61	1.2	1.15	0.9	0.75	0.9	0.81

*When lime is present in soils, extractable Ca is without meaning, and extractable Mg is often unreliable.
[a]Silty Clay Loam
[b]Cation exchange capacity
[c]DTPA extractable
[d]$NaHCO_3$
[e]NH_4OAc
[f]H_2O soluble

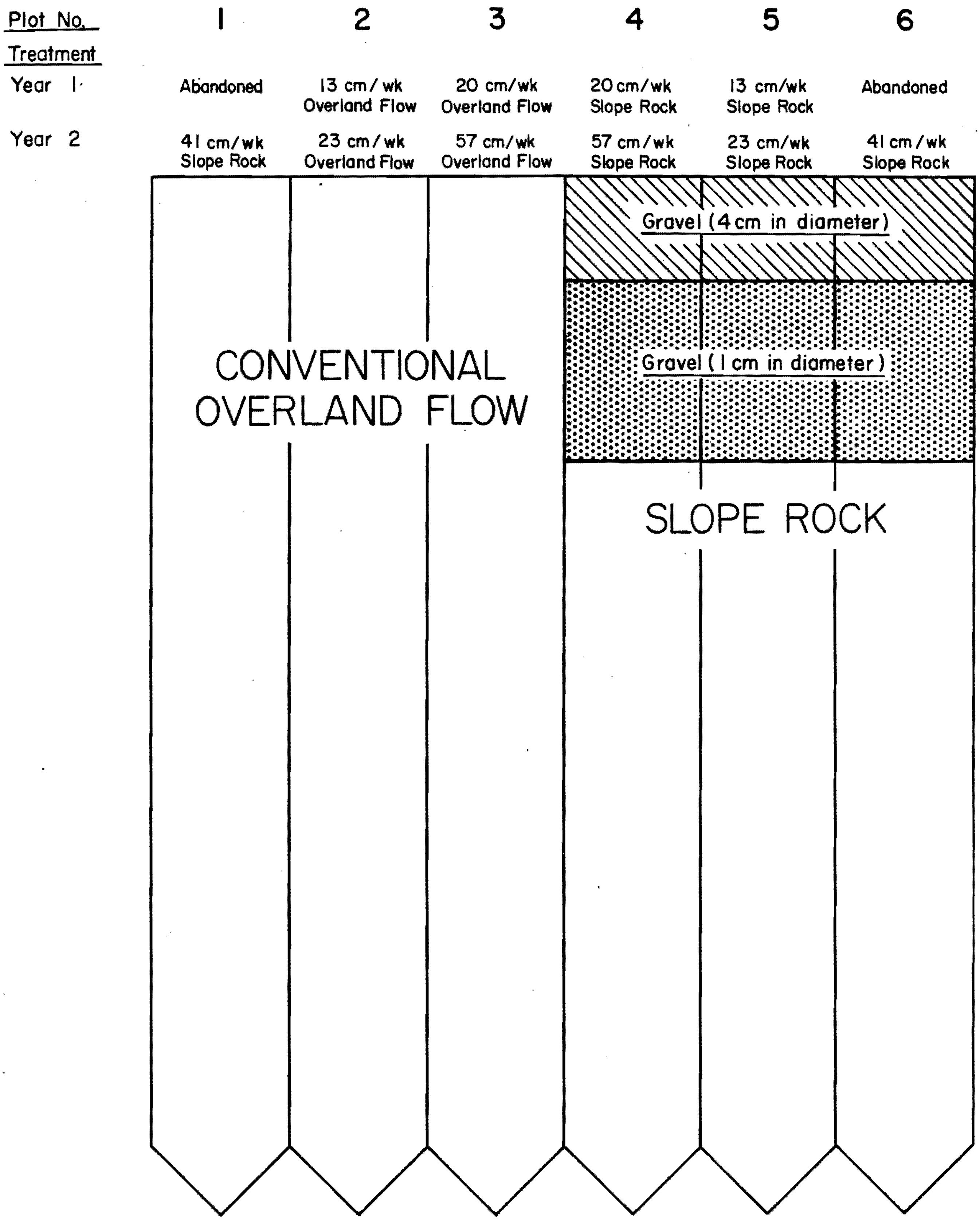


Figure 16. Plot numbers assigned to wastewater treatment fields.

slightly higher in the top 15 cm than at the 46 cm depth. The CEC declined slightly after the second year of operation but increased during the first year from 19 meq/100 g to 23 to 29 meq/100 g. Soil pH ranged from 7.6 to 8.1 with no apparent change. Electroconductivity of the soil solution declined, probably due to salt leaching from the fields. The sodium concentration, however, did not change. Exchangeable calcium and magnesium values are variable due to the presence of lime naturally occurring in the soils. The metals zinc and copper did not change. Significant levels of heavy metals were probably not present in the applied wastewater.

The carbon to nitrogen ratio the first year was about 10:1, suggesting that organic carbon did not limit potential denitrification (Mitchell 1974). Iron, clay, and calcium may be responsible for the increase in soil nitrogen and phosphorus concentrations. The CEC, percent organic carbon, and phosphorus and nitrogen concentrations were greater in the topsoil where most of the wastewater renovation occurred.

A significant increase in total iron was noted in the soil from all plots. Therefore, the increase in iron must have come from the applied wastewater. The wastewater, however, was not evaluated for iron content.

Biochemical Oxygen Demand and Suspended Solids

Biochemical oxygen demand

Complete biochemical oxygen demand (BOD_5) removal is not possible with overland flow treatment of wastewater because leaching of organic material and microorganisms from the soil or surface microbial mass contributes BOD_5 to the effluent (Jenkins et al. 1978, Kemp et al. 1978, Lee and Peters 1978, Thomas 1978). Federal and state standards require 85 percent efficiency in BOD_5 removal and, by 1985, the State of Utah will have a standard of 15 mg/l for BOD_5.

Both the overland flow and slope rock plots met these requirements by averaging 87 to 93 percent BOD_5 removals for both years of operation, and with maximum mean BOD_5 levels of only 12 mg/l (Table 20). Statistical comparisons of the means for the first year of operation indicated no significant difference in effluent BOD_5 among the treatments. During the second year, a significant difference was found in effluent BOD_5 concentrations between the two slopes loaded at 41 cm/wk, but no statistically significant differences were found between the two 23 cm/wk slopes nor between the 57 cm/wk slopes. The 41 cm/wk overland flow plot suffered from short-circuiting and channeling which reduced its BOD_5 treatment capacity, but not enough to violate effluent limits.

Both 23 cm/wk hydraulically loaded plots yielded lower BOD_5 effluent concentrations than did the higher loadings. BOD_5 effluent concentrations from both 23 cm/wk plots were significantly lower than the 41 and 57 cm/wk overland flow plots, but were not significantly lower than the 41 and 57 cm/wk slope-rock plots.

The 57 cm/wk slope rock plot exhibited the poorest soluble BOD_5 removal (77 percent). Other researchers have indicated that soluble organic removal decreases when the hydraulic loading rate increases above 15 to 20 cm/wk (Hinrichs et al. 1980). The results of this study also displayed this tendency.

The differences in BOD_5 effluent concentrations could not be attributed to the gravel modification at any of the hydraulic loadings. The biological process which reduces biochemical oxygen demand by overland flow is similar to that in a conventional trickling filter. A bacterial or biological growth, similar to the zoogleal mass growing on trickling filter media (Hinrichs et al. 1980), occurs on the soil surface. Considerable biological growth was observed on the soil surface but not on the gravel layers. This could have been due to the periodic drying periods. The gravel layer was not deep enough to perform as a trickling filter or to have sufficient buffer capacity upon drying (Cook and Wu 1979). The solids on the surface gravel dehydrated noticeably during the overnight resting periods, while solids in the grass stayed wet. Even after 5 days of drying before harvest, the solids in the grass were still damp. The removal of solids from the gravel to reduce clogging may also have removed a portion of

Table 20. Mean influent and effluent BOD_5 concentrations for both years of operation.

Year/ Loading Rate/ Treatment	Influent Total BOD_5 (mg/l)	Influent Soluble BOD_5 (mg/l)	Effluent Total BOD_5			Effluent Soluble BOD_5[4]	
			Mean[2,3] (mg/l)	Standard Deviation (mg/l)	Removal (%)	(mg/l)	Removal (%)
Year 1	101	--	-	24	-		
13 cm/wk							
Overland Flow			7[a]	3	93	-	-
Slope-Rock			8[a]	3	92	-	-
20 cm/wk							
Overland Flow			10[a]	5	90	-	-
Slope-Rock			12[a]	4	88	-	-
Year 2	86	13	-	19	-		
23 cm/wk							
Overland Flow			6[A]	2	93	1	92
Slope-Rock			6[A]	3	93	1	92
41 cm/wk							
Overland Flow			11[C]	3	87	2	85
Slope-Rock			9[AB]	3	89	1	92
57 cm/wk							
Overland Flow			10[BC]	2	88	2	85
Slope-Rock			8[AB]	3	91	3	77

[1]Influent BOD_5 values for the two years of operation are not statistically different.

[2]Number of values used to calculate mean was 14 for the first year and 10-12 for the second year.

[3]Means followed by the same letter are not significantly different at the 0.05 level. Lower case letters are used to compare means within the first year of operation while capital letters are used to compare means within the second year.

[4]No. of data points = 1.

the microbial population responsible for wastewater renovation.

Hydraulic application rate also had no effect on BOD_5 removal (Figure 17). These results confirm an earlier report by Thomas (1976). A plot of BOD_5 removal data from other operating and research overland flow systems (Figure 18) shows scattered results because of differences in climate, soil, site preparation, grass cover, wastewater characteristics, and other local factors. From laboratory and field studies of slow rate and rapid infiltration systems, Lance et al. (1973) and Bouwer et al. (1974) reported that prolonged flooding and obvious depletion of oxygen did not seem to affect BOD_5 removal. data obtained during this research showed no apparent effect of loading rate on overland flow runoff BOD_5 concentration under similar operating conditions. There is, of course, an upper limit to the loading rate determined by the ability of the system to maintain an aerobic environment for degradation of organics and the capacity of the grass to filter out particulate solids.

Weekly variations in influent and effluent BOD_5 concentrations are shown in Figure 19 for the first year of operation. Between weeks 4 and 5, high concentrations were seen when the plots were harvested using a gasoline-powered lawnmower, and the grass removed by raking. Effluent BOD_5 concentrations stabilized after the seventh week of operation and remained below the present state requirement of 25 mg/l and the future requirement of 15 mg/l (Table 1).

The pattern during the second year of operation (Figure 20) showed effluent BOD_5 peaks at weeks 7 and 15 due to crop removal. The slopes were dried prior to harvesting during weeks 6 and 14. No samples were taken during weeks 8 and 11, although wastewater was applied. Data were not recorded for week 17 for the 23 and 57 cm/wk overland flow plots, because the wastewater had flowed from the 57 cm/wk plot to the 23 cm/wk plot. Tall grass which had not been cut and removed during the previous harvest clogged and diverted the flow onto the other plot.

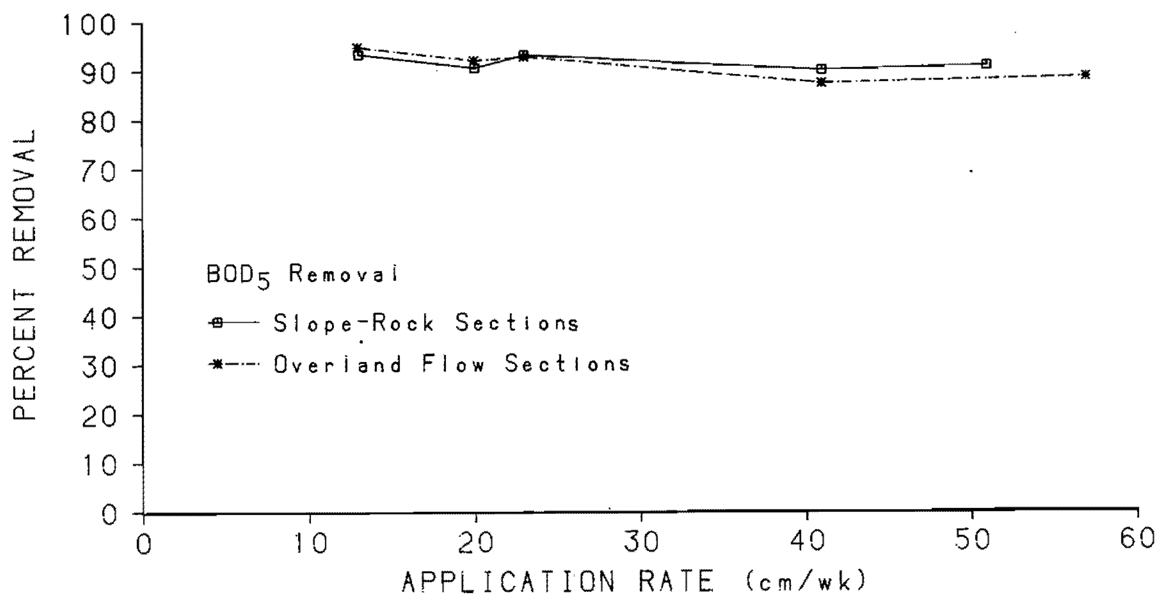


Figure 17. BOD_5 removal versus hydraulic application rate.

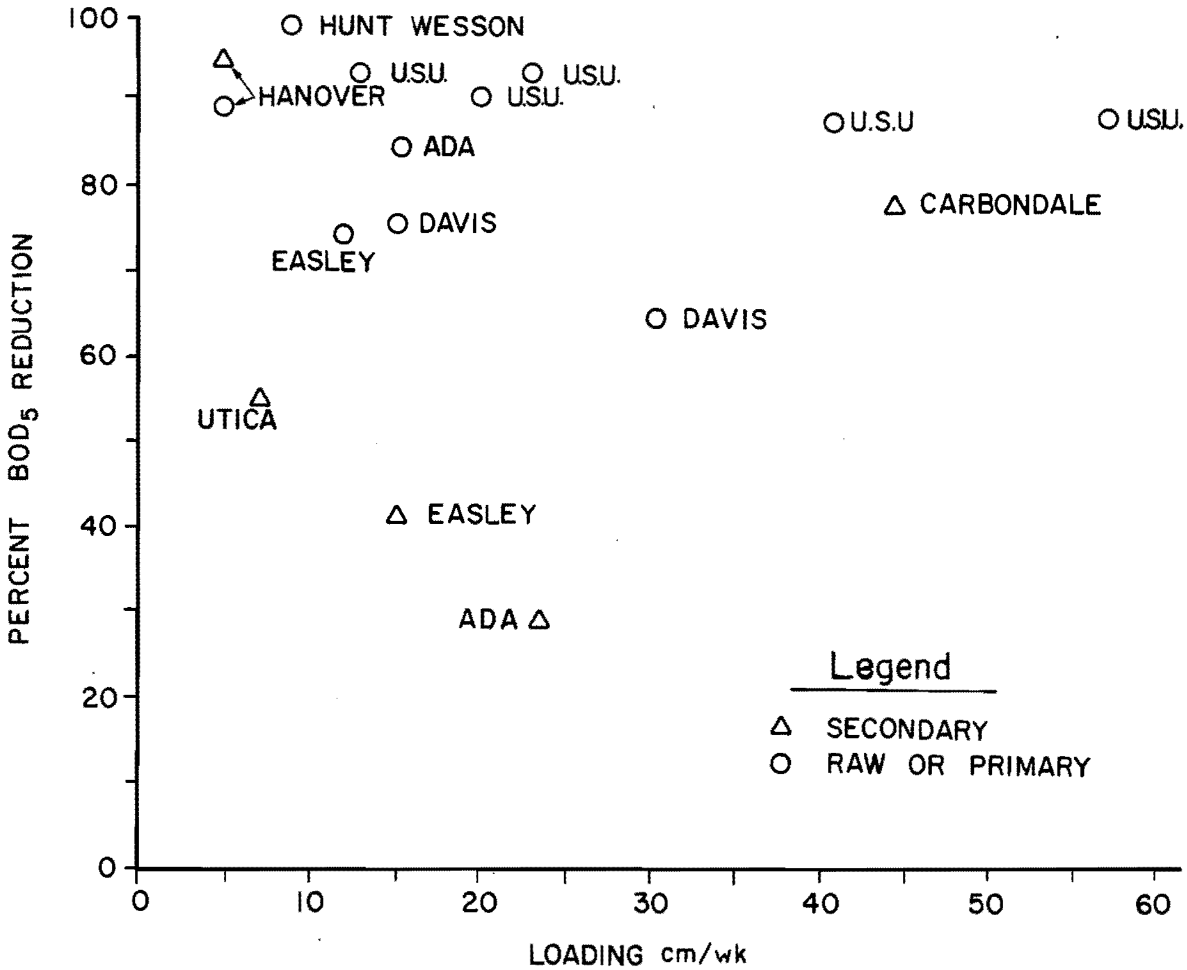


Figure 18. Relationship between percent BOD_5 removal and hydraulic loading rate for overland flow wastewater treatment systems in the United States (from Hinrichs et al. 1980).

Effluent BOD_5 concentrations were higher for week 13. During the week, 1.6 cm of rain inundated the slopes. Wastewater application continued, and the runoff became a pale yellow color, probably from the erosion of suspended solids and algae. Jenkins et al. (1978) recommended that overland flow operations be suspended when precipitation rates exceed 1.3 cm/day to reduce erosion. Martel et al. (1980b) reported that runoff samples taken after a rainstorm showed no flushing effect when wastewater application was discontinued during heavy rain events.

Initial BOD_5 values during the second year were high after winter shutdown. Similar results were seen by Jenkins et al. (1978) and Hinrichs et al. (1980). The effluent from the higher loadings did not stabilize to within acceptable standards until the end of week 7.

Specific mass removal of BOD_5 rates are given in Table 21. These rates, show little or no influence of gravel addition in rate of removal of BOD_5.

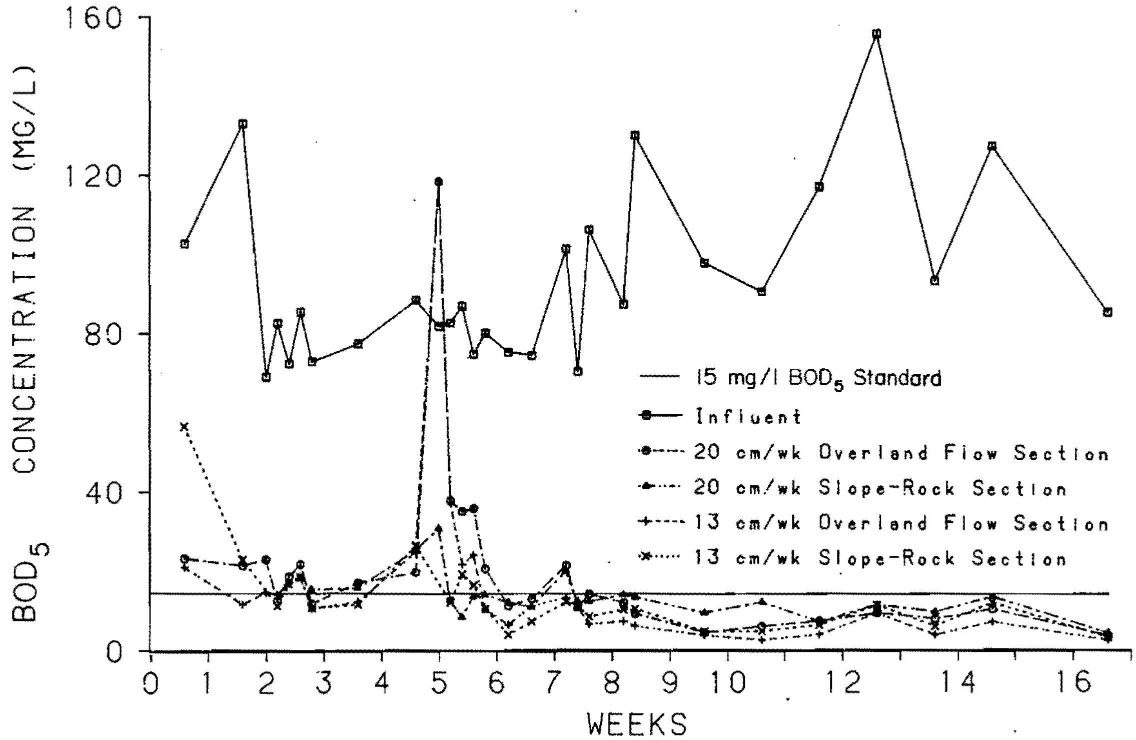


Figure 19. Influent and effluent BOD5 concentrations during the first year of operation.

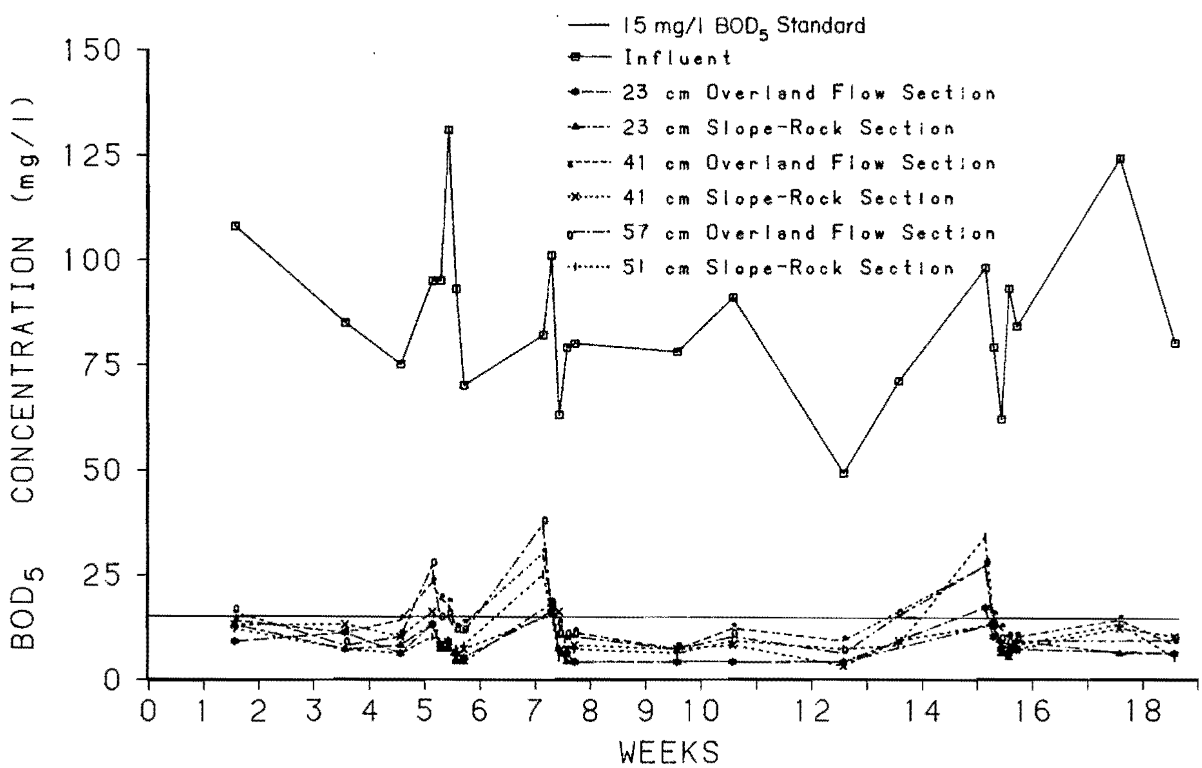


Figure 20. Influent and effluent BOD5 concentrations during the second year of operation.

Table 21. Specific BOD_5 mass removal rates.

Parameter	13 cm/wk		20 cm/wk	
	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock
Year 1: BOD_5 Removal Rate (kg/ha/day)	25	25	38	38

Parameter	23 cm/wk		41 cm/wk		51 cm/wk	
	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock
Year 2: BOD_5 Removal Rate (kg/ha/day)	37	36	67	63	80	89

Increased loading rates, however, resulted in increased mass removal rates.

System age

Removal of BOD_5 by overland flow systems increases with the age of the system (Thomas et al. 1974, Jenkins et al. 1978, Hinrichs et al. 1980). After startup in the spring, treatment efficiency may not be satisfactory until the microbial population has had a chance to establish itself (Jenkins et al. 1978). Thomas et al. (1974) found that approximately 100 days were required.

In this study, only 5 to 7 weeks were required for BOD_5 and suspended solids to stabilize below acceptable levels (Table 1). During the first year, 1 to 2 weeks were required to flush loose soil particles and debris from the systems. Suspended solids levels stablized 2 weeks earlier than BOD_5 in the second year. This was probably due to the grass filter undergoing quicker recovery than the microbial populations. After the winter, clumps of dead grass were cut and removed one month prior to startup. Existing bare spots gradually filled in with new grass.

Suspended solids

All treatments gave satisfactory performance in terms of mean total suspended solids (SS) removal (Table 22). All effluent levels were below 10 mg/l, the proposed 1985 standard. Removal efficiencies ranged from 93 to 97 percent. There was no statistically significant difference in suspended solids removal performance attributable to the addition of gravel to the plots. During the second year of operation, suspended solids removals were slightly lower on the 41 cm/wk and 57 cm/wk plots than on the 23 cm/wk plots. In general, hydraulic loading rate had no effect on suspended solids removal, as shown in Figure 21.

In the influent wastewater, volatile suspended solids (VSS) comprised 80-81 percent of the total suspended solids. During the first year of operation, volatile solids comprised 91 to 95 percent of the total solids in the effluents from the slopes, while during the second year, volatile suspended solids represented only 77 to 88 percent of the total suspended solids. For the second year of operation, linear correlation of VSS with SS gave (with r2 = 0.95):

Table 22. Mean influent and effluent suspended solids concentrations.

Year/Loading Rate/ Treatment	Influent Total Suspended Solids[1] Mean[2] (mg/l)	Influent Total Suspended Solids[1] Standard Deviation (mg/l)	Effluent Total Suspended Solids Mean[2,4] (mg/l)	Effluent Total Suspended Solids Standard Deviation (mg/l)	Effluent Total Suspended Solids Removal (%)	Influent Volatile Suspended Solids[1] Mean[3] (mg/l)	Influent Volatile Suspended Solids[1] Standard Deviation (mg/l)	Effluent Volatile Suspended Solids Mean[3,4] (mg/l)	Effluent Volatile Suspended Solids Standard Deviation (mg/l)	Effluent Volatile Suspended Solids Removal (%)	Volatile Solids as Percentage of Total Solids (%)
Year 1	129	20				104	14				81
13 cm/wk											
Overland Flow			6.3[a]	2.1	95			6.0[a]	2.2	94	95
Slope-Rock			7.1[a]	3.1	95			6.7[a]	3.1	97	94
20 cm/wk											
Overland Flow			7.0[a]	2.1	95			6.5[a]	2.1	94	93
Slope-Rock			7.9[a]	3.3	94			7.2[a]	3.0	93	91
Year 2	102.0	22.6				81.3	4.0				80
23 cm/wk											
Overland Flow			5.6[A]	3.7	95			4.4[A]	3.4	95	79
Slope-Rock			5.2[A]	2.7	97			4.0[A]	2.7	95	77
41 cm/wk											
Overland Flow			9.1[C]	4.0	91			7.8[C]	3.8	90	86
Slope-Rock			7.5[ABC]	4.2	93			6.4[ABC]	3.8	92	85
57 cm/wk											
Overland Flow			8.4[BC]	4.0	92			7.4[BC]	3.7	91	88
Slope-Rock			7.0[AB]	5.4	93			5.8[AB]	4.8	93	83

[1]Influent values for the two years of operation are statistically different.

[2]Number of values used to calculate the means was 12 for the first year and 14-15 for the second year for both influent and effluent SS.

[3]Number of values used to calculate the means was 11 for the first year and 14-15 for the second year for both influent and effluent SS.

[4]Means followed by the same letter are not significantly different at the 0.05 level. Lower case letters are used to compare means within the first year while capital letters are used to compare means within the second year.

$$VSS = -0.61 + 0.92 (SS)$$

As with total suspended solids, the statistical analyses showed the gravel layer at the top of the slope rock plots had no significant effect on VSS effluent quality. Efficiencies of removal of VSS ranged from 90 to 97 percent.

Weekly variations of SS and VSS during the two seasons are shown in Figures 22 and 23. During the first 4 to 5 weeks of operation each year, solids concentrations fluctuated and then stabilized, except for periods of harvesting. Harvesting (between weeks 4 and 5 in the first year and during weeks 7 and 15 in the second year) caused increases in SS and VSS concentrations. SS effluent quality was less affected by the rainfall in week 13 of the second year than was the BOD_5.

Specific mass removal rates are given in Table 23. Mass removal rates were nearly the same between plots hydraulically loaded at the same rate, except for the highest rate (57 cm/wk). This difference is likely due to a difference in actual loading rate due to differences in the surface area of the two plots, resulting in an actual application rate of 51 cm/wk on the slope-rock plot vs. 57 cm/wk on the overland flow plot. Specific mass removals increased with application rate due to the larger mass of wastewater applied.

Intermediate samples

Mean BOD_5 concentrations for influent and intermediate (9 m downslope) samples are given in Table 24. During the first year, removal of BOD_5 in the first 9 m ranged from 59 to 77 percent and during the second year ranged from 48 to 73 percent. Statistical analyses indicated that the addition of gravel did not affect BOD_5 removal at any loading rate, but increased loading rate did decrease BOD_5 removal in the first 9 m.

Weekly variations in BOD_5 concentrations during the second year of operation are illustrated in Figure 24. The intermediate concentrations varied with influent quality and showed substantial removals of BOD_5.

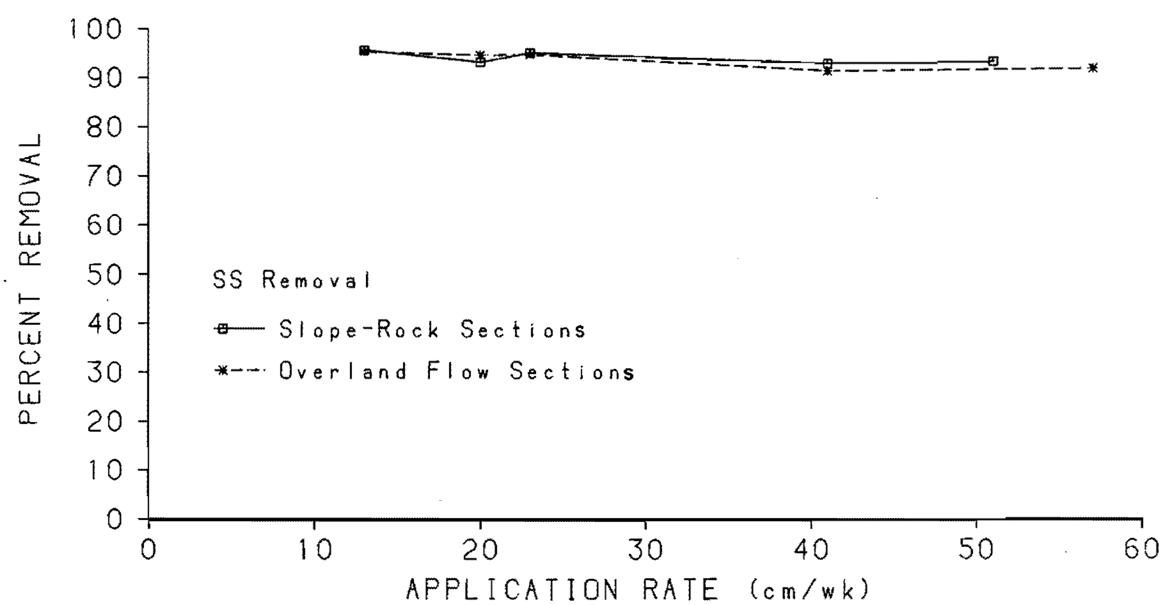


Figure 21. Suspended solids removal versus hydraulic application rate.

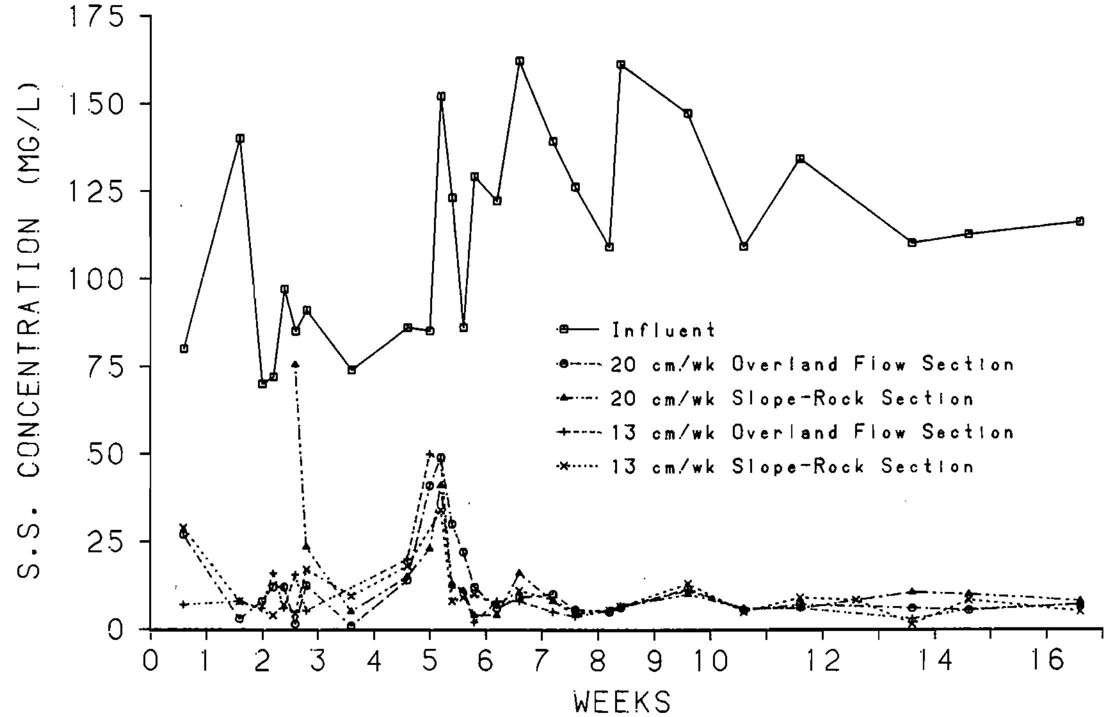


Figure 22a. Influent and effluent suspended solids concentrations during the first year of operation.

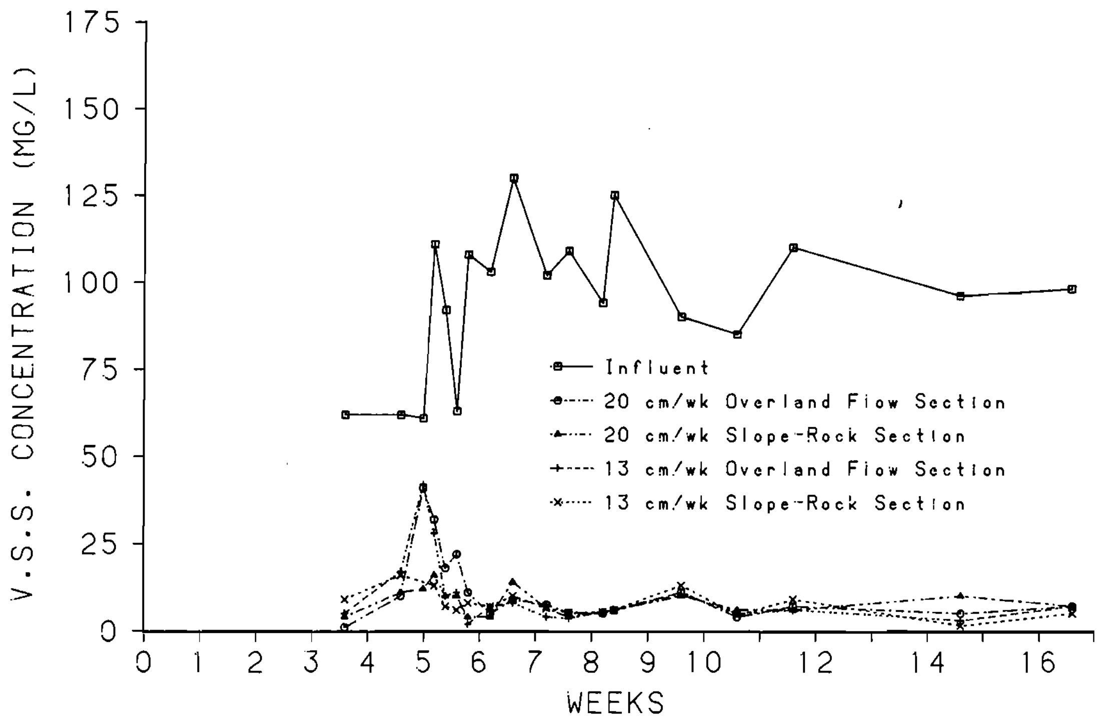


Figure 22b. Influent and effluent volatile suspended solids concentrations during the first year of operation.

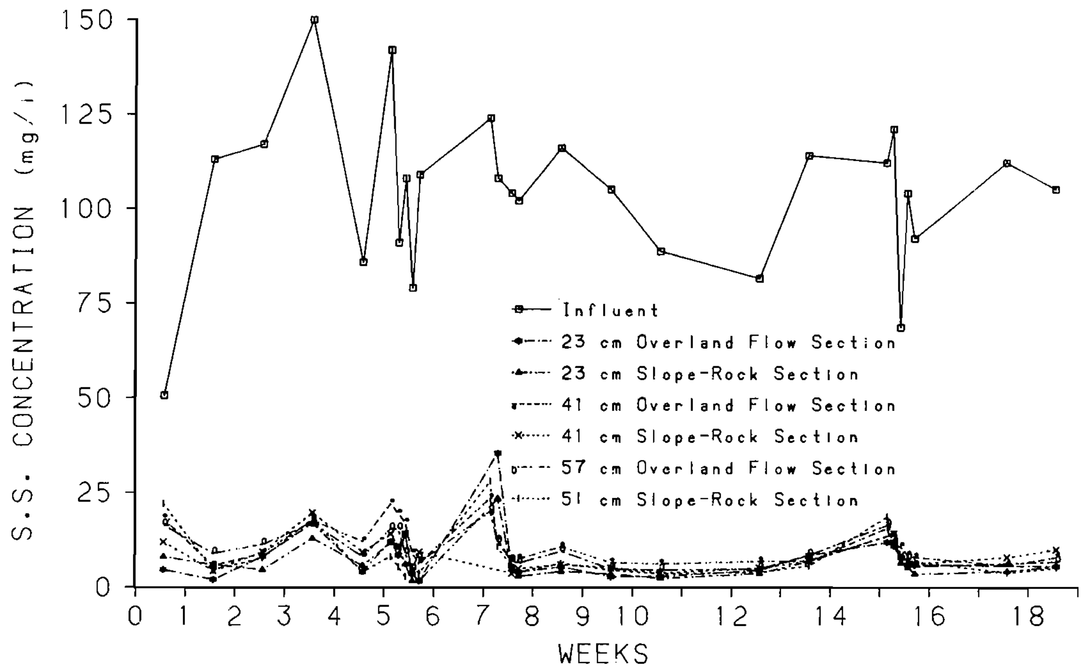


Figure 23a. Influent and effluent suspended solids concentrations during the second year of operation.

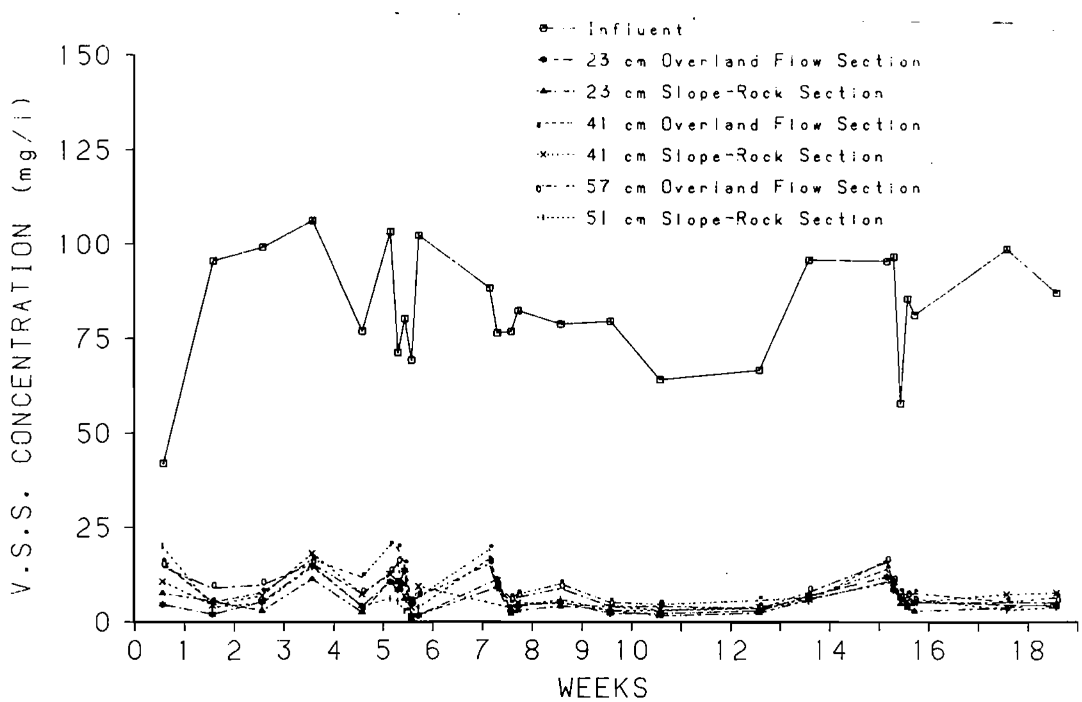


Figure 23b. Influent and effluent volatile suspended solids concentrations during the second year of operation.

Table 23. Specific suspended solids mass removal rates.

Parameter	13 cm/wk		20 cm/wk	
	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock
Year 1:				
Suspended Solids Removal Rate (kg/ha/day)	32	32	50	51
Volatile Suspended Solids Removal Rate (kg/ha/day)	26	26	40	41

Parameter	23 cm/wk		41 cm/wk		57 cm/wk	
	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock	Overland Flow	Slope-Rock
Year 2:						
Suspended Solids Removal Rate (kg/ha/day)	44	45	77	80	109	96

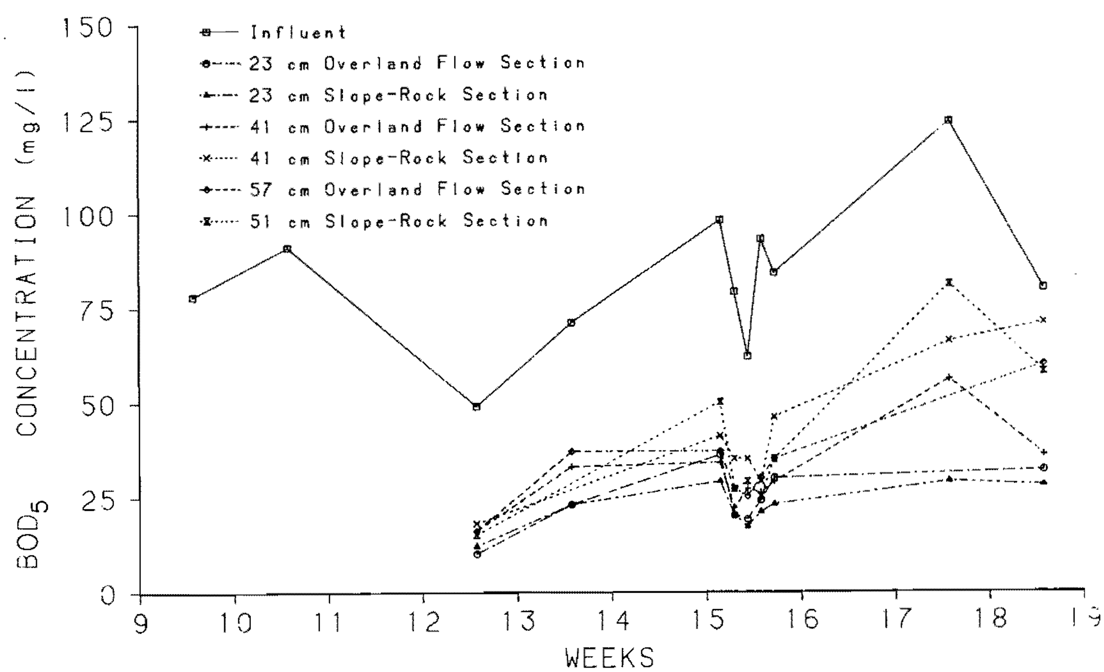


Figure 24. Intermediate BOD_5 variations during the second year of operation.

Sedimentation appeared to be the primary mechanism for removal of BOD_5 in the first 9 m of the plots. Although the samples were not analyzed for total suspended nor volatile suspended solids the first year, a build-up of fibrous, cellulose type material was noted on the first 2 or 3 m of all the slopes. The upper part of the slope rock plots with the larger gravel had to be partially bypassed because the accumulated solids ponded wastewater upslope of the influent point and produced water flow between sections. This solid material was removed once a week. When dry, 50 to 100 cm long sections could be removed intact (Figure 25). The solids on the overland flow sections, however, remained moist, except for occasional drying during a 2-day rest period. The moist environment was more conducive to bio-oxidation of the organic material and incorporation of the more refractory material into the soil humus.

Volatile suspended solids were analyzed in the intermediate samples (Table 25) during the second year of

Table 24. Mean influent and intermediate BOD_5 concentrations.

Year/Loading Rate/ Treatment	Influent BOD_5[1]		BOD_5 at 9 m Sampling Point		
	Mean[2] (mg/1)	Standard Deviation (mg/1)	Mean[3] (mg/1)	Standard Deviation (mg/1)	Removal (%)
Year 1	101	24			
13 cm/wk					
Overland Flow			23[a]	9	77
Slope-Rock			34[a]	20	66
20 cm/wk					
Overland Flow			41[a]	26	59
Slope-Rock			32[a]	15	68
Year 2	83	28			
23 cm/wk					
Overland Flow			24[AB]	8[AB]	71
Slope-Rock			22[A]	16[A]	73
41 cm/wk					
Overland Flow			31[ABC]	11	63
Slope-Rock			43[C]	18	48
51 cm/wk					
Overland Flow			33[ABC]	13	60
Slope-Rock			40[BC]	22	52

[1]Influent BOD_5 values for the two years of operation are not statistically different.

[2]Number of values used in calculating mean was 14 for the first year and 10-12 for the second year.

[3]Means followed by the same letter are not significantly different at the 0.05 level. Lower case letters are used to compare means within the first year of operation while capital letters are used to compare means within the second year.

Table 25. Mean influent and intermediate VSS concentrations for the second year of operation.

Year/Loading Rate/ Treatment	Influent VSS		VSS at 9 m Sampling Point		
	Mean (mg/l)	Standard Deviation (mg/l)	Mean[1] (mg/l)	Standard Deviation (mg/l)	Removal (%)
Year 2	92.0	7			
23 cm/wk					
Overland Flow			37.2	11.0	60
Slope-Rock			37.6	9.1	59
41 cm/wk					
Overland Flow			38.1	8.8	59
Slope-Rock			51.0	13.7	45
51 cm/wk					
Overland Flow			53.5	20.3	42
Slope-Rock			43.2	22.0	53

[1]There was no significant difference among treatments at the 0.05 level.

Figure 25. Solids accumulated on the rock layer.

operation. Total suspended solids were not measured because installation of the samplers loosened the surrounding soil, thus contaminating the samples with soil solids. The volatile solids removed in the first 9 m ranged from 42 to 60 percent. However, there were no significant differences in VSS concentrations among loading rates. Weekly VSS at the intermediate sampling points, as with BOD_5, varied with influent quality (Figure 26).

Two-year summary

The presence of the gravel layer had no statistically significant effect on the effluent BOD_5 or suspended solids concentration. Hydraulic application rate also had practically no effect on BOD_5 or suspended solids removal. Mean BOD_5 removals ranged from 87 to 93 percent, with 48 to 73 percent being removed in the first 9 m. BOD_5 effluent averages ranged from 6 to 12 mg/l. The slopes with the highest loading, 57 cm/wk, were able to meet the 15 mg/l BOD_5 1985 State of Utah discharge standard (30-day average). None of the test sections violated the 10 mg/l suspended solids effluent discharge limit.

The results of this study indicate that heavy rainfall events deteriorate effluent runoff quality from the overland flow and slope-rock fields when wastewater application was not discontinued. The effluent turned from clear to yellow-green and the BOD_5 effluent concentration increased (in one case effluent BOD_5 exceeded 24 mg/l). Effluent suspended solids, however, increased only slightly, remaining below the 10 mg/l suspended solids discharge limit.

Phosphorus

Mean influent and effluent phosphorus concentrations for both years of operation are shown in Table 26. None of the treatments effectively reduced total phosphorus concentrations in the wastewater. Effluent total phosphorus concentrations ranged from 4.44 to 5.27 mg/l, with a range of removal efficiencies of 20 to 33 percent. The observed effluent total phosphorus level was approximately equal to the 5 mg/l total phosphorus concentration predicted by Thomas et al. (1976) for overland flow systems treating raw wastewater and operating at 10 cm/wk but slightly higher than the 4 mg/l predicted by EPA (1977) (Table 4). Removal efficiencies were lower than those predicted by EPA (1975) (Table 3). Additional treatment is necessary if further phosphorus reductions are desired.

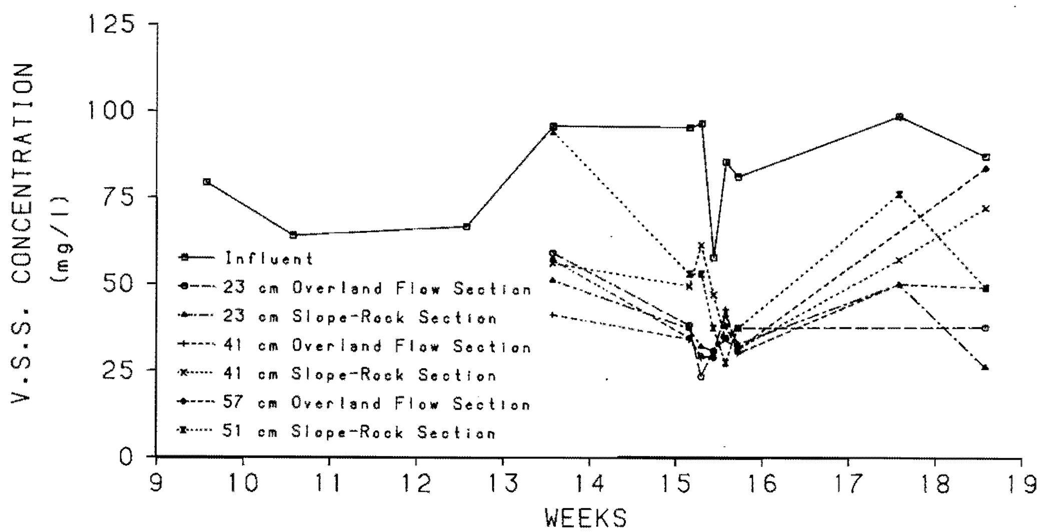


Figure 26. Intermediate volatile suspended solids variations during the second year of operation.

Table 26. Mean influent and effluent phosphorus concentrations.

Year/Loading Rate/ Treatment	Influent Total Phosphorus[1]		Effluent Total Phosphorus			Influent Orthophosphate[2]		Effluent Orthophosphate		Orthophosphate as a Percentage of Total Phosphorus
	Mean[3] (mg/l)	Standard Deviation (mg/l)	Mean[3,5] (mg/l)	Standard Deviation (mg/l)	Removal (%)	Mean[4] (mg/l)	Standard Deviation (mg/l)	Mean[3,5] (mg/l)	Standard Deviation (mg/l)	(%)
Year 1	6.91	0.85				3.64	0.72			53
13 cm/wk										
Overland Flow			4.60^{a}	0.56	33			4.26^{a}	0.70	93
Slope-Rock			5.27^{a}	0.31	24			4.79^{a}	0.66	91
20 cm/wk										
Overland Flow			5.14^{a}	0.43	26			4.72^{a}	0.59	92
Slope-Rock			5.02^{a}	0.27	27			4.62^{a}	0.65	92
Year 2	6.40	1.43				3.09	0.61			48
23 cm/wk										
Overland Flow			4.56^{ABC}	1.08	29			4.07^{AB}	0.76	89
Slope-Rock			4.48^{AB}	0.94	30			4.09^{AB}	0.75	91
41 cm/wk										
Overland Flow			5.12^{D}	1.06	20			4.51^{C}	0.85	88
Slope-Rock			4.44^{A}	0.91	31			3.95^{A}	0.74	89
51 cm/wk										
Overland Flow			4.97^{BCD}	1.04	22			4.39^{BC}	0.76	88
Slope-Rock			5.01^{CD}	1.05	22			4.51^{C}	0.80	90

[1]Influent values for the two years of operation are not statistically different.

[2]Influent values for the two years of operation are statistically different.

[3]Number of values used to calculate the means was 12 for the first year and 14-15 for the second year.

[4]Number of values used to calculate the means was 13 for the first year and 14-15 for the second year.

[5]Means followed by the same letter are not significantly different at the 0.05 level. Lower case letters are used to compare means within the first year while capital letters are used to compare means within the second year.

The small removal efficiencies were due to limited soil-water contact on the overland flow and slope rock plots. Calcium carbonate (lime) and iron (Fe) were available in the soil for adsorption and precipitation of phosphorus. Because of the high pH values of the soil (Tables 18 and 19), most phosphorus removal probably occurred due to the interactions with the calcium, though a small portion of the iron in the soil would be available for reaction. Saturation of the soil during wastewater application may have created anaerobic conditions, resulting in the production of ferrous iron. The ferrous iron may destroy adsorbing surfaces and release phosphate previously adsorbed upon the oxide particles (Taylor and Kunishi 1974). The problem is alleviated by allowing the soil to drain and aerate in the intervals between applications. Harvesting and removing the grass from the slopes as well as microbial immobilization may also have accounted for some of the phosphorus removal in the system.

During the first year of operation, there were no statistically significant differences between effluent means, indicating no effect of loading rate or gravel addition on effluent total phosphorus or orthophosphate levels. During the second year, except for the 41 cm/wk plots, there were no significant differences in mean phosphorus concentrations seen between plots at each loading rate. The differences at the 41 cm/wk plots may be due to differences in water losses on the slope rock plot and severe channeling on the overland flow slope. The statistical analyses also suggest no consistent difference attributable to loading rate. Orthophosphate concentrations did appear to increase slightly with hydraulic application rate (Figure 27), but rate had little effect on total phosphorus levels (Figure 28). Linear regression models for orthophosphorus increase versus flow rate are presented in Table 27. When effluent flows were used instead of application rates, better correlations were obtained.

Orthophosphate phosphorus concentrations, which constituted about 50 percent of the influent total phosphorus, were about 4.5 mg/l in effluent from all the plots, thus constituting about 90 percent of the effluent total phosphorus. These effluent values agree with data presented by Reddy et al. (1978) which indicated that phosphorus losses were 85 to 92 percent

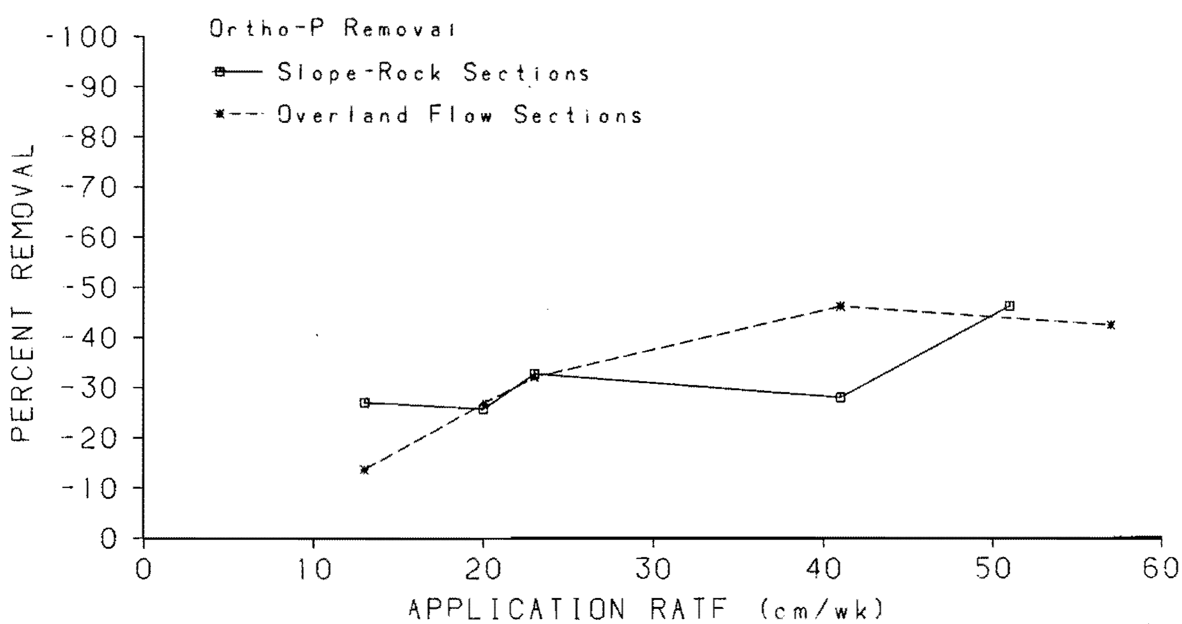


Figure 27. Orthophosphorus concentration increase versus hydraulic application rate.

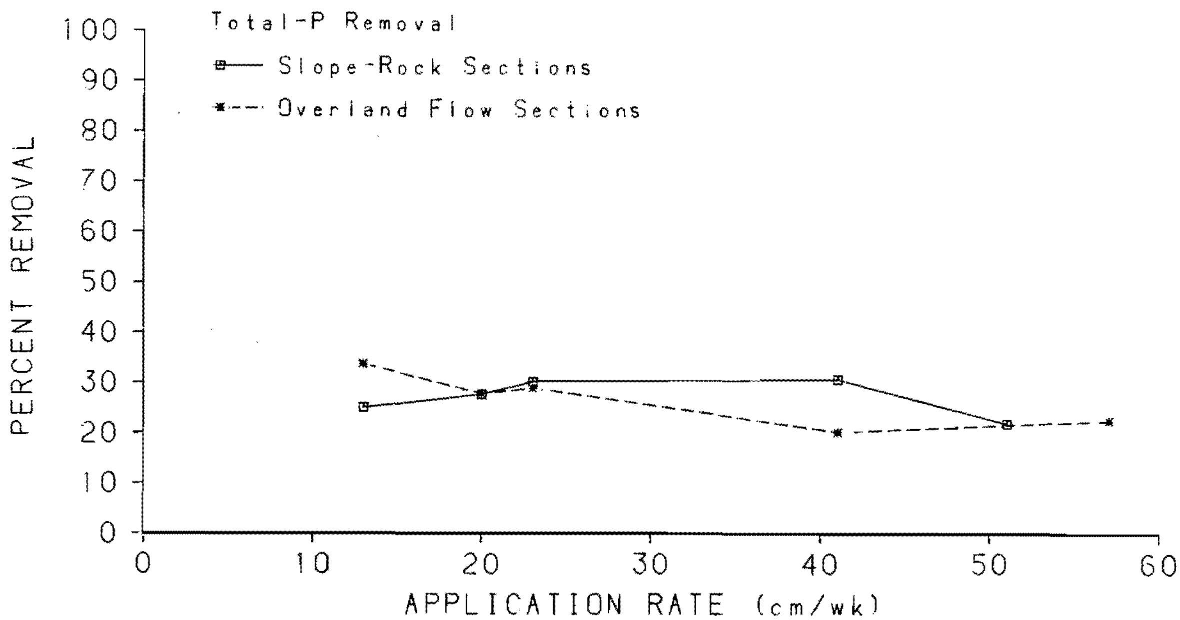


Figure 28. Total phosphorus concentration reduction versus hydraulic application rate.

Table 27. Linear regression equations for orthophosphate increase as a function of flow rate.

	Y = Percent Orthophosphate Removal[a]	
	X = Application Rate (cm/wk)	
Slope-Rock	Y = - 0.30 X - 24.45	r^2 = 0.38
Overland Flow	Y = - 0.54 X - 16.78	r^2 = 0.73
Both	Y = - 0.43 X - 20.24	r^2 = 0.57
	X = Effluent Flow Rate (cm/wk)	
Slope-Rock	Y = - 0.75 X - 25.51	r^2 = 0.85
Overland Flow	Y = - 0.06 X - 19.39	r^2 = 0.69
Both	Y = - 0.88 X - 23.00	r^2 = 0.70

[a]Negative removal indicates increase.

inorganic. Phosphorus may have changed from organic to inorganic forms by mineralization of complex organic phosphorus compounds which release orthophosphates (Alexander 1967). Inorganic phosphorus in the runoff may also be attributed to the hydrolysis and reduction of aluminum and iron phosphates in the soil. Phosphorus removal and leaching can occur simultaneously, causing little or no net change in total phosphorus to be observed.

Weekly variations of total and orthophosphate phosphorus influent and effluent concentrations are presented in Figures 29 and 30. For the first year, poor linear correlations (0.01 to 0.1) were seen between influent and effluent total phosphorus concentrations. The linear correlation of influent orthophosphate to effluent orthophosphate yielded correlation coefficients ranging from 0.43 to 0.67 (Zirschky 1980). In the second year of operation, except for the harvesting periods during weeks 7 and 15, the total phosphorus in the effluent was a direct function of influent total phosphorus concentration. Regressions of influent versus effluent concentrations gave correlations equal to 0.53 to 0.71 (Wightman 1982). Similar findings were reported by Kemp et al. (1978).

The mass removal efficiencies for total phosphorus (Figures 31 and 32) ranged from 35 percent to 72 percent and were all higher than the respective concentration removal efficiencies for each plot (Table 26). The slope-rock sections achieved greater phosphorus removal than did the overland flow slopes. The larger water losses due to infiltration and percolation occurring on the slope-rock sections decreased the mass of phosphorus in the runoff. The mass of orthophosphate increased in the effluent from the 57 cm/wk slope rock plot and all the overland flow plots, except on the 13 cm/wk plot. Thus orthophosphate was being leached from the soil on these plots.

Mean specific mass removal rates are given in Table 27. The slope rock plots achieved higher removal rates of total phosphorus than the corresponding overland flow plots, associated with larger water losses from these plots (Table 17). Higher removal rates were also associated with higher loading rates.

Intermediate samples

Mean influent and intermediate orthophosphate concentrations are given in Table 28. Total phosphorus was not analyzed for the 9 m samples. The orthophosphorus levels increased in the first 9 m on all the plots and continued to increase on most plots (Figures 33 and 34). The flow characteristics on these slopes may have been such that the lower portions of the slopes were saturated, resulting in phosphorus solubilizing all the way down the slopes. Each pair of treatments were statistically similar except for the 13 cm/wk plots and the 41 cm/wk plots. Figure 35 shows the weekly variations of the intermediate orthophosphate concentrations. Orthophosphate increased after harvesting.

System age

System age did seem to affect effluent orthophosphate and total phosphorus concentrations during the first year. Effluent total phosphorus concentrations stabilized by the fifth week. However, during the second year, system age did not affect the treatment efficiency of the test plots. Two years of wastewater application did not exhaust the capacity of the soil to remove phosphorus. Extractable phosphorus increased in the soils.

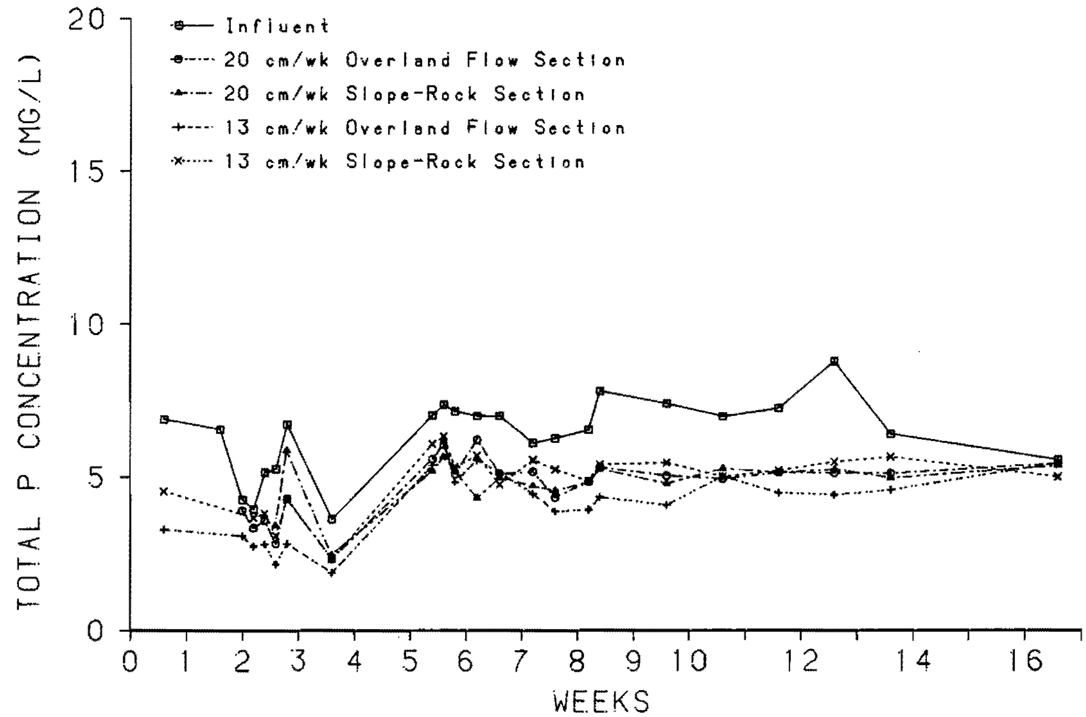


Figure 29a. Influent and effluent total phosphorus concentrations for the first year of operation (after Zirschky 1980).

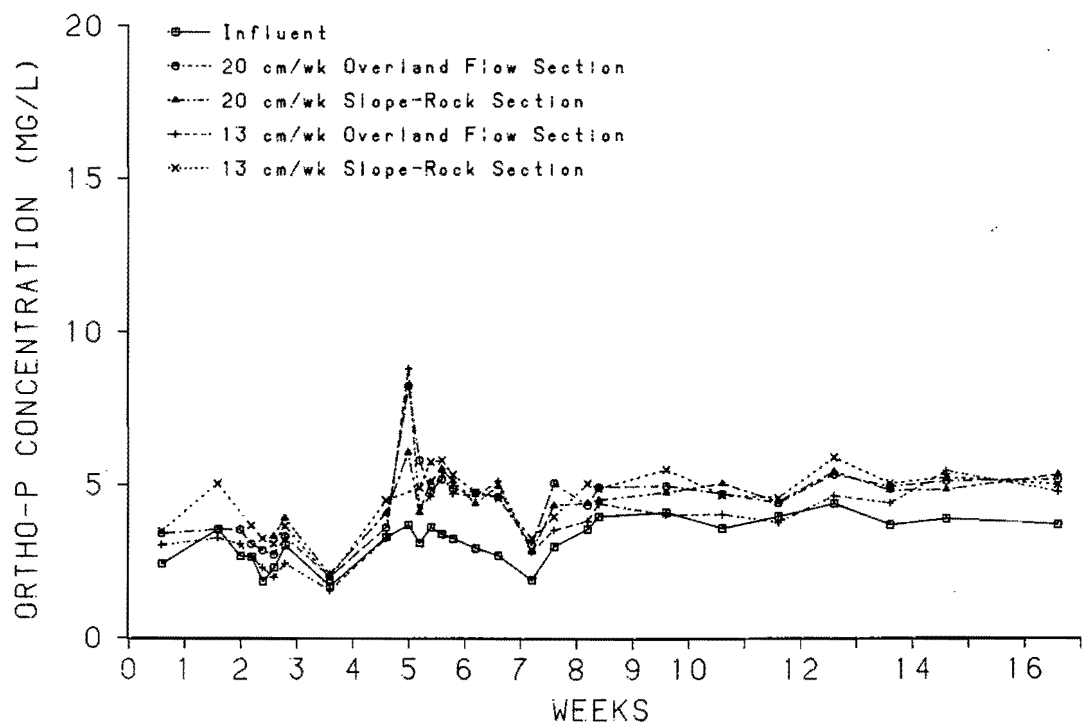


Figure 29b. Influent and effluent orthophosphate phosphorus concentrations for the first year of operation (after Zirschky 1980).

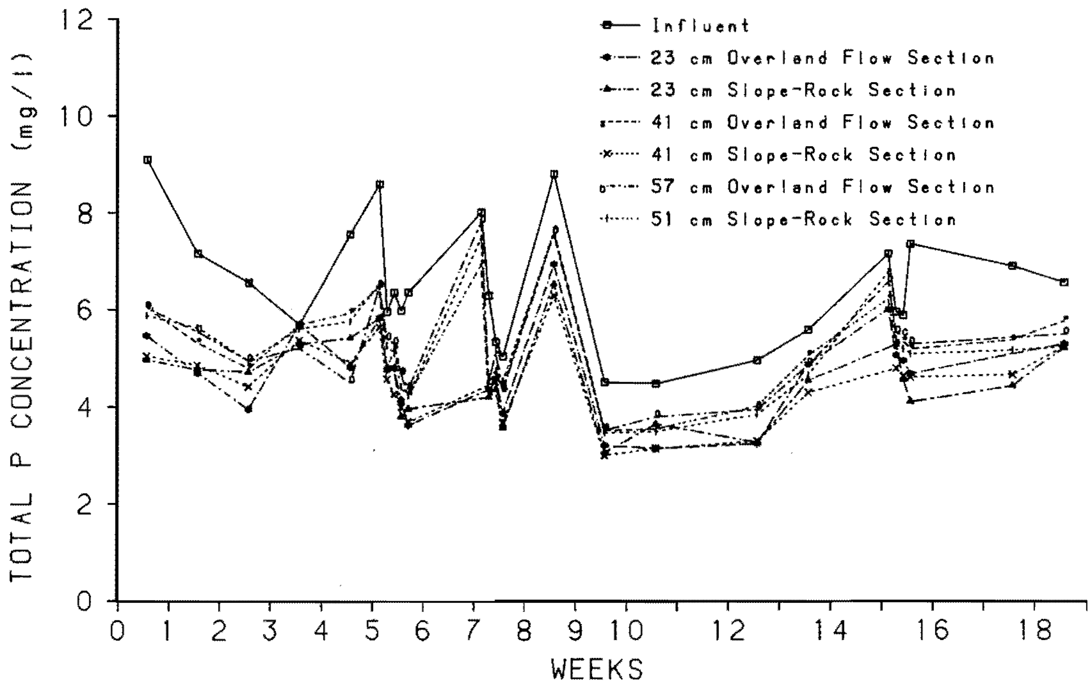


Figure 30a. Influent and effluent total phosphorus concentrations during the second year of operation.

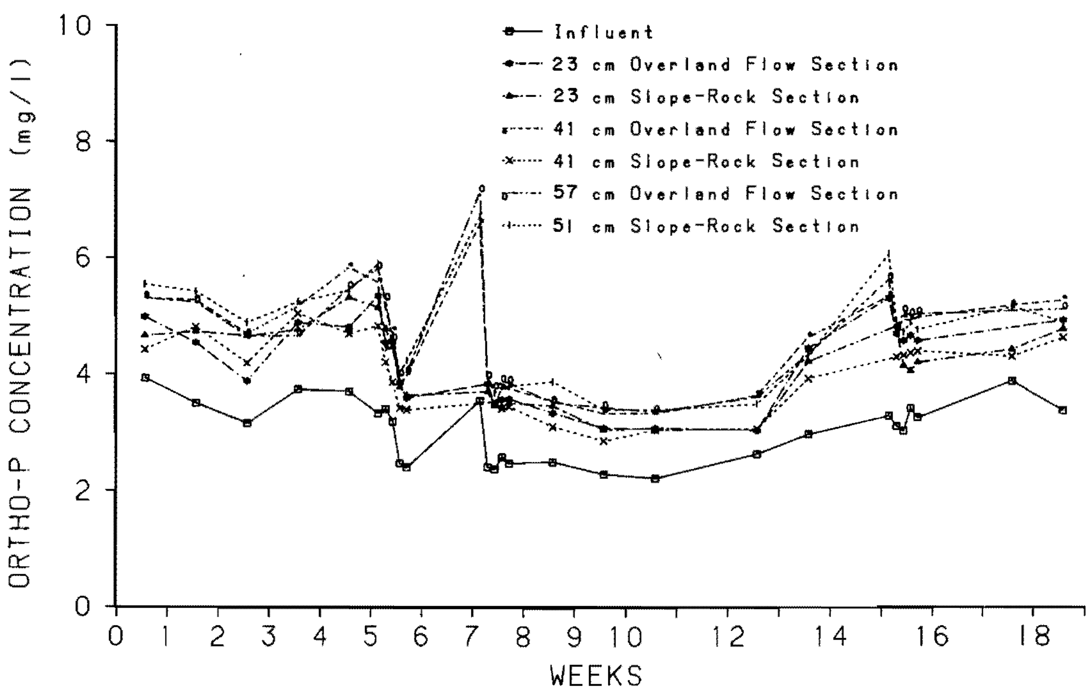


Figure 30b. Influent and effluent orthophosphorus concentrations during the second year of operation.

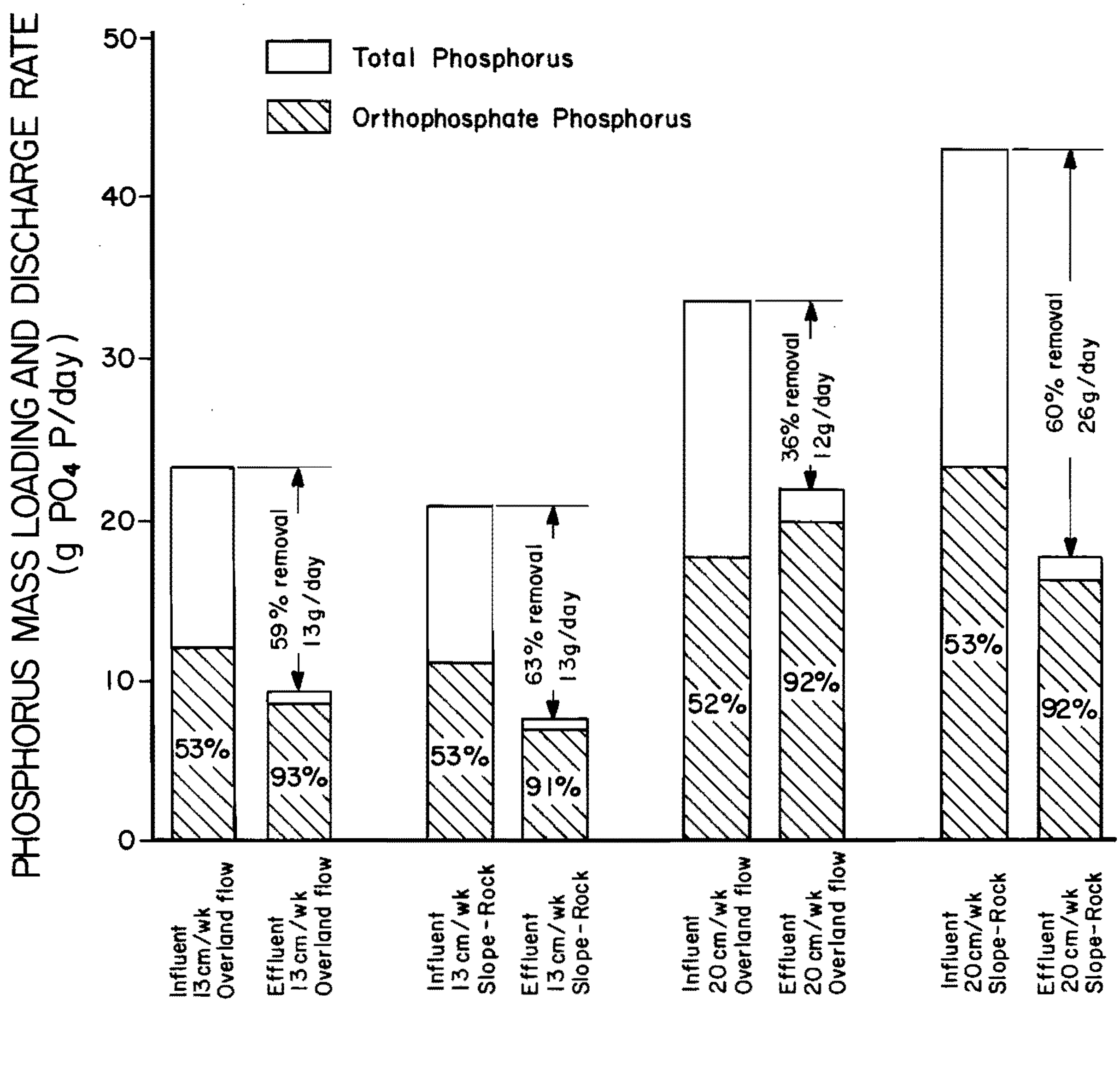


Figure 31. Comparison of mean phosphorus mass loadings and discharge rates for the first year of operation.

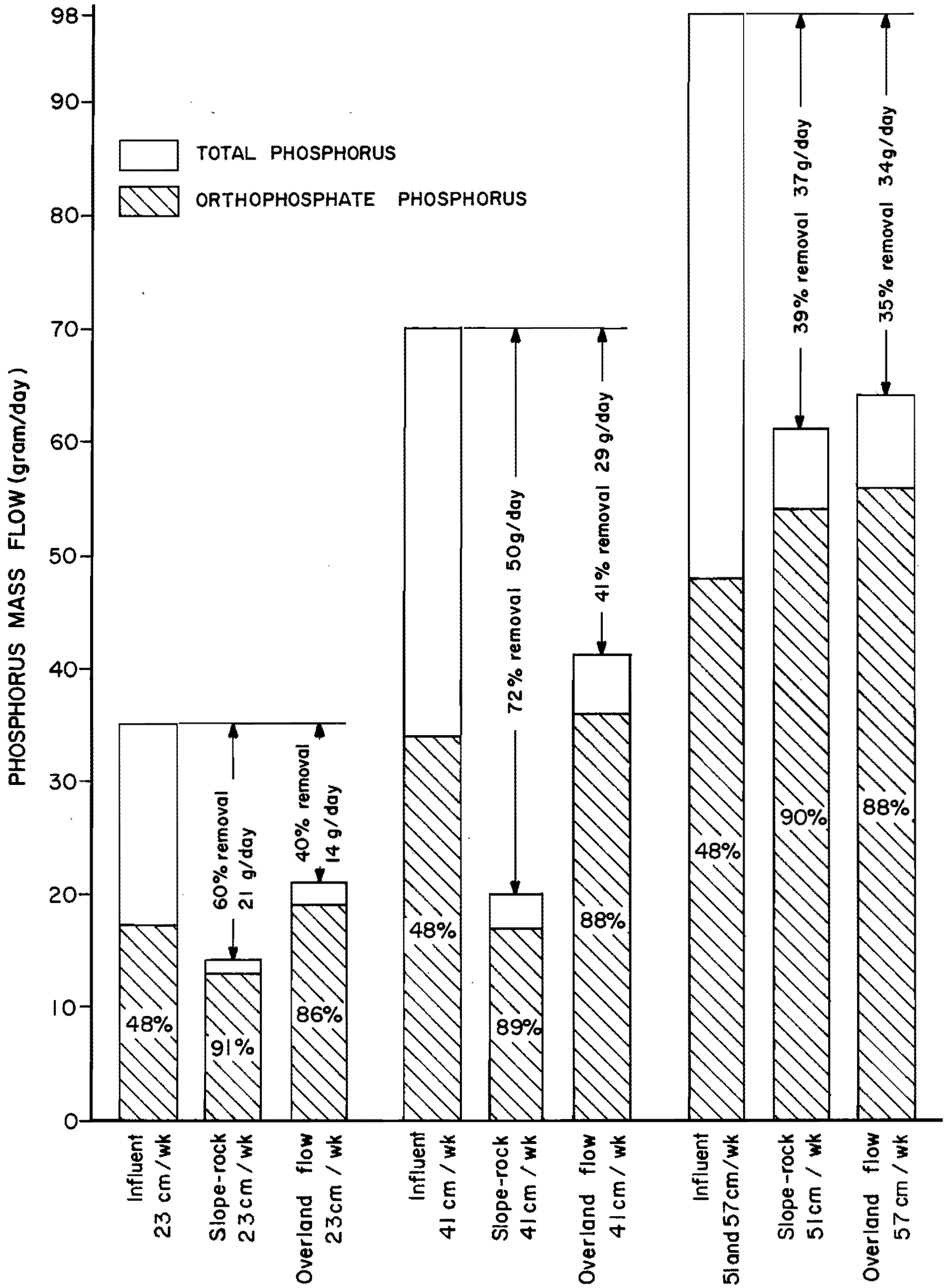


Figure 32. Comparison of mean phosphorus mass loadings and discharge rates for the second year of operation.

Table 28. Mean influent and intermediate phosphorus concentrations.

Year/Loading Rate/ Treatment	Influent Orthophosphate Mean ($mg/l\ PO_4$-P)	Influent Orthophosphate Standard Deviation ($mg/l\ PO_4$-P)	Orthophosphate at 9 m Sampling Point Mean[1] ($mg/l\ PO_4$-P)	Orthophosphate at 9 m Sampling Point Standard Deviation ($mg/l\ PO_4$-P)	Removal (%)
Year 1	3.64	0.72			
13 cm/wk					
Overland Flow			5.01[ab]	0.63	
Slope-Rock			5.42[b]	0.59	
20 cm/wk					
Overland Flow			4.58[a]	0.45	
Slope-Rock			4.86[ab]	0.34	
Year 2	3.25				
23 cm/wk					
Overland Flow			4.37[C]		
Slope-Rock			4.61[C]		
41 cm/wk					
Overland Flow			4.36[B]		
Slope-Rock			4.18[A]		
51 cm/wk					
Overland Flow			4.35[B]		
Slope-Rock			4.30[AB]		

[1]Means followed by the same letter are not significantly different at the 0.05 level. Lower case letters are used to compare means within the first year while capital letters are used to compare means within the second year.

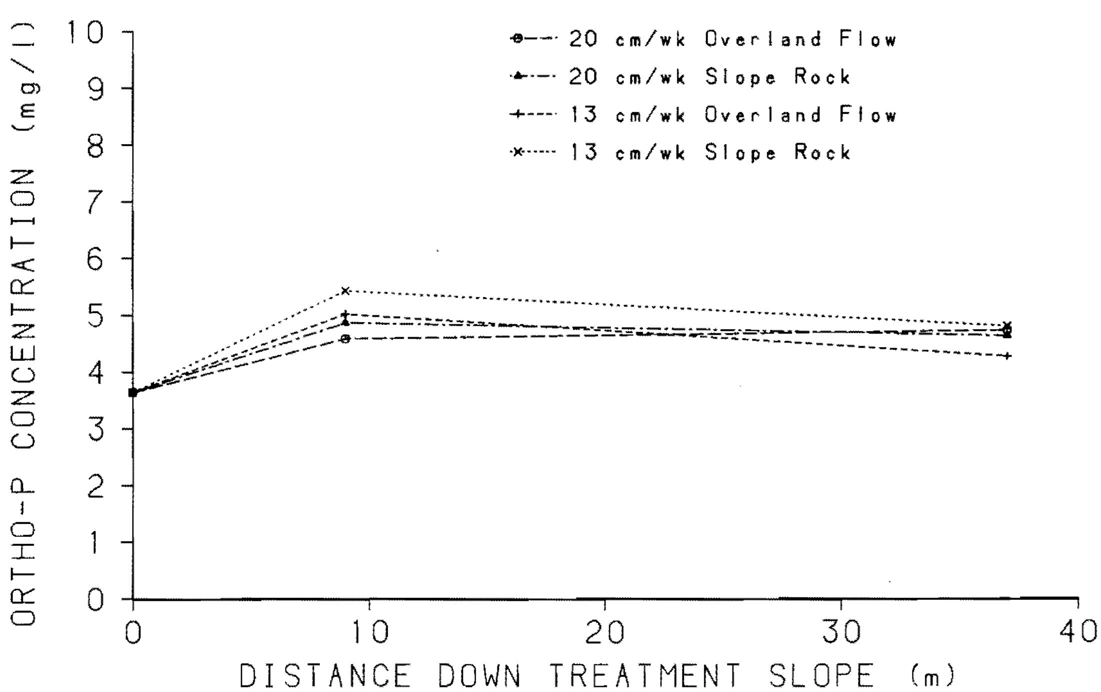


Figure 33. Orthophosphorus concentration versus distance down the treatment slope for the first year of operation.

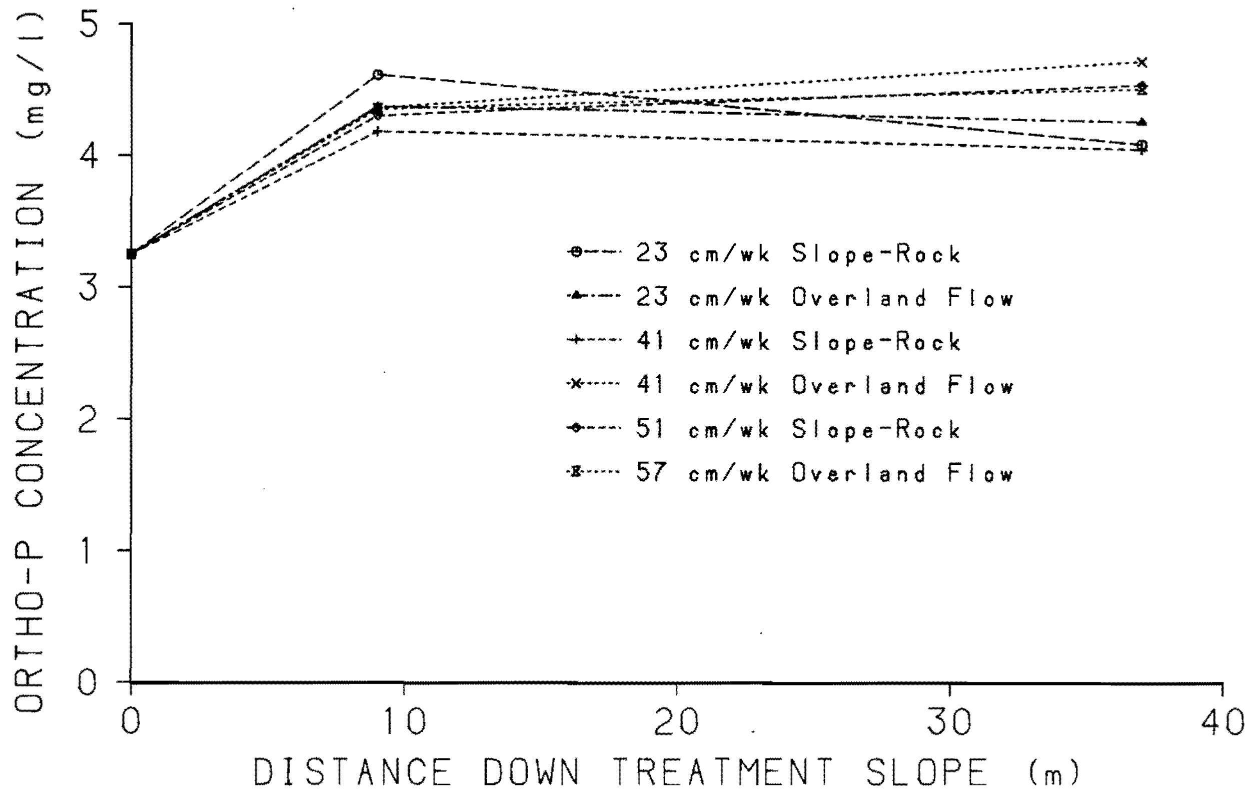


Figure 34. Orthophosphorus concentration versus distance down the treatment slope for the second year of operation.

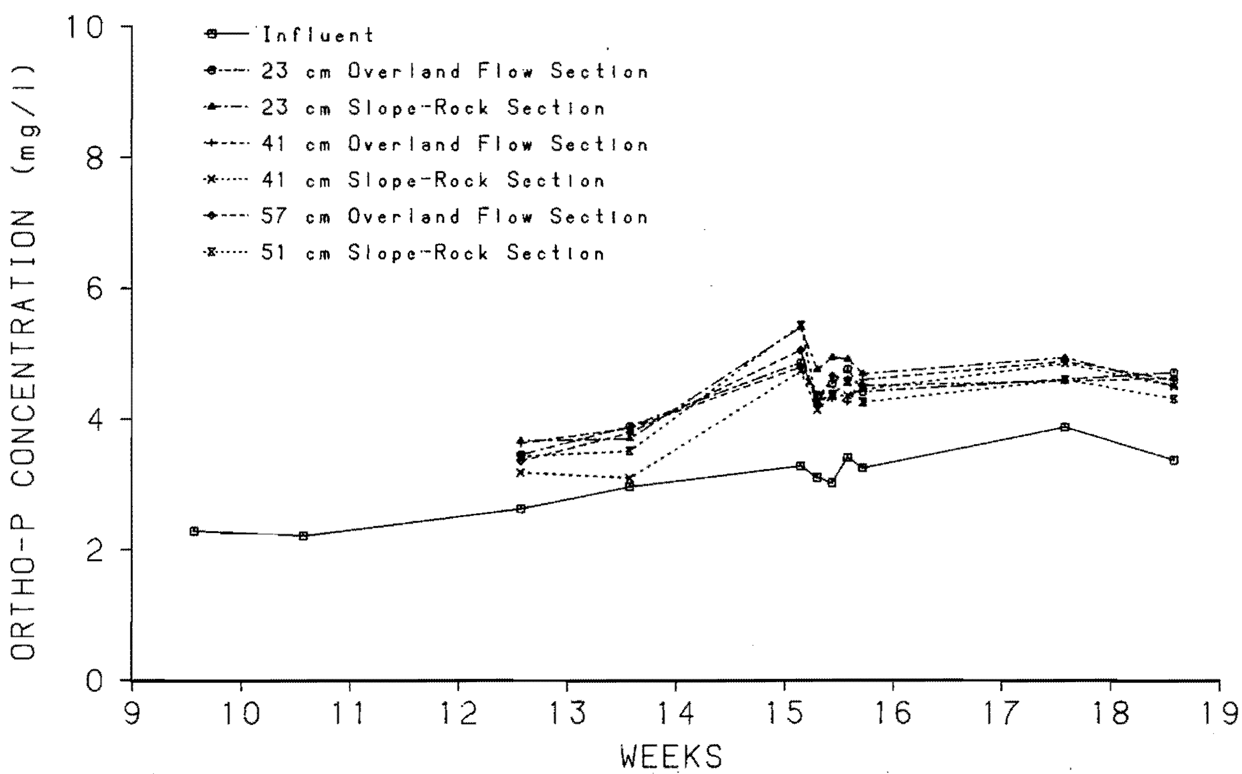


Figure 35. Intermediate orthophosphate phosphorus variations during the second year of operation.

Two-year summary

None of the treatment slopes effectively reduced total phosphorus concentrations in the wastewater. Hydraulic application rate also had no effect. Inefficient phosphorus removal from overland flow treatment systems is due to limited contact between wastewater and soil.

Though total phosphorus levels were decreased by overland flow treatment, orthophosphate concentrations increased. Total phosphorus concentrations in the runoff were 88 to 93 percent inorganic while the applied wastewaters contained approximately 50 percent organic phosphorus and 50 percent inorganic. Mass removals of phosphorus were generally greater than removal efficiencies based on concentration.

Nitrogen

A summary of nitrogen removal and transformation data is presented in Table 29.

Ammonia-nitrogen

The highest ammonia-nitrogen removal (85-93 percent) was accomplished by the 13 cm/wk plots with an average effluent value of 1 to 2 mg/l, while the lowest removal (36-44 percent) was recorded on the 57 cm/wk plots, with average effluent values of about 10 to 12 mg/l. Only on the plots hydraulically loaded at 13 cm/wk were the effluent ammonia-nitrogen concentrations within the range (< 2 mg/l) reported by EPA (1977). Generally, lower wastewater flows, longer detention times, and subsequent thinner surface films associated with lower hydraulic loading rates achieved the best effluent quality. Similar results were obtained by Thomas (1978) and Hinrichs et al. (1980). Thomas (1978) and Hinrichs et al. (1980) concluded that increasing the flow rate decreases ammonia removal by limiting nitrification. Kemp et al. (1978) reported that the thicker water film and faster flow rate associated with greater flow resulted in inefficient diffusion of nitrate into the soil. Bouwer and Chaney (1974), Hoeppel et al. (1974), EPA (1977), and Kemp et al. (1978) suggested that the efficiency of nitrogen removal depends on time of contact between water and soil surface.

Treatment performance was not affected by the addition of the gravel layer. Mean effluent values were not statistically different for pairs of treatments, except for the 41 cm/wk plots, which had different flow characteristics. The 41 cm/wk slope-rock section lost a portion of the applied wastewater off the sides of the plot, resulting in smaller flows and a longer detention time. Channeling observed on the 41 cm/wk overland flow plot resulted in higher velocities and a shorter detention time.

Weekly variations of influent and effluent ammonia-nitrogen are illustrated in Figures 36 and 37. During the first year, the high effluent ammonia concentrations which occurred during week 5 was related to harvesting. During the second year, after the fifth week (acclimation during spring start-up), higher effluent ammonia levels were observed from the 41 cm/wk overland flow plot and the 57 cm/wk plots. The effluents from these slopes appeared to fluctuate with influent concentrations. As during year 1, concentrations also increased after periods of harvesting.

Overall, ammonia-nitrogen and total nitrogen removal efficiencies decreased as flow rate increased. Figure 38 shows regression analyses of ammonia removal versus application rate over both years and all slopes. Table 30 gives the regression analyses of ammonia removal versus application rate, effluent flow rate, and mean and mode detention times. These results agree with the results obtained by Kemp et al. (1978) (8 to 23 cm/wk), Thomas (1978) (15 to 23 cm/wk) and Hoeppel et al. (1974) (3 to 6 cm/wk), that increasing the flow rate decreases ammonia removal by limiting

Table 29. Mean influent and effluent nitrogen concentrations and removal efficiencies for both years of operation.

Year 1 Parameter	n	Influent Mean	Influent Standard Deviation	13 cm/wk Overland Flow Mean	13 cm/wk Overland Flow Standard Deviation	13 cm/wk Slope Rock Mean	13 cm/wk Slope Rock Standard Deviation	20 cm/wk Overland Flow Mean	20 cm/wk Overland Flow Standard Deviation	20 cm/wk Slope Rock Mean	20 cm/wk Slope Rock Standard Deviation
Ammonia Nitrogen (mg/l)[a]	13	14.7	4.6	0.99^{a}	1.1	2.14^{ab}	2.0	4.64^{c}	1.9	3.39^{bc}	1.9
Removal (%)				93		85		68		77	
Nitrate Nitrogen (mg/l)[a]	9	0.12	0.13	0.41^{a}	0.32	1.3^{a}	0.81	1.6^{a}	0.96	1.7^{a}	1.1
Nitrite Nitrogen (mg/l)[a]	9	0.21	0.22	0.47^{a}	0.48	0.83^{ab}	0.63	1.2^{ab}	0.60	1.7^{b}	0.27
Nitrate plus Nitrite Nitrogen (mg/l)[a]	10	0.39	0.38	0.88^{a}	0.67	2.2^{a}	1.1	2.85^{a}	1.2	3.32^{a}	1.0
Total Kjeldahl Nitrogen (mg/l)	6	22	4.6	4	1.3	6	2.1	9	0.96	8	2.5
Removal (%)				82		73		55		64	
Total Nitrogen (mg/l)[b]	-	22.39	-	4.88	-	8.2	-	11.85	-	11.32	-
Removal (%)				78		63		47		49	

Year 2 Parameter	n	Influent Mean	Influent Standard Deviation	23 cm/wk Overland Flow Mean	23 cm/wk Overland Flow Standard Deviation	23 cm/wk Slope Rock Mean	23 cm/wk Slope Rock Standard Deviation	41 cm/wk Overland Flow Mean	41 cm/wk Overland Flow Standard Deviation	41 cm/wk Slope Rock Mean	41 cm/wk Slope Rock Standard Deviation	57 cm/wk Overland Flow Mean	57 cm/wk Overland Flow Standard Deviation	57 cm/wk Slope Rock Mean	57 cm/wk Slope Rock Standard Deviation
Ammonia Nitrogen (mg/l)[a]	14-15	17.6	5.8	5.2^{a}	2.8	3.6^{a}	2.3	10.8^{b}	3.0	5.3^{a}	2.9	11.7^{b}	3.5	9.9^{b}	3.1
Removal (%)				70				39		70		36		44	
Nitrite Nitrogen (mg/l)[a]	13-14	0.18	0.04	0.16^{a}	0.09	0.12^{a}	0.10	0.37^{bc}	0.13	0.44^{c}	0.29	0.16^{a}	0.07	0.20^{ab}	0.14
Nitrate Nitrogen (mg/l)[a]	13-14	1.28	0.27	3.45^{a}	2.92	4.43^{a}	3.29	2.95^{a}	2.34	3.81^{a}	3.90	2.55^{a}	2.19	3.37^{a}	2.21
Total Kjeldahl Nitrogen (mg/l)[a]	8	20.0	5	7^{abc}	6	4^{a}	3	11^{c}	4	7^{ab}	4	11^{c}	4	10^{bc}	3
Removal (%)				65		80		45		65		45		50	
Total Nitrogen (mg/l)[c]	-	21.46	-	10.61	-	8.55	-	14.32	-	11.25	-	13.71	-	13.57	-
Removal (%)				51		60		33		48		36		37	

[a]Means followed by the same letter are not significantly different at the 0.05 level.

[b]Total nitrogen was calculated as the sum of (total Kjeldahl nitrogen) + (nitrate plus nitrite nitrogen).

[c]Total nitrogen was calculated as the sum of (total Kjeldahl nitrogen) + (nitrite nitrogen) + (nitrate nitrogen).

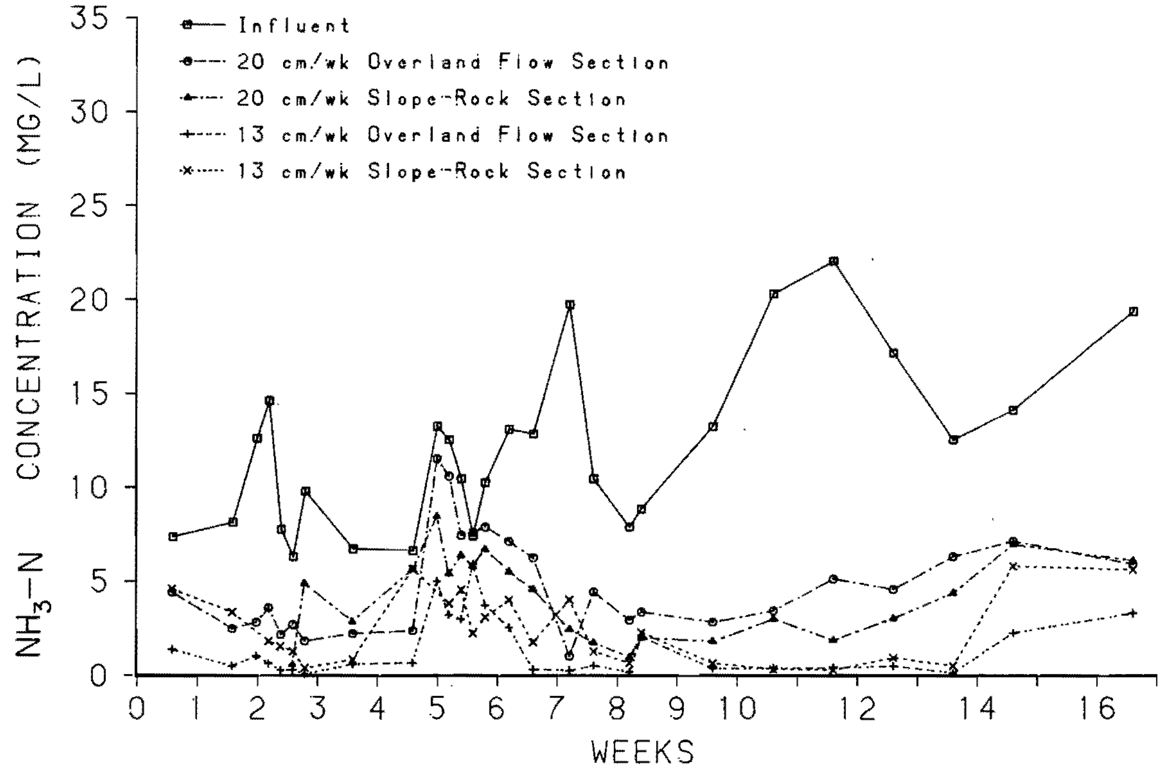


Figure 36. Influent and effluent ammonia-nitrogen concentrations during the first year of operation.

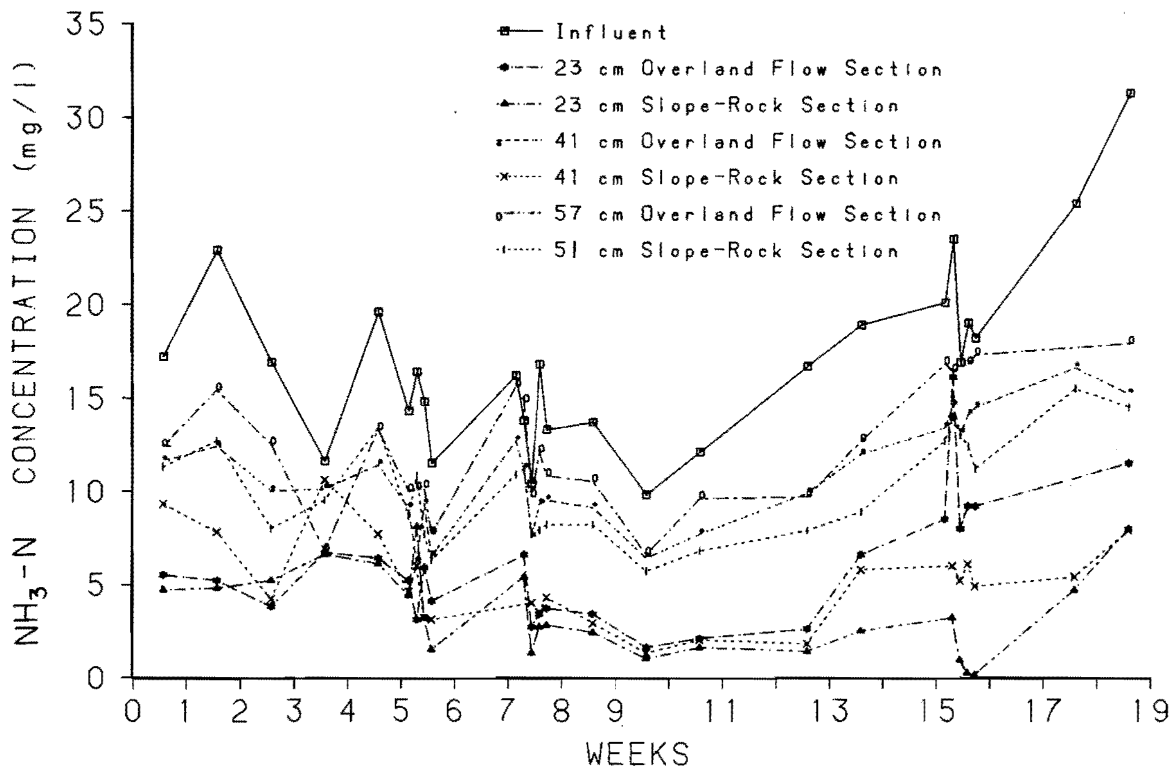


Figure 37. Influent and effluent ammonia-nitrogen concentrations during the second year of operation.

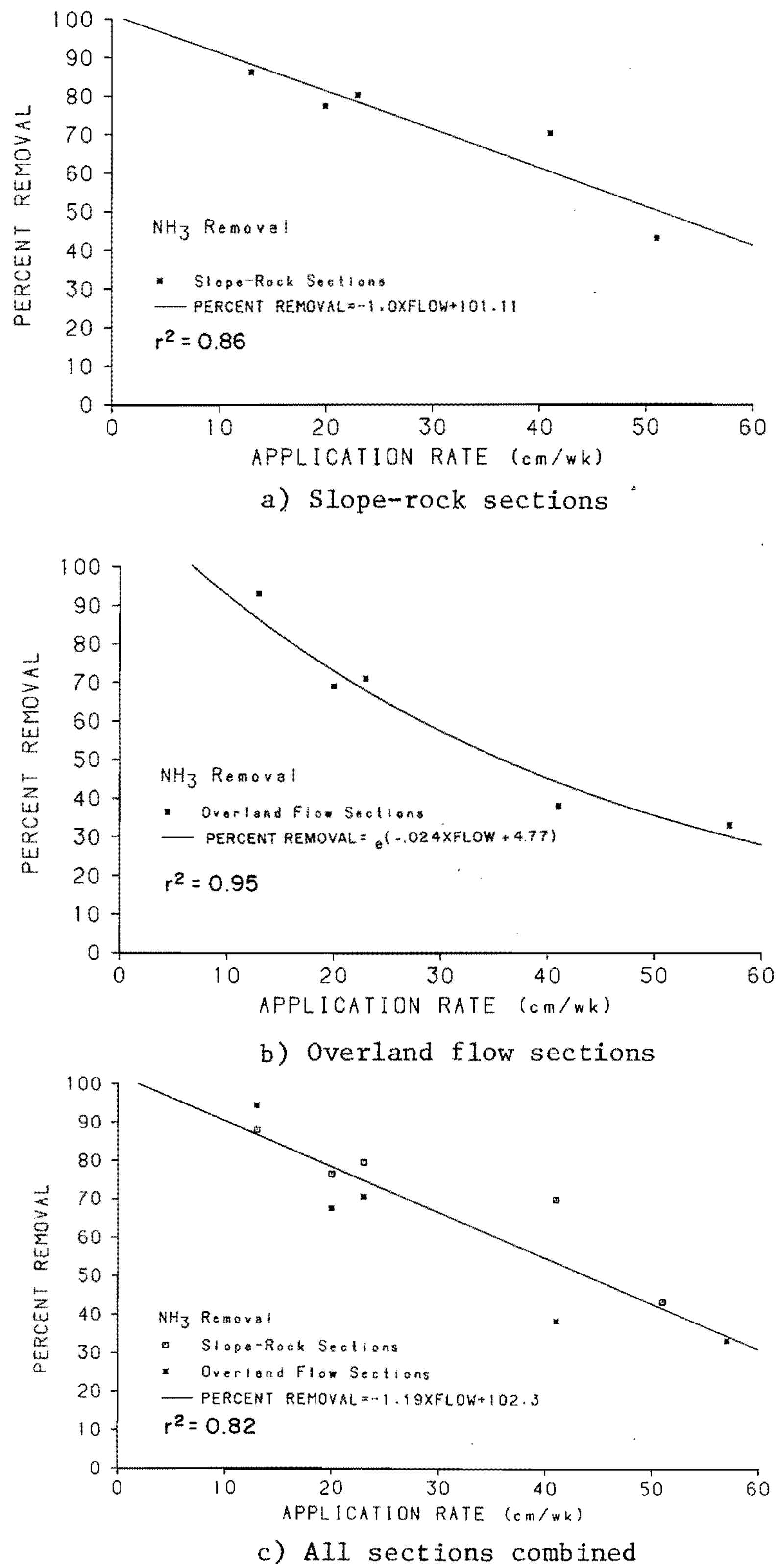


Figure 38. Ammonia removal as a function of hydraulic application rate under the experimental conditions of this study for a) slope rock sections, b) overland flow sections, and c) all sections combined.

Table 30. Ammonia removal regression analyses.

	Y = Percent Ammonia Nitrogen Removal	
	X = Application Flow Rate (cm/wk)	
Slope-Rock	$Y = -1.00 X + 101.11$	$r^2 = 0.86$
Overland Flow	$Y = -1.32 X + 101.62$	$r^2 = 0.89$
Both	$Y = -1.19 X + 102.25$	$r^2 = 0.82$
Overland Flow	$LnY = -.024 X + 4.77$	$r^2 = 0.95$
	X = Effluent Flow Rate (cm/wk)	
Slope-Rock	$Y = -1.96 X + 91.52$	$r^2 = 0.98$
Overland Flow	$Y = -2.65 X + 95.88$	$r^2 = 0.86$
Both	$Y = -2.36 X + 93.78$	$r^2 = 0.88$
	X = Mean Detention Time (min)	
Slope-Rock	$Y = 0.87 X - 45.66$	$r^2 = 0.77$
Overland Flow	$Y = 0.83 X - 35.43$	$r^2 = 0.82$
Both	$Y = 0.80 X - 34.40$	$r^2 = 0.81$
	X = Mode Detention Time (min)	
Slope-Rock	$Y = 0.51 X + 24.22$	$r^2 = 0.77$
Overland Flow	$Y = 0.64 X + 7.06$	$r^2 = 0.92$
Both	$Y = 0.60 X + 12.78$	$r^2 = 0.85$

nitrification. Thicker water films and faster flows limit oxygen diffusion and soil water contact for NH_4^+ adsorption and subsequent nitrification. Also, less time is available for plant uptake. Therefore, shallow flows and long detention times are required for significant ammonia removal.

Nitrite plus nitrate nitrogen

Nitrification occurred on all treatment fields, as seen by the increase in nitrite- and nitrate-nitrogen, compared to influent values (Figures 39-42).

In general, nitrite-nitrogen accumulations never exceeded 1.0 mg/l. However, effluent nitrite concentrations the first year exceeded 1 mg/l NO_2-N on both slopes loaded at 20 cm/wk. Others have rarely found effluent nitrite-nitrogen levels over 1 mg/l from overland flow systems (Culp undated). Morrill and Dawson (1967) found that nitrite accumulated at pH values greater than 7.3 with further oxidation to nitrate occurring at a slow rate. Toxic accumulation of nitrite can occur at pH values about 9.5 (Mitchell 1974). Nitrite oxidation is inhibited by free ammonia in liquid systems when the pH is alkaline. In soils, however, adsorption of ammonium prevents this inhibition from becoming a practical consideration in most circumstances. Under the operating conditions of this study (mean influent of 14.7 mg/l NH_3-N), with temperatures of 20°C and a wastewater pH of 7.7, a free ammonia concentration of 0.3 mg/l could have been present on the upper portions of the slopes. Nitrite oxidizing bacteria (Nitrobacter) are inhibited by concentrations of 0.1 to 1 mg/l free ammonia (Anthonisen et al. 1976). The Nitrobacter could have been inhibited, resulting in the observed accumulations of nitrite.

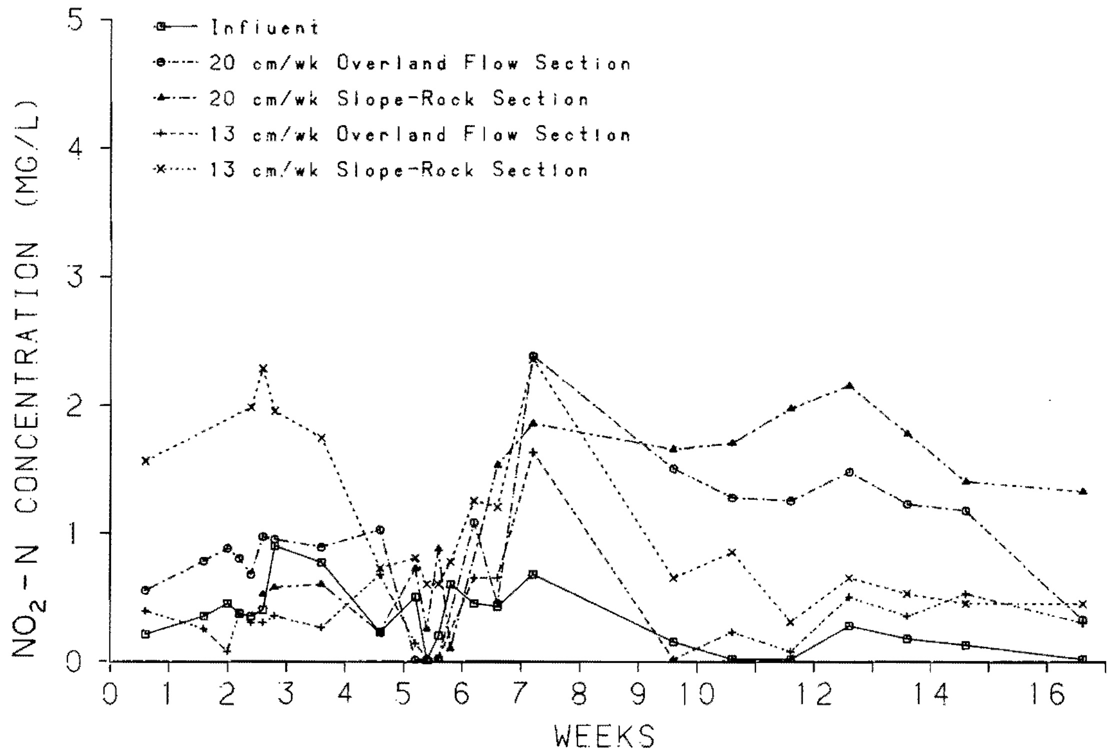


Figure 39. Influent and effluent nitrite-nitrogen concentrations during the first year of operation.

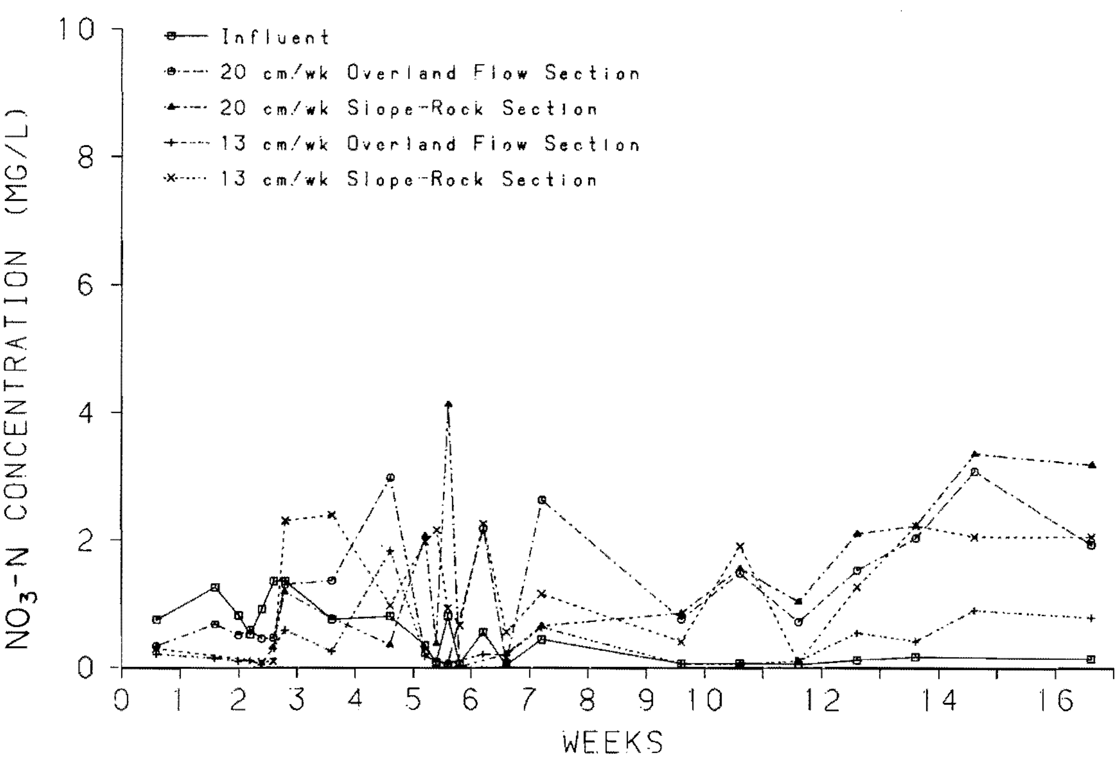


Figure 40. Influent and effluent nitrate-nitrogen concentrations during the first year of operation.

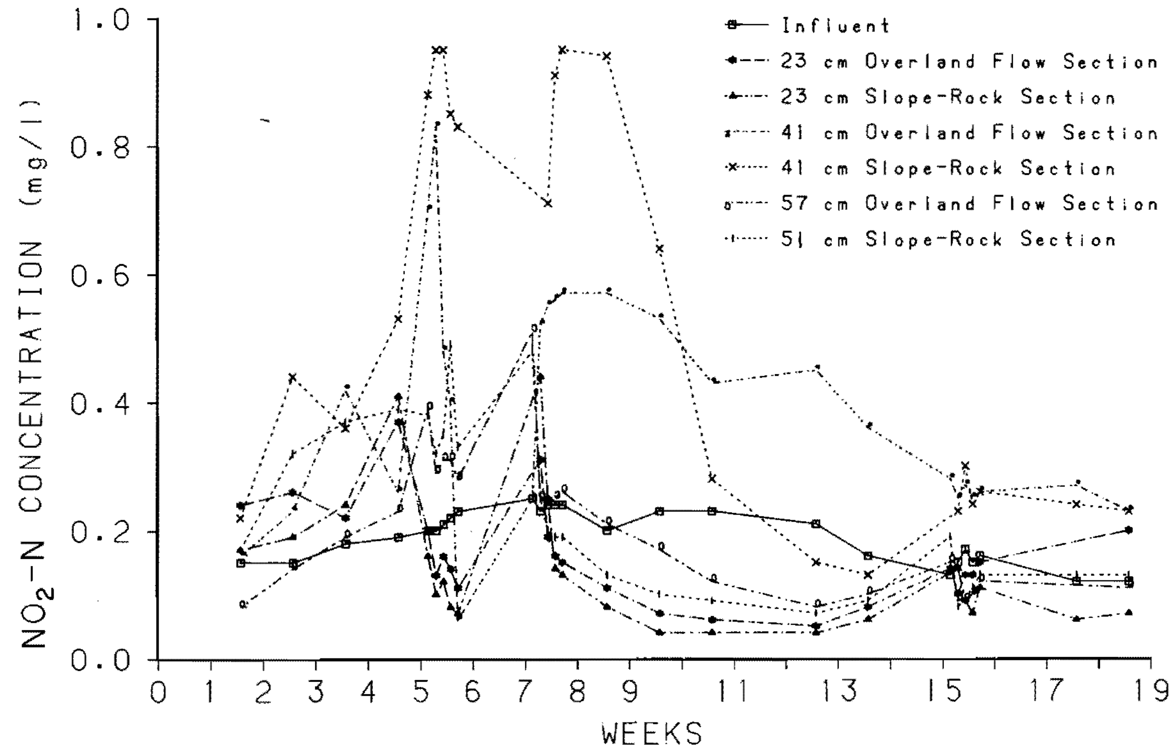


Figure 41. Influent and effluent nitrite-nitrogen concentrations during the second year of operation.

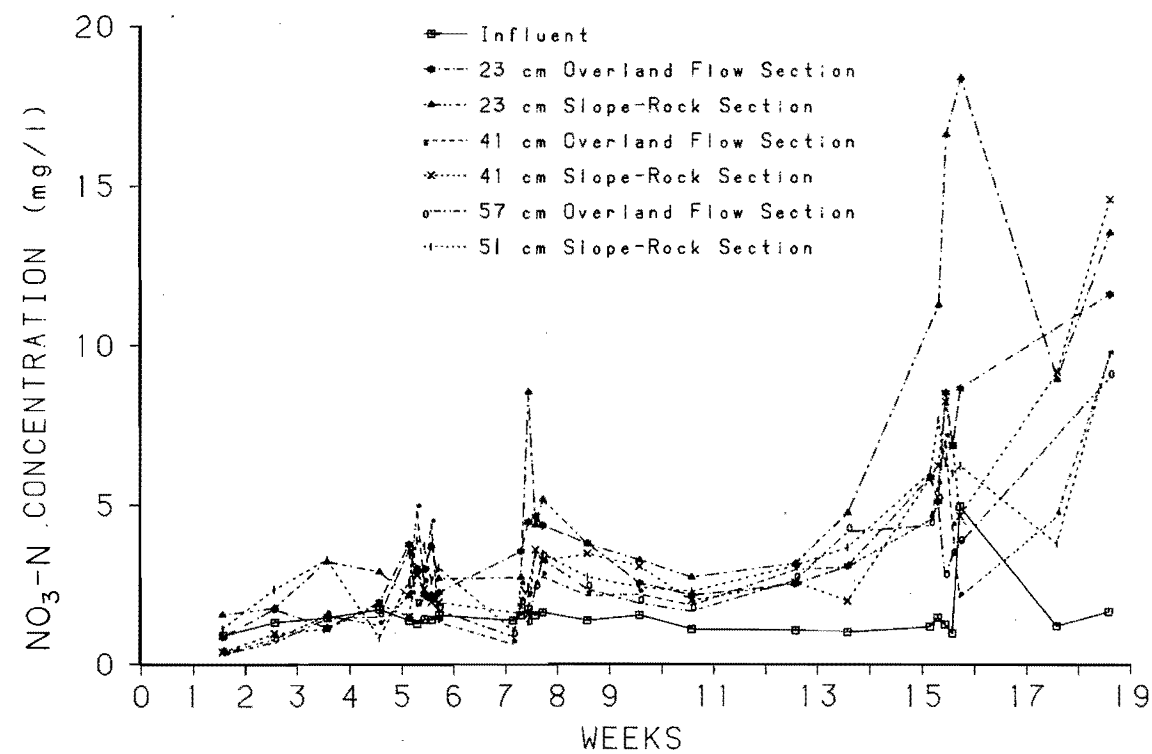


Figure 42. Influent and effluent nitrate-nitrogen concentrations during the second year of operation.

Nitrite-nitrogen also increased during spring start-up (approximately 4 weeks) and following harvesting during the second year of operation (Figure 41). This transient nitrite accumulation in the first 4 weeks was probably due to system acclimation, species proliferation, and saturation of the soil by spring rains (oxygen transport is limited in saturated soils). A lag period of about 4 weeks also occurred prior to production of nitrates, particularly in effluent from the conventional overland flow slopes (Figure 42). After Nitrobacter are established, nitrite bio-oxidation usually occurs at a rapid rate in soils of pH 7.8 (Morrill and Dawson 1967, Engler et al. 1976). Nitrite can also be readily taken up by plants (Ellis 1978).

Except for the 41 cm/wk plots and the 57 cm/wk slope-rock plot (Table 29), no significant differences between mean effluent nitrite-nitrogen concentrations could be attributed to gravel addition or to increased loading rate. The divergent nitrite pattern exhibited by the 41 cm/wk plots may have been due to the nature of the soil used to check water losses off the fringes of the experimental field. The highly organic soil placed on the periphery of both 41 cm/wk plots may have inhibited nitrification. Because the organic soil was restricted to small portions on one side of each slope, the impact was not substantial. Nor were statistical differences found between any pairs of mean effluent nitrate concentrations. During the second year, after an initial 4-week acclimation period, nitrate concentrations increased on all the plots, with the 23 cm/wk slope-rock plot exhibiting the greatest increase.

Nitrate-nitrogen forms soluble salts which do not extensively react with soils. Denitrification was limited because the nitrate ion must travel to a reducing microzone immediately below the soil surface for denitrification to occur, and most of the water traveled across the soil surface. Corey et al. (1976) found the travel velocity of the nitrate ion through soil to be one-fourth the seepage velocity for a clay loam soil due to assimilation by plants or microbes denitrification, or transport into runoff (Thomas 1972, Jenkins et al. 1978).

Nitrate-nitrogen concentrations also increased after harvesting (week 5 in the first year and weeks 7 and 15 in the second year) and toward the end of the growing season. The end-of-season increase was probably attributable to reduction in plant uptake due to senescence and colder temperatures. More oxygen can be dissolved in wastewater at lower temperatures, which is more conducive to nitrification. However, if wastewater temperatures drop below 5°C, nitrification is inhibited (Mitchell 1974, EPA 1977). During the second year of operation, the effluent temperatures did not fall below 12°C.

Total Kjeldahl nitrogen (TKN)

During the first year of operation, less TKN (which includes ammonia and organic nitrogen forms) was removed at the 20 cm/wk loading (55-64 percent) than at the 13 cm/wk (73-82 percent) (Table 29), results which are consistent with the ammonia-nitrogen results. During the second year, the 23 and 41 cm/wk slope rock and the 23 cm/wk overland flow plots achieved the highest TKN removal (65-80 percent). Short-circuiting on the 41 cm/wk overland flow slope reduced TKN removal. Increase in effluent TKN concentrations are expected at the higher loading rates because nitrification and removal of reduced forms of nitrogen are more limited at higher flow rates.

Because of time lapses greater than 1 week between sampling and analysis for some samples during the first year, effluent TKN concentrations were not compared statistically. However, for the second year, statistical analyses indicated that the gravel layer did not improve TKN removal.

Total nitrogen

When nitrate- and nitrite-nitrogen concentrations are combined with TKN concentrations, overall total nitrogen removal achieved ranged from 47 to 78 percent in the first year and 33 to 60 percent the second year. In general, lower removals were seen at higher loading rates. However, the poorest removal occurred on the 41 cm/wk overland flow slope, which was affected by severe short-circuiting and which had the shortest detention time.

Regression analyses between total nitrogen removal and flow rates are shown in Table 31. A hyperbolic model provides the best r2 (0.92) for the overland flow slopes and linear models resulted in the highest r2 (0.78) for the slope-rock sections for removal versus application rates.

The nitrogen data collected during this study were compared to data obtained from overland flow systems reviewed by Hinrichs et al. (1980) (Figure 43). Except for the Carbondale system, a hyperbolic relationship between percent nitrogen removal and hydraulic loading rate under the operating conditions of this experiment is suggested by the data. Nitrogen reductions decreased with increasing flow rate. Nitrogen removal is dependent on such factors as depth of surface flow; organic content of wastewater; texture, structure and organic content of underlying soil; duration and frequency of flooding (Carlson et al. 1974); type of cover crop (whether sod or clump forming); and management.

Mass analysis

Nitrogen mass loading and discharge rates are shown in Figures 44 and 45.

The 13 cm/wk plots discharged the least nitrogen mass (11-18 g/day) while the 41 cm/wk slope rock plot removed the greatest mass of nitrogen (185 g/day). In general, the slope-rock plots, except for the 13 cm/wk plot, gave greater mass nitrogen removals than the overland flow slopes because of their greater water losses.

Mass removal rates per unit surface area are shown in Table 32. Differences among treatments are attributed to the amount of runoff collected. Greater mass removal rates were obtained by both the treatment slopes at higher hydraulic loadings due to the greater amounts of wastewater applied.

Table 31. Total nitrogen removal regression analyses.

	Y = Percent Total Nitrogen Removal	
	X = Application Rate	
Slope-Rock	$Y = -0.58X + 69.28$	$r^2 = 0.78$
Overland Flow	$Y = -0.82X + 74.12$	$r^2 = 0.63$
Both	$Y = -0.71X + 72.17$	$r^2 = 0.65$
Overland Flow	$1/Y = -.304(1/X) + .036$	$r^2 = 0.92$

A relationship due to nitrification was sought by linear regression analysis between the relative change in ammonia mass and the relative changes in the mass of nitrite, nitrate, and the sum of nitrite plus nitrate. The results are summarized in Table 32. The highest coefficients of determination (r2 = 0.50 and 0.57) resulted from correlating the change in ammonia-nitrogen to nitrite-nitrogen on the 20 cm/wk sections. These higher correlations confirm that nitrification was somewhat inhibited on these slopes. The low coefficients of determination obtained overall suggest that nitrogen conversions may have resulted from such other processes as plant and microbial uptake, soil fixation, denitrification, and volatilization.

Intermediate samples

Figures 46 and 47 present results of the intermediate samples (9 m downslope) for ammonia-nitrogen for the 2 years of operation. During the first year, ammonia removals averaged between 5 and 20 percent in the first 9 m, with lower removals seen on the slope-rock

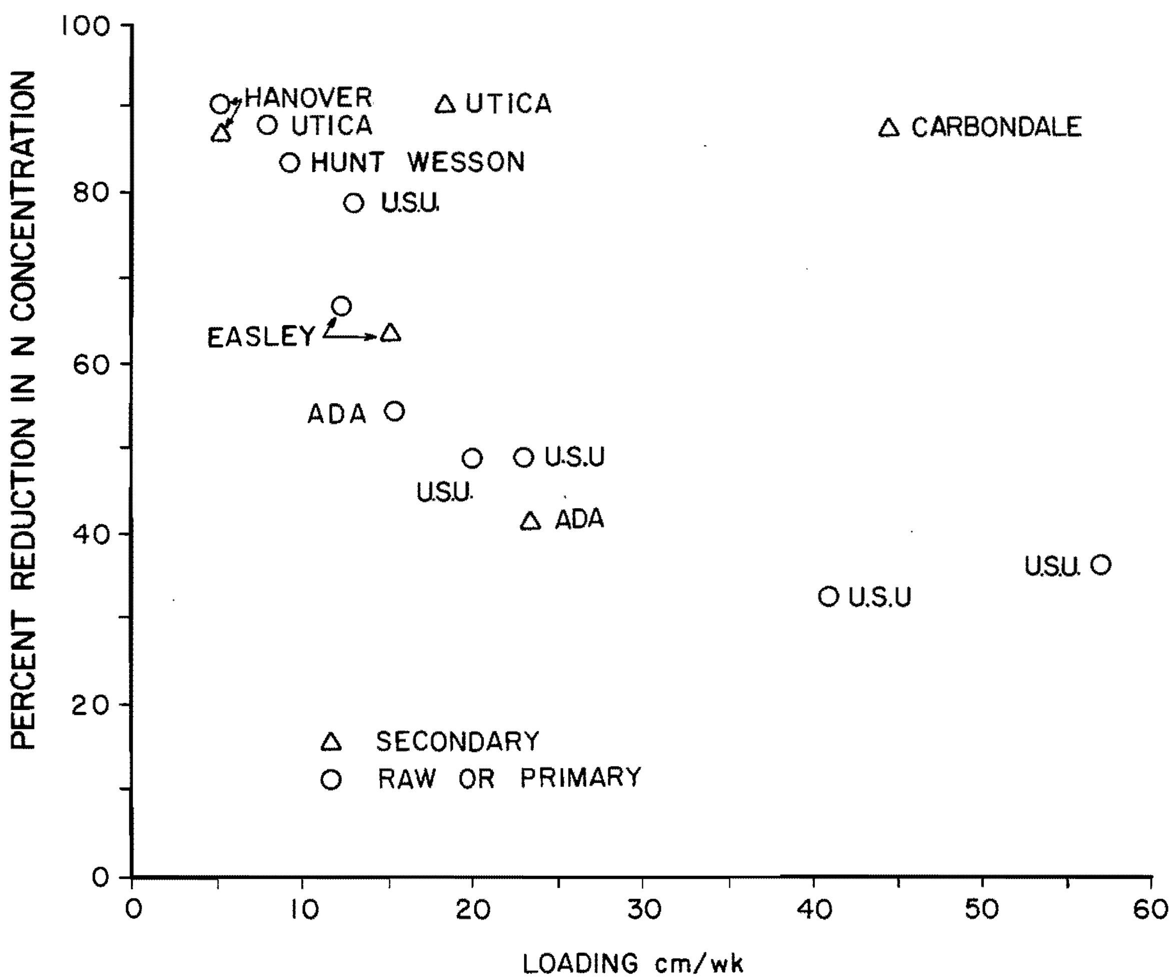


Figure 43. Relationship between percent nitrogen removal and hydraulic loading rate for overland flow wastewater treatment under various operating conditions (adapted from Hinrichs et al. 1980).

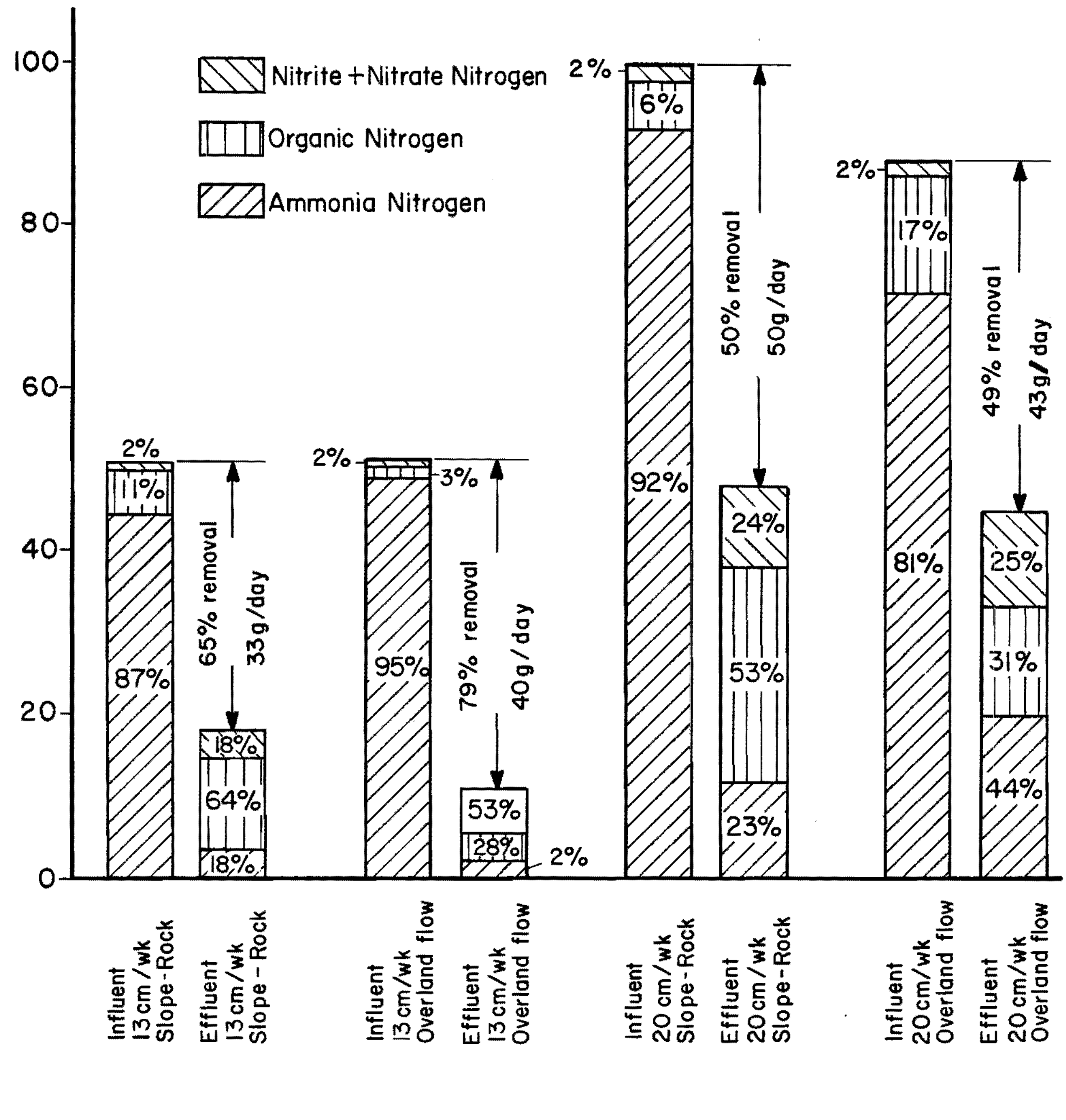


Figure 44. Nitrogen mass loading and discharge rates for the first year of operation.

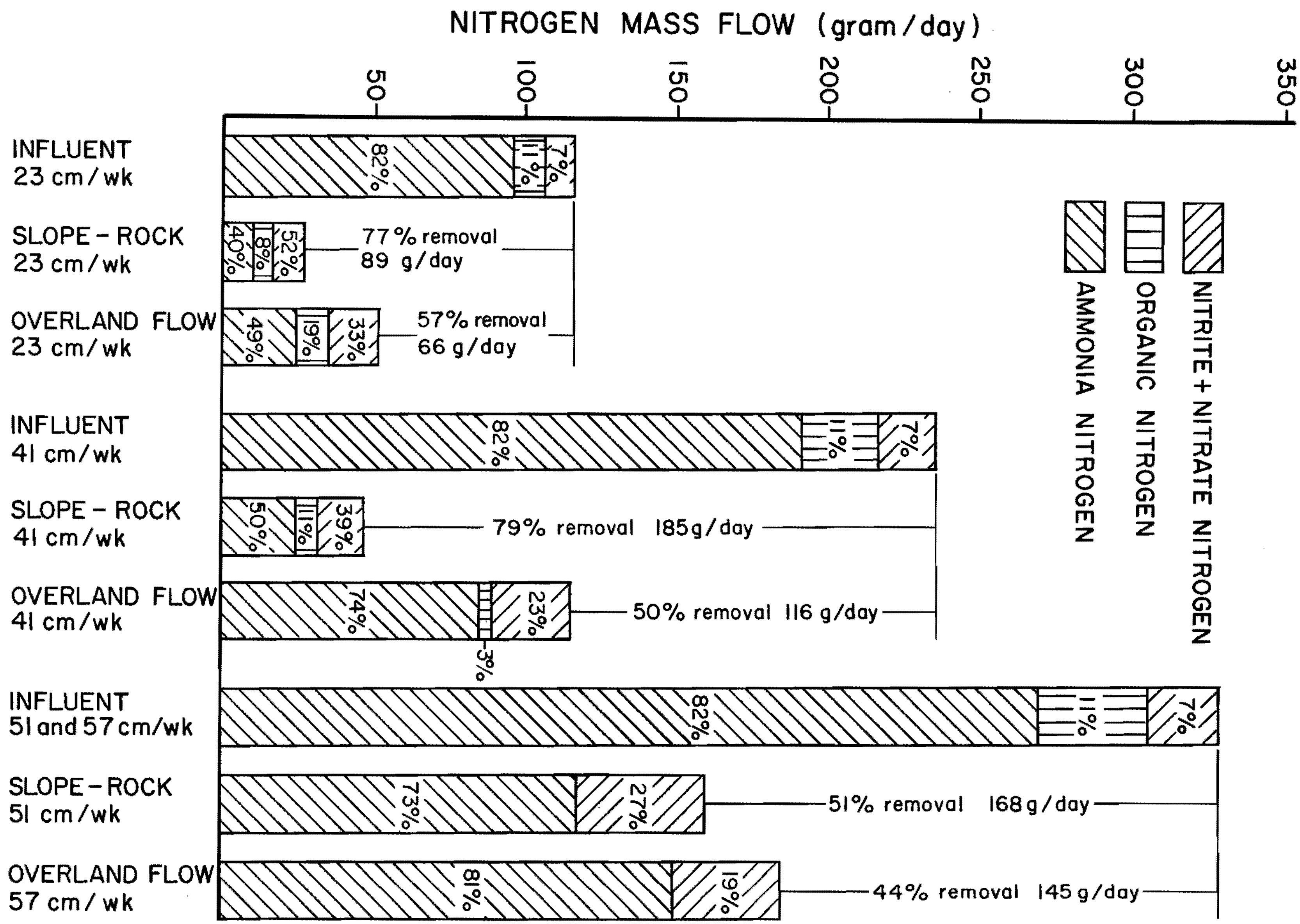


Figure 45. Nitrogen mass loading and discharge rate for the second year of operation.

Table 32. Specific nitrogen mass removal rates.

Year/Loading Rate/ Treatment	Ammonia-Nitrogen	Nitrate-Nitrogen	Nitrite-Nitrogen	Total Kjeldahl Nitrogen	Total Nitrogen
Year 1:			kg/ha/day		
13 cm/wk					
Overland Flow	3.59	-0.17	-0.17	4.98	4.64
Slope-Rock	3.45	-0.13	-0.05	4.82	4.64
20 cm/wk					
Overland Flow	4.36	-0.52	-0.34	5.91	5.05
Slope-Rock	5.22	-0.34	-0.30	7.12	6.48
Year 2:					
23 cm/wk					
Overland Flow	1.99	1.30	0.06	2.79	4.15
Slope-Rock	0.91	1.12	0.03	1.09	2.24
41 cm/wk					
Overland Flow	6.35	1.75	0.22	6.62	8.59
Slope-Rock	1.77	1.25	0.14	2.15	3.62
57 cm/wk					
Overland Flow	11.08	2.64	0.15	10.40	12.97
Slope-Rock	7.74	2.42	0.16	7.68	10.48

plots compared to overland flow plots at similar loadings. In the second year, on the 23 cm/wk slopes, ammonia removals averaged 20 to 36 percent. On the other sections only 4 to 13 percent of the ammonia was removed. Statistical analysis of data for the second year showed that the ammonia concentration of the intermediate sample for the 23 cm/wk slope rock plot was significantly different from the ammonia concentration for the 57 cm/wk slope rock plot, which was probably due to the differences in depth of flow and contact time. Otherwise, ammonia concentrations at the 9 m sampling points were not statistically different. Ammonia concentrations continued to decrease as the wastewater flowed down the treatment slope.

During the first year of operation, nitrite plus nitrate concentrations (Figure 48) increased from 0.39 to 5.24 and 5.59 mg/l in the first 9 m of the slope-rock plots, while smaller increases (0.39 to 1.87 and 2.07) were noted in the first 9 m of the overland flow fields. After the initial increase in the first 9 m, nitrite plus nitrate concentrations declined down the slope, except for the 20 cm/wk overland flow slope, where concentrations continued to increase slightly. On the slope-rock plots, nitrite plus nitrate concentrations increased more than on the overland flow sections. Loading rate had no effect on performance in the first 9 m.

In general, the slope-rock plots exhibited lower ammonia removals but higher nitrite and nitrate production. The mineralization of organic nitrogen to ammonia and subsequent nitrification appeared to be enhanced by the gravel layer compared to the grass of the overland flow plots.

During the second year of operation, except for the 23 cm/wk slope rock

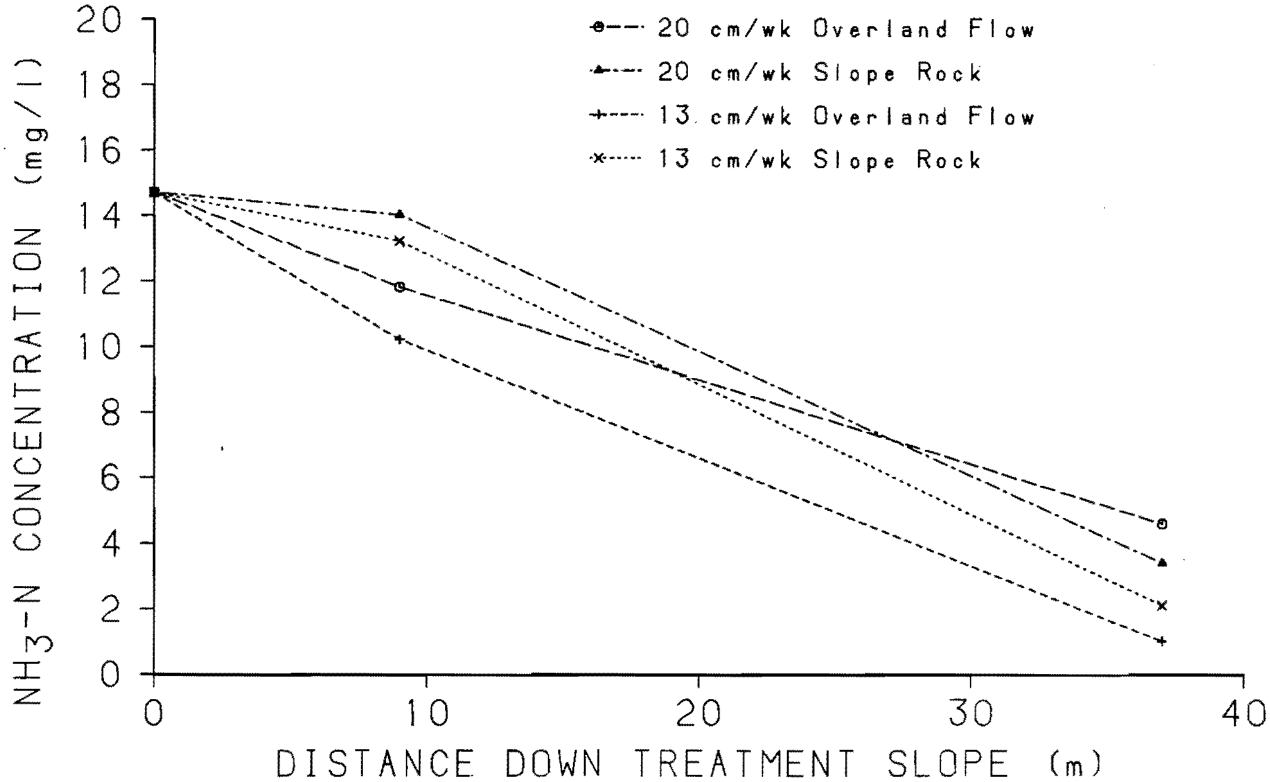


Figure 46. Ammonia-nitrogen concentrations versus distance down the treatment slope for the first year of operation.

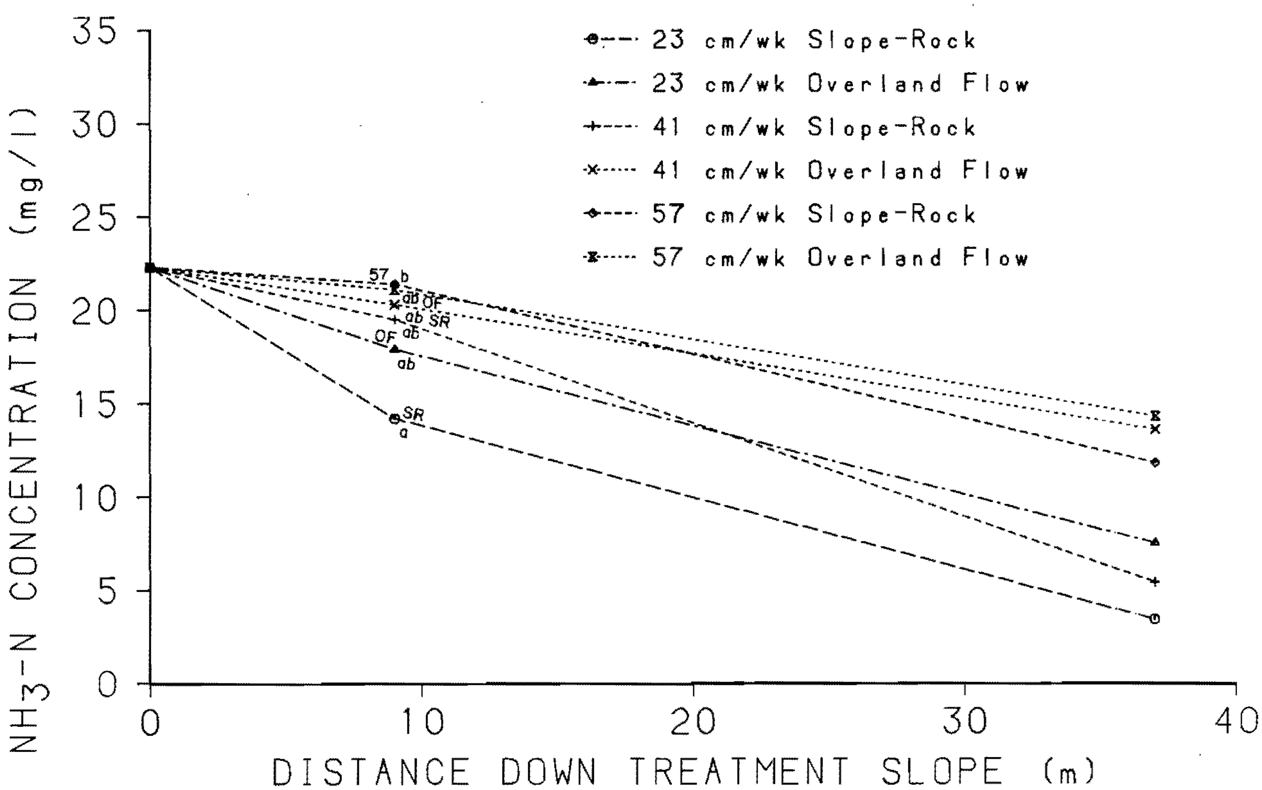


Figure 47. Ammonia-nitrogen concentration versus distance down treatment slope the second year of operation.

section, nitrate concentrations (Figure 49) did not greatly increase by the 9 m sampling point. However, only the intermediate nitrate concentrations on the 23 cm/wk slope rock section and the 57 cm/wk slope-rock section were significantly different. Nitrification at the higher loading rates, and especially on the 57 cm/wk slope rock section may have been limited due to shorter contact times and thicker surface water films which hindered oxygen diffusion and therefore nitrification.

Nitrite accumulated on the 41 cm/wk slope-rock section without further nitrification to nitrate (Figure 50). However, no significant differences in nitrite levels between slopes were observed at the 9 m sampling point.

Twenty-one percent of the total Kjeldahl nitrogen was removed by the gravel zone on the 23 cm/wk slope-rock plot (Figure 51). The 57 cm/wk overland flow slope achieved 11 percent removal. The TKN on the 41 cm/wk overland flow and 51 cm/wk slope-rock sections increased in the first 9 m. The 23 cm/wk slope-rock plot was significantly different than these two slopes with TKN increases. Statistical analyses indicated no TKN removal improvements attributable to any treatment type or hydraulic loading rate for the first 9 m. Since TKN includes NH_3-N, most of the activity in the first 9 m was likely conversions to different nitrogen forms. Since the total Kjeldahl nitrogen removal at the 9 m sampling points did not match effluent removal trends, the loose soil may have contaminated the intermediate samples and interfered with the TKN determinations.

For the second year of operation, weekly variations in ammonia and nitrite- plus nitrate-nitrogen concentrations are shown in Figures 52 and 53. Peak concentrations at week 15 were due to harvesting operations.

System age

During the first year of operation ammonia-nitrogen effluent concentrations did not appear to change with system age except for the slope-rock sections which took at least 4 weeks to stabilize (harvesting perturbed the system at week 5). During the second year, the period for the systems to acclimate to achieve maximum ammonia removal by nitrification was approximately 4 weeks. During both years, at the onset of cooler weather, ammonia and nitrate levels in the influent, effluent, and intermediate samples increased. Possible contributing factors to these increases include reduced nutrient requirement by the cover crop due to a decline in growth rate and greater dissolved oxygen concentrations in the water, which are conducive to nitrification competing with lower temperatures which reduce the kinetic rate for the nitrification process.

Two-year summary

Overall, ammonia-nitrogen and total nitrogen removal efficiencies decreased as flow rate increased. Ammonia removals ranged from 68 to 93 percent at the lower loadings (13 cm/wk to 23 cm/wk), while at the highest loading (57 cm/wk), only 36 to 44 percent was removed. Total nitrogen removal ranged from 33 to 79 percent, with lower removals generally seen at higher loading rates. Nitrification occurred on all slopes, as nitrate-nitrogen and nitrite-nitrogen were present in effluents from each slope.

Fecal Coliforms

The results of fecal coliform analyses for both years of operation are given in Table 33. During the first year of operation, approximately 99 percent removal of fecal coliforms was obtained by the overland flow sections at the 13 and 20 cm/wk application rates, while the slope-rock sections achieved only 87-88 percent removal

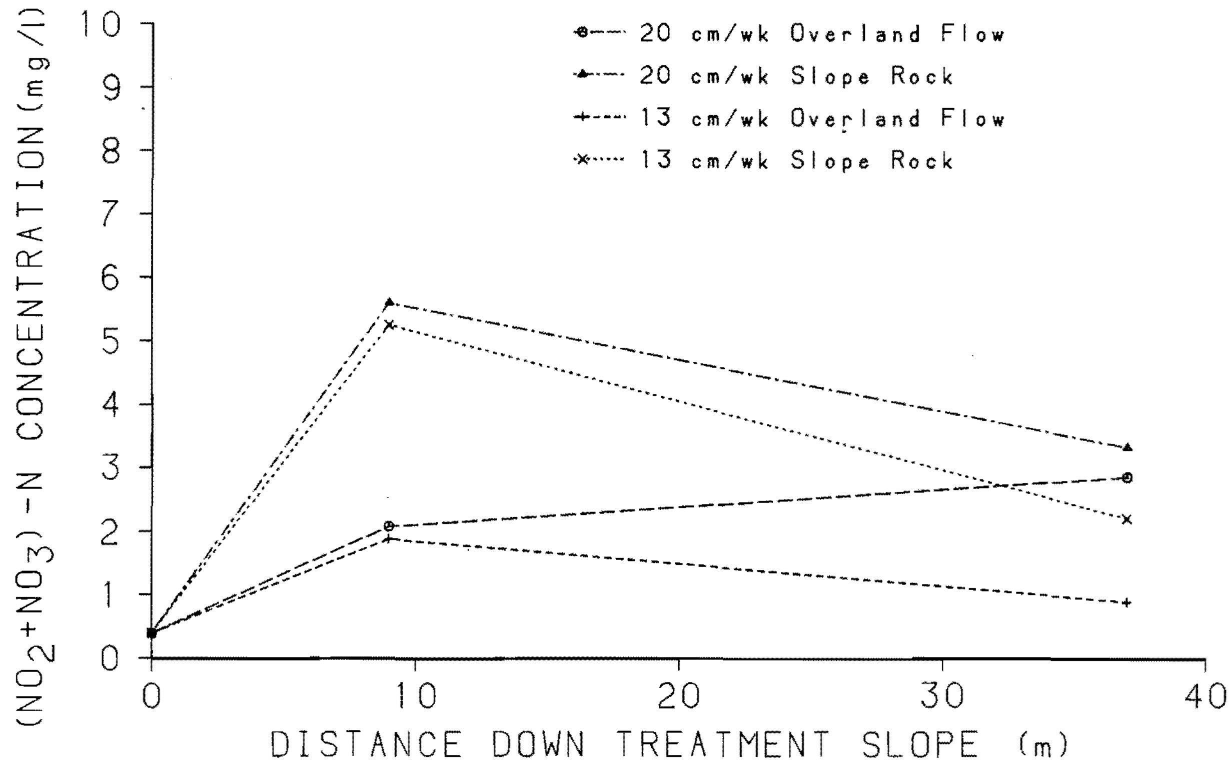


Figure 48. Nitrite- plus nitrate-nitrogen versus distance down the treatment slope for the first year of operation.

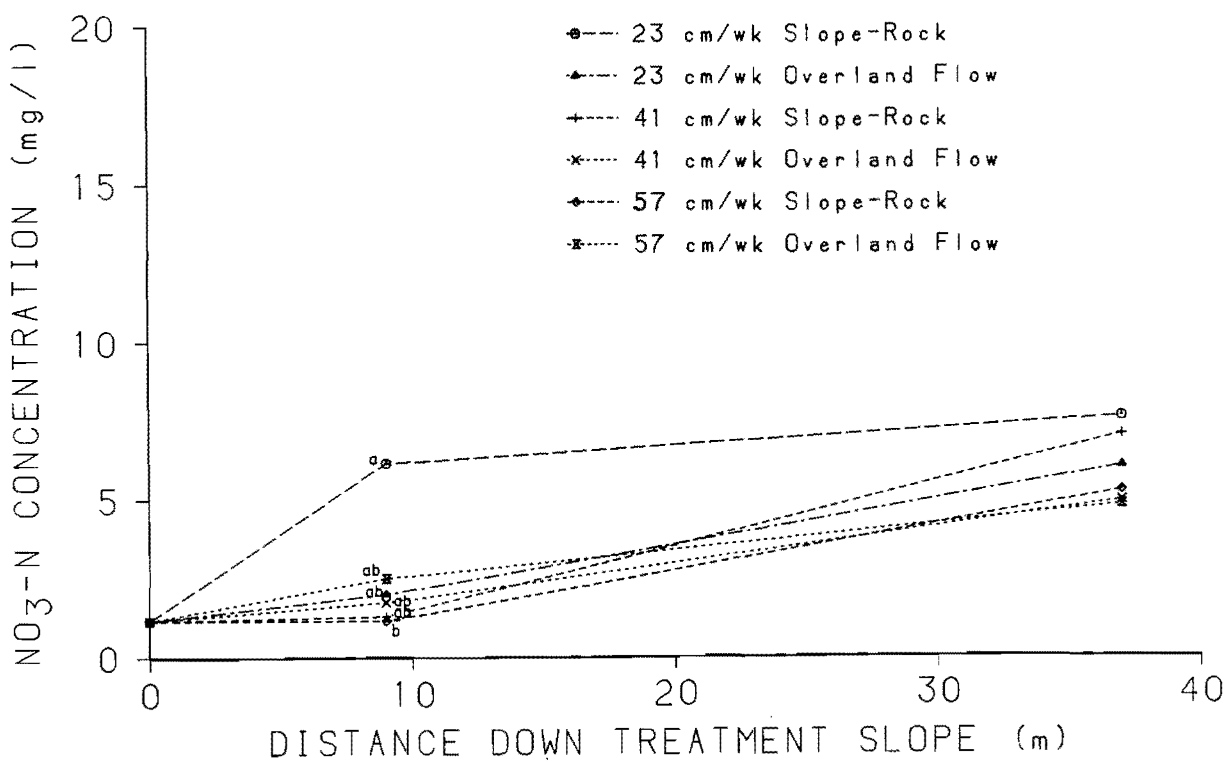


Figure 49. Nitrate-nitrogen concentration versus distance down the treatment slope the second year of operation.

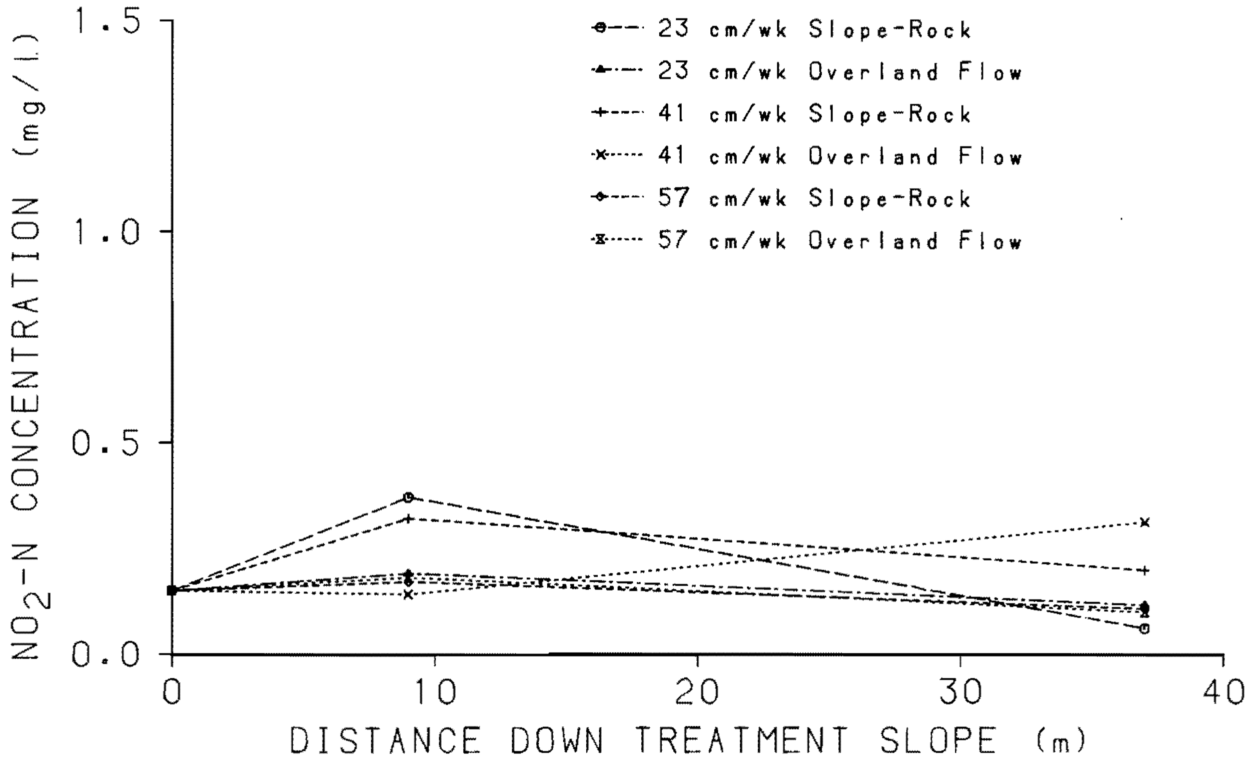


Figure 50. Nitrite-nitrogen concentration versus distance down the treatment slope for the second year of operation.

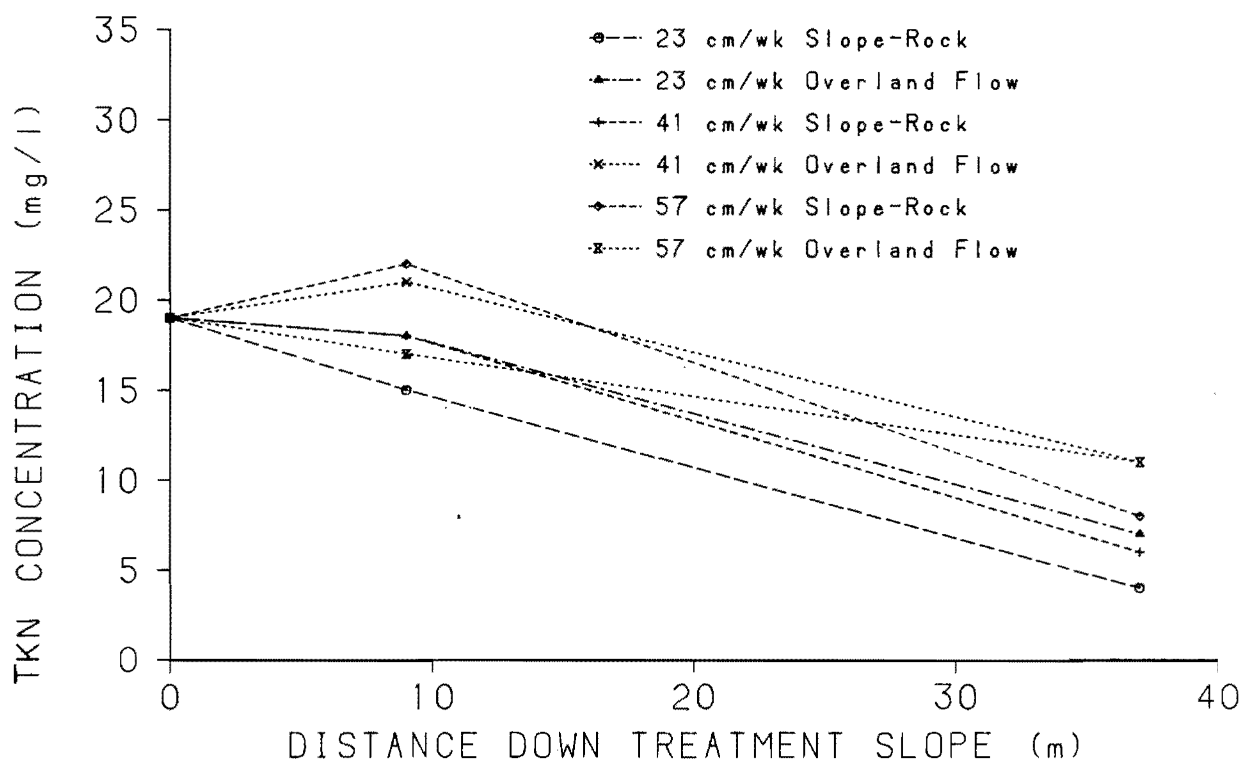


Figure 51. Total Kjeldahl nitrogen concentration versus distance down the treatment slope for the second year of operation.

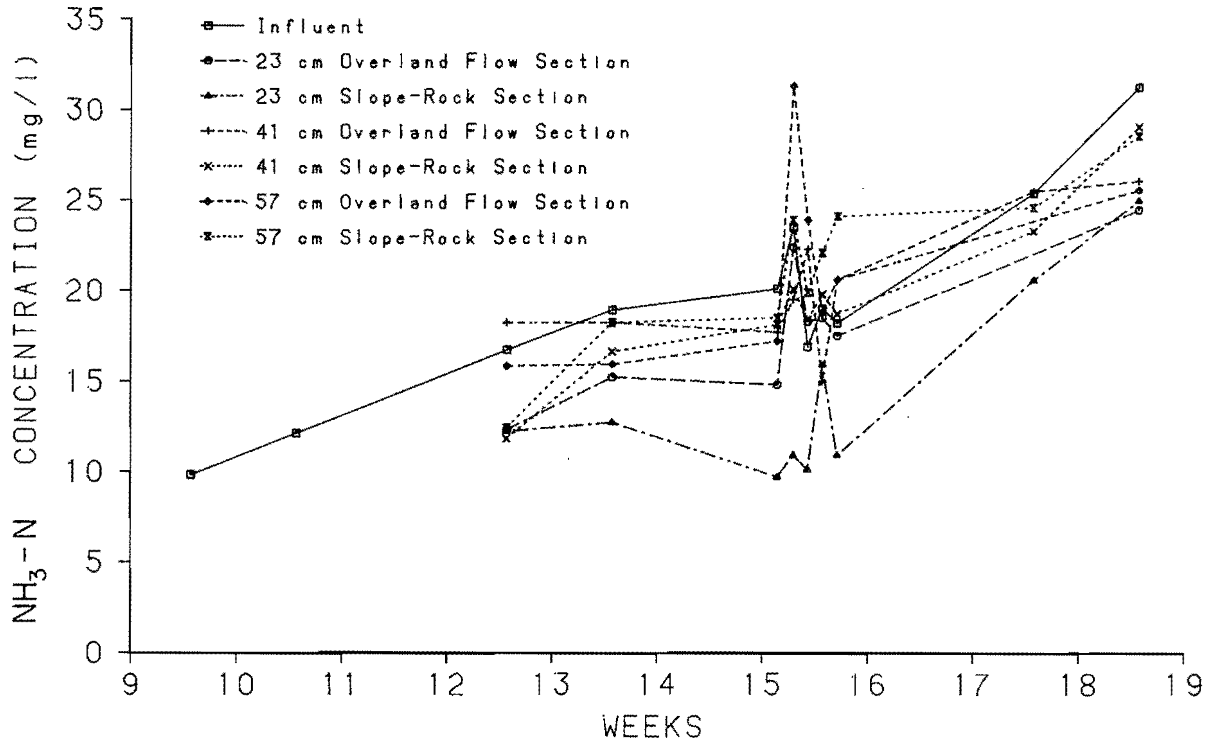


Figure 52. Intermediate ammonia-nitrogen variations during the second year of operation.

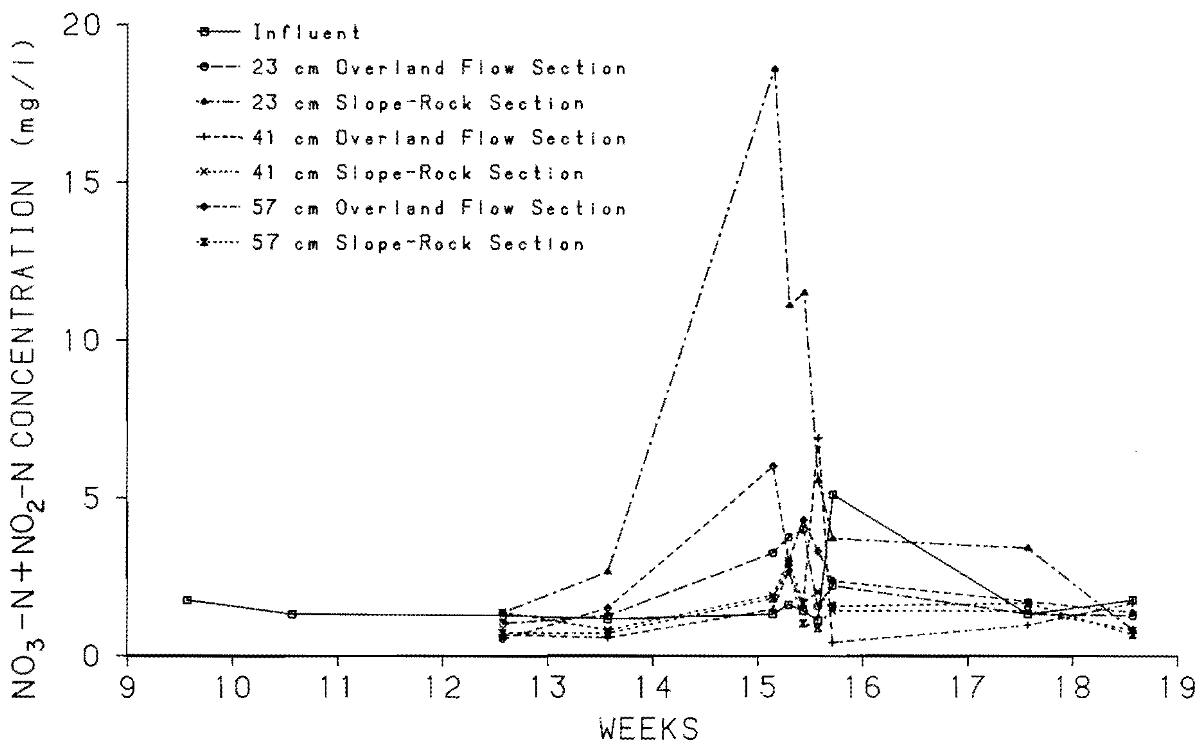


Figure 53. Intermediate nitrite- plus nitrate-nitrogen variations during the second year of operation.

at the same rates. In the second year, only one sample was analyzed. The results from the sample showed 97-99 percent removal for all plots, except for 92 percent removal on the 23 cm/wk overland flow plot. The results agree with data reported by Thomas et al. (1976) that greater than 90 percent removal of fecal coliforms can be realized.

Even with these high removal levels, mean levels of fecal coliforms in effluents from the plots ranged from 2.4×10^4 to 1.9×10^6 colonies/100 ml. Therefore, disinfection of the effluent from all the treatment systems would be necessary before discharge.

Harvesting

During the first year of operation, grass was harvested during the 2-day rest period between the fourth and fifth weeks of wastewater application and harvested again at the end of the growing season. No time was allowed for drying the plots since only a small gasoline mower was used to cut the grass. The clippings were raked and removed, but a substantial amount of residue remained on the slopes. The effluent from the slopes became yellowish-green and caused foaming at the effluent collection pump. The foaming subsided by the end of the fifth day following cropping.

Prompted by the visible degradation in effluent quality, the effects of harvesting on the performance of the treatment slopes were closely monitored the fifth week (July 30 to August 3, 1979) of wastewater application. During week 6, samples were taken on Tuesday and Thursday and during week 7 on Tuesday, Wednesday, and Thursday for comparison of daily effluent variations

Table 33. Removal of fecal coliform bacteria.

Year/Loading Rate/ Treatment	Fecal Coliform Bacteria: Influent n	Influent Mean (colonies/ 100 ml)	Influent Standard Deviation (colonies/ 100 ml)	Effluent n	Effluent Mean (colonies/ 100 ml)	Effluent Standard Deviation (colonies/ 100 ml)	Removal (%)
Year 1:	4	1.5×10^7	6.0×10^6				
13 cm/wk							
Overland Flow				4	1.7×10^5	8.5×10^4	99
Slope-Rock				4	1.9×10^6	1.1×10^6	87
20 cm/wk							
Overland Flow				4	1.9×10^5	1.3×10^5	99
Slope-Rock				4	1.8×10^6	1.2×10^6	88
Year 2:	1	3.5×10^6					
23 cm/wk							
Overland Flow				1	2.7×10^5		92
Slope-Rock				1	1.1×10^5		97
41 cm/wk							
Overland Flow				1	2.4×10^4		99
Slope-Rock				1	4.6×10^4		99
51 cm/wk							
Overland Flow				1	6.0×10^4		98
Slope-Rock				1	2.4×10^4		99

without harvesting. No Monday samples were available for comparison due to little or no runoff for analyses.

Different harvesting equipment was used for the two harvest periods in the second year of operation. During the first harvest (between the sixth and seventh weeks after 5 to 6 days of drying the slopes), a tractor-driven rotary blade was used. For the second harvest, prior to week 15, a tractor-powered sickle bar mower was used to crop the grass after 5 days of drying the slopes. Daily data were collected during week 5 as a control to compare treatment efficiency during the 5-day application cycle before cropping to the treatment efficiency after harvesting (weeks 7 and 15).

As in the first year, a deterioration in effluent quality from all slopes was visible after harvesting of the cover crop. The effluent changed from clear to a yellow-green color which faded as the week progressed. The visible deterioration of effluent quality after harvesting was probably due to the release of chlorophyll pigments when the blades of grass were severed and wetted. Algae on the surface of the fields could also have contributed to the green color in this runoff.

Organic pollutants and suspended solids

During the harvest period the first year (week 5), effluent BOD_5 concentrations from both overland flow sections were 118 mg/l, which exceeded the influent BOD_5 concentration (82 mg/l) the first wastewater application day following cropping (Figure 54). Effluent BOD_5 concentrations improved through the week, but only the 20 cm/wk slope-rock plot recovered in one day of wastewater application. The 13 cm/wk slopes fell below 20 mg/l by the second day. In comparison, the BOD_5 effluent values for the sixth and seventh weeks did not exceed 13 mg/l. Therefore, the elevated BOD_5 levels in the effluent were derived from mulched residues left on the slopes.

Effluent suspended solids were also affected by the harvesting procedure (Figure 55). The 13 and 20 cm/wk overland flow slopes yielded 50 and 41 mg/l effluent suspended solids, respectively, the first day following cropping. There was no improvement the second day. The effluent SS from the slopes hydraulically loaded at 13 cm/wk subsequently improved, falling well below 15 mg/l by the third day. The 20 cm/wk overland flow slope did not improve to satisfactory levels until the last day of the application cycle. The average effluent BOD_5 and SS concentrations for the week following harvesting are given in Table 34. Both BOD_5 (20 mg/l, 7-day average) and SS (12 mg/l, 7-day average) State of Utah discharge limits were exceeded by the conventional overland flow treatment slopes. Only the SS effluent discharge limits were exceeded by the slope-rock fields.

Suspended solids data collected for weeks 7 and 8 were below 15 mg/l. Consistent SS removal below 10 mg/l was achieved the second week after harvesting except for the violation of SS discharge standards caused by the harvesting procedure.

Effluent volatile suspended solids reached maximum values of 41 and 42 mg/l immediately after harvesting. The suspended solids in the runoff were almost entirely (84 to 100 percent) volatile (Figure 56).

The mulched grass clippings, inadvertently left on the slopes, increased effluent BOD_5, SS, and VSS concentrations when wastewater application resumed. System startup contributed some background BOD_5, but satisfactory suspended solids removal had already been attained before the harvest.

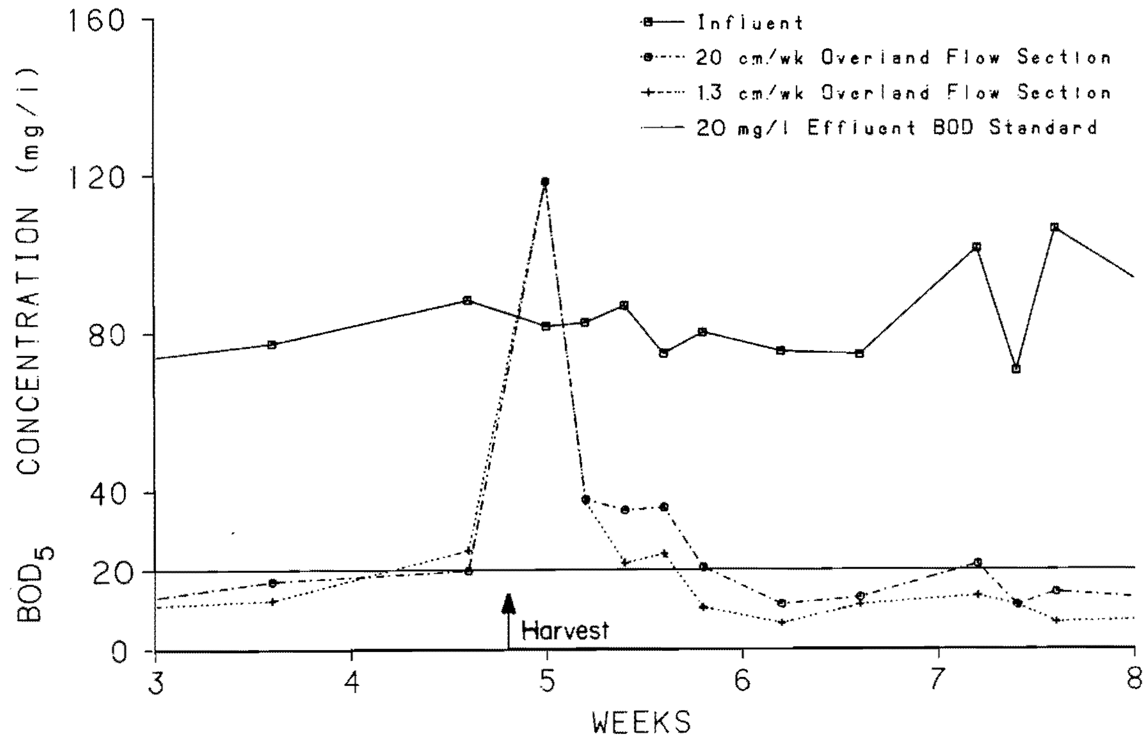


Figure 54. Influent and effluent BOD_5 concentrations following cropping of the overland flow sections.

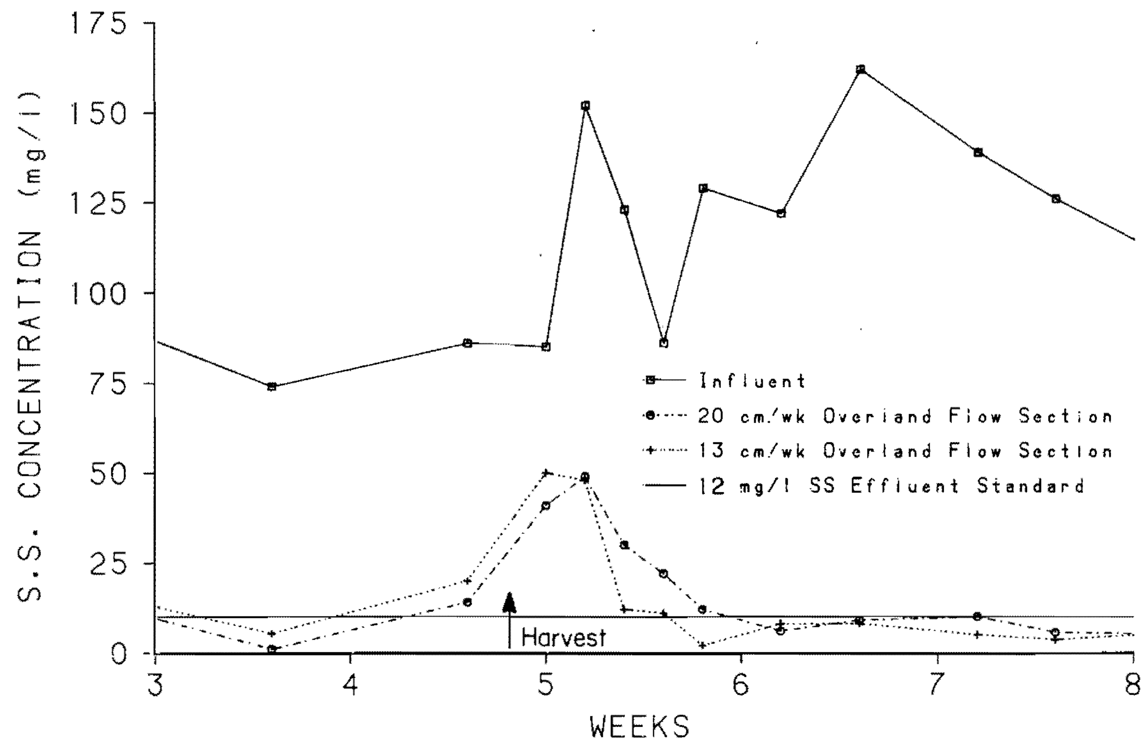


Figure 55. Influent and effluent suspended solids concentrations following cropping of the overland flow sections.

Table 34. Mean effluent BOD_5 and SS concentrations for the week following cropping (week 7) the first year of operation.[a]

Parameter	20 cm/wk Overland Flow	13 cm/wk Overland Flow	20 cm/wk Slope-Rock	13 cm/wk Slope-Rock
BOD_5 (mg/l)	49 (39)	16 (9)	42 (44)	14 (4)
Suspended Solids (mg/l)	30.8 (14.7)	18.4 (14.4)	24.6 (22.6)	14.5 (13.0)

[a]Standard deviations in parentheses.

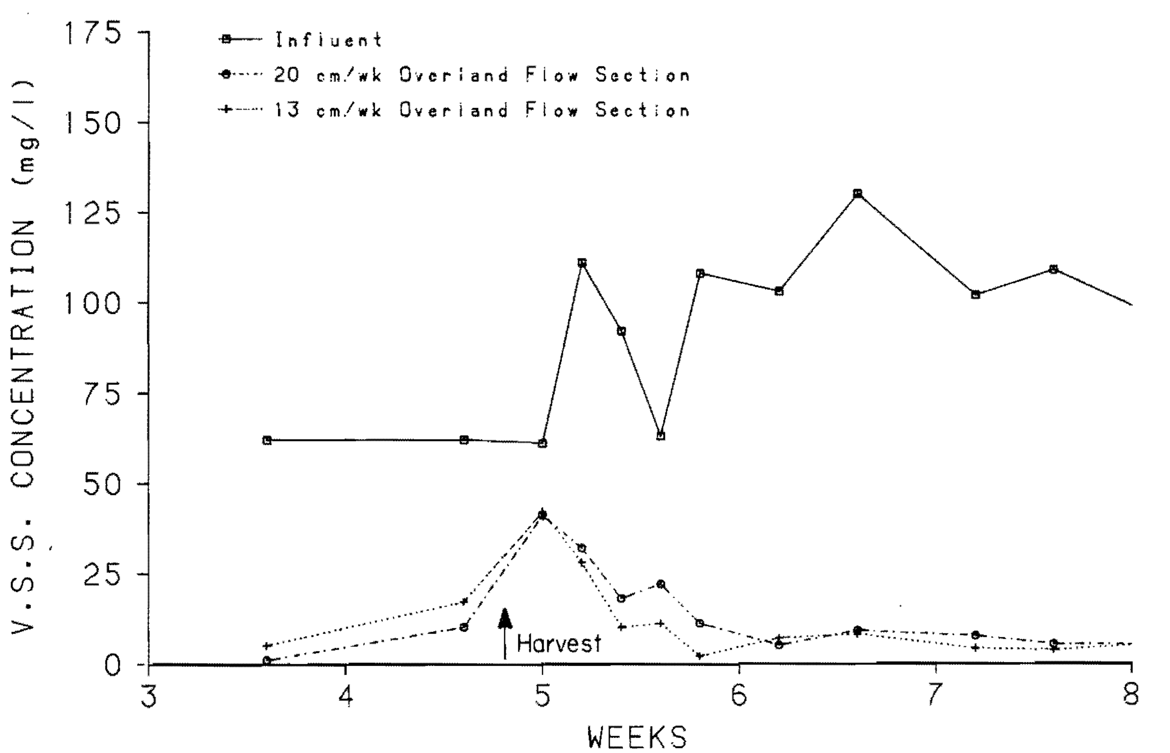


Figure 56. Influent and effluent volatile suspended solids following cropping of the overland flow sections the first year of operation.

Effluent BOD_5 and SS concentrations before cropping (week 5) and following cropping (weeks 7 and 15) are shown in Figures 57 and 58 for the second year of operation. The mean BOD_5 and SS values for weeks 5, 7, and 15 are given in Table 35. The Utah State BOD_5 effluent limit (20 mg/l, 7-day average) was not exceeded following harvesting operations. However, the Utah criteria for effluent SS (12 mg/l, 7-day average) was exceeded after the harvest (week 7) using a rotary blade but not after the harvest (week 15) using the sickle-bar mower. The violation occurred because of an initial high value (35.3 mg/l SS) that may have resulted from the dirt added to seal the effluent collection troughs and not as a result of harvesting operations. The extended drying had severely shrunk and cracked the clay near the aluminum collection baffles. Topsoil was used to fill in the cracks, and loose dirt may have then contaminated the effluent samples.

Figure 59 compares BOD_5 removal efficiencies during weekly cycles both before and after harvesting. BOD_5 removal increased as the application cycle progressed at all loading rates. The slopes loaded at the highest rates exhibited the least drop in BOD_5 removal efficiency (55 to 65 percent) at the start of an application cycle, both with and without harvesting.

Table 35. Mean effluent BOD_5 and SS concentrations for the weeks before harvesting and the weeks following harvest the second year of operation.

Parameter	Week	23 cm/wk Slope-Rock	23 cm/wk Overland Flow	57 cm/wk Slope-Rock	57 cm/wk Overland Flow
BOD_5 (mg/l)	5	7 (4)[a]	6 (2)	8 (1)	16 (7)
	7	8 (7)	8 (5)	12 (8)	17 (12)
	15	8 (4)	9 (5)	12 (12)	11 (9)
Suspended Solids (mg/l)	5	6.3 (4.7)	7.9 (5.3)	5.0 (2.0)	10.0 (5.1)
	7	10.2 (11.4)	14.9 (17.7)	11.9 (11.1)	11.2 (6.4)
	15	7.2 (4.7)	8.4 (2.8)	10.0 (5.0)	9.5 (4.4)

[a]Standard deviations in parentheses.

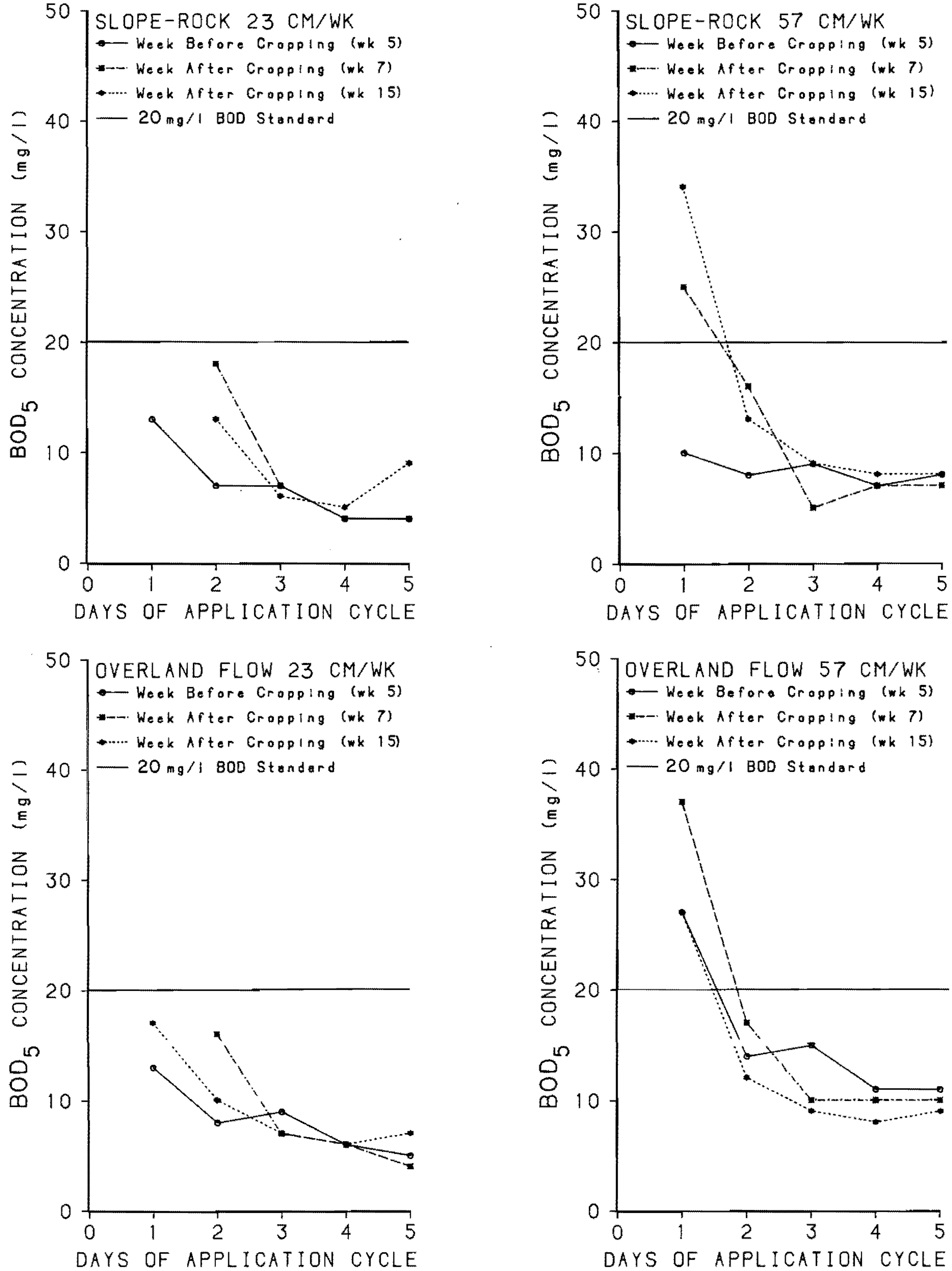


Figure 57. Effluent biochemical oxygen demand response to harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

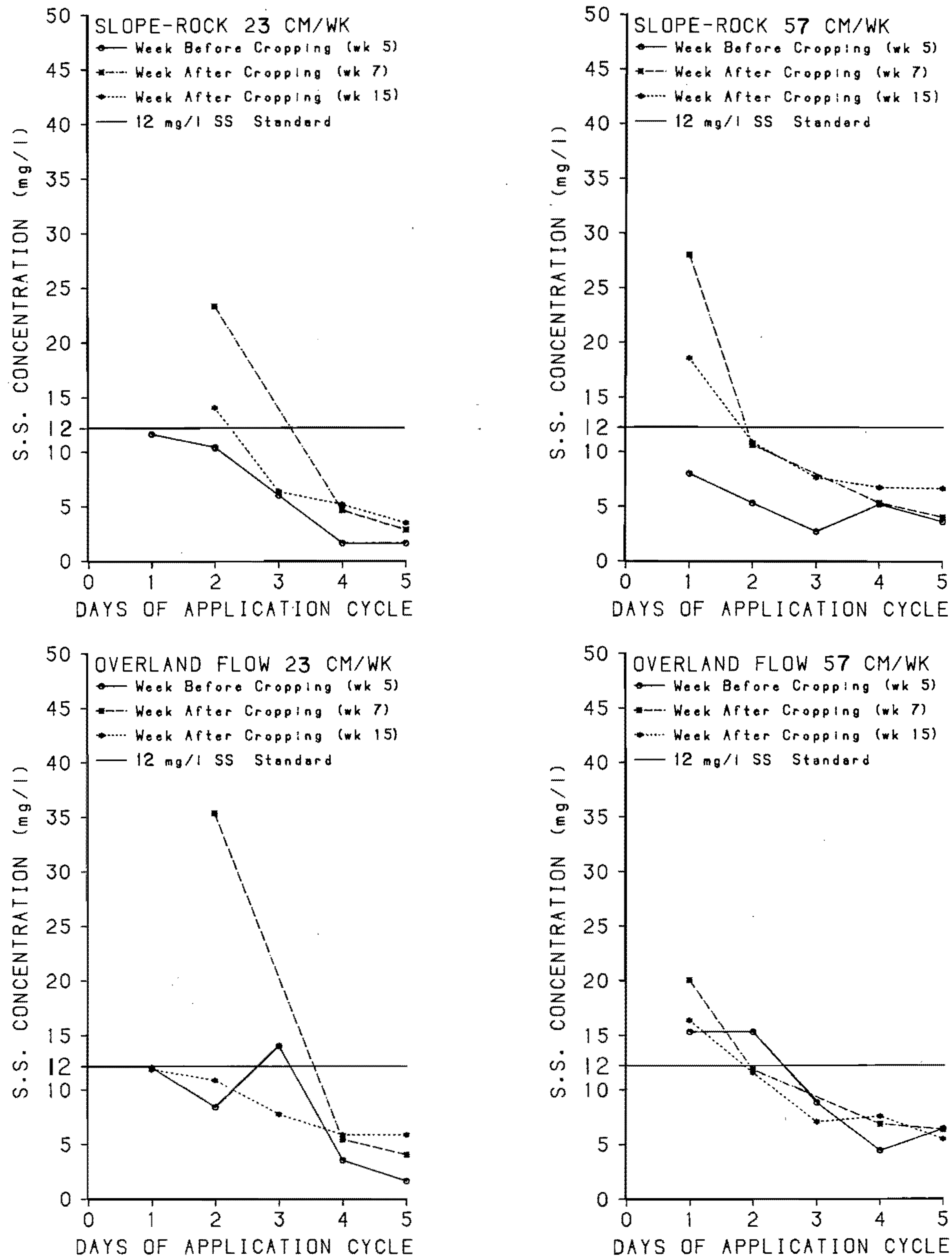


Figure 58. Effluent suspended solids response to harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

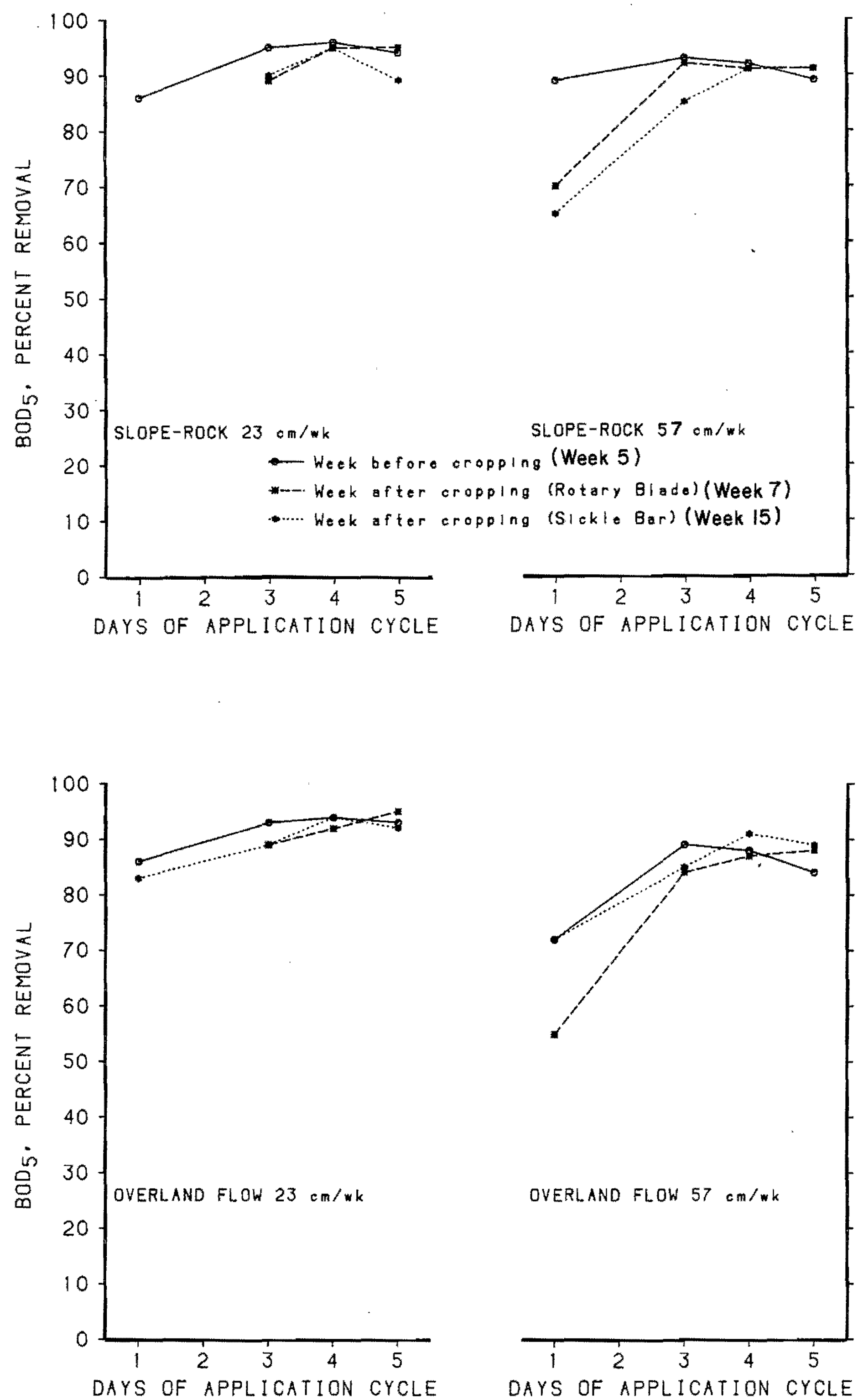


Figure 59. BOD_5 removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

BOD_5 percent removal was slightly lower with harvesting than without harvesting at the beginning of an application cycle. Removal efficiencies, however, returned to levels observed before harvesting by the end of the week following harvesting. Thus, the effects of harvesting on BOD_5 removal lasted less than one week and did not result in effluent discharge violations (greater than 20 mg/l, 7-day average) when the residue was removed from the slopes.

Figure 60 shows suspended solids removal efficiencies without and with harvesting. In both cases efficiencies increased as the application cycle progressed. Prior to harvest, SS removal efficiencies (83-95 percent) were higher at the beginning of the week than they were after harvest (67-91 percent). The lowest removal efficiency (67 percent) was likely due to soil contamination. By the end of the week SS removal efficiencies both before and after harvesting, at all loading rates, were as high as 93-99 percent.

Table 36 displays the volatile percentage of the total suspended solids. Most of the suspended solids in the effluent were volatile solids except for day 4 in weeks 5 and 7 for the 23 cm/wk plots. Harvesting does not appear to have any noticeable effects on volatile percentage of suspended solids

Table 36. Volatile suspended solids as a percentage of total suspended solids.

Week	Day	20 cm/wk Slope-Rock	20 cm/wk Overland Flow	57 cm/wk Slope-Rock	57 cm/wk Overland Flow
Before Harvest		------------------------	%	------------------------	
5	1	90	87	75	83
5	2	100	99	100	100
5	3	100	96	100	64
5	4	50	43	85	91
5	5	100	100	100	100
After Harvest (Rotary Blade)					
7	1	-	-	59	76
7	2	39	31	83	76
7	3	-	-	-	-
7	4	43	50	68	74
7	5	100	100	100	100
After Harvest (Sickle Bar)					
15	1	-	100	88	96
15	2	78	77	93	91
15	3	75	79	79	83
15	4	71	72	72	87
15	5	79	88	91	93
Ave.		77	79	83	88

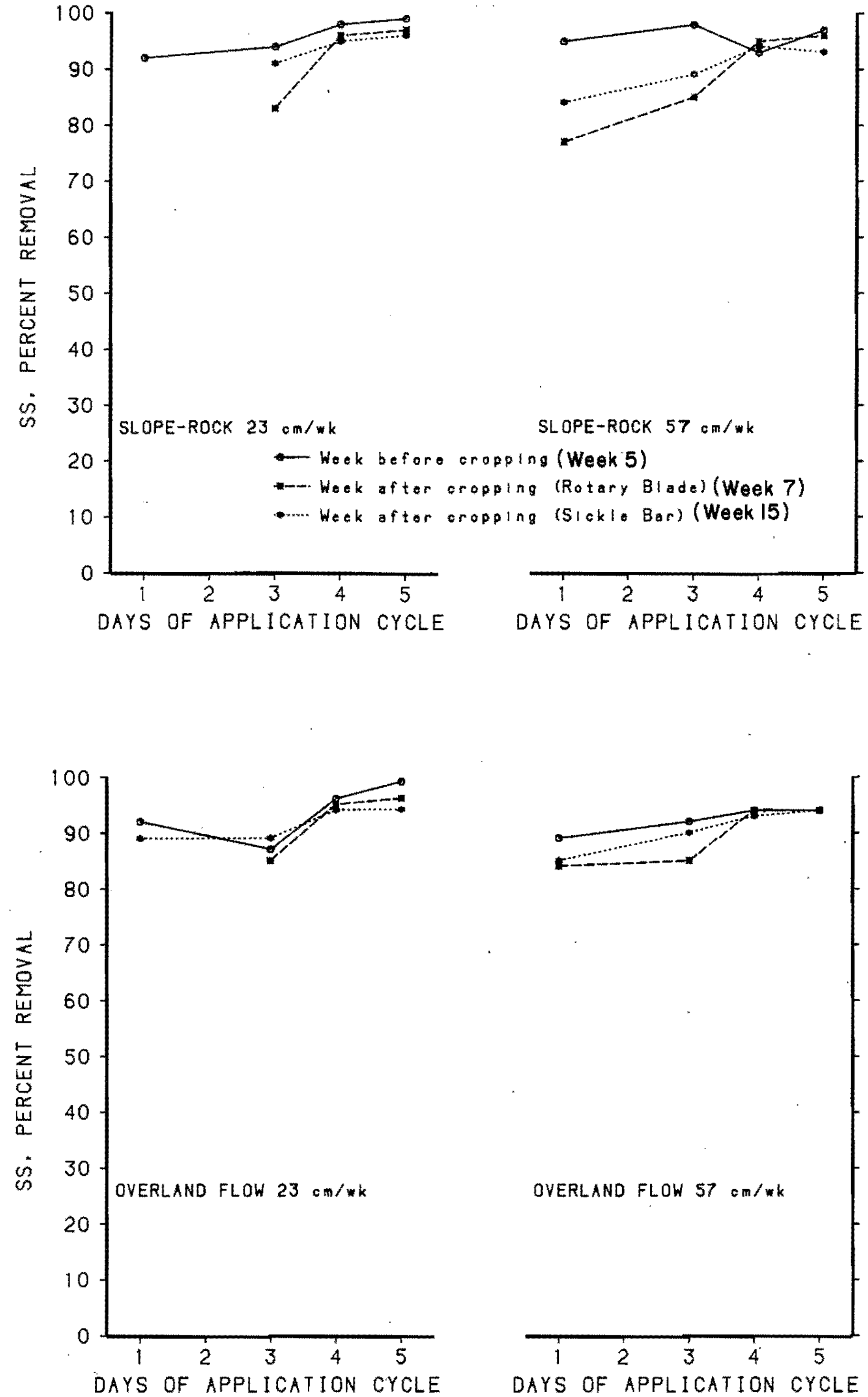


Figure 60. Suspended solids removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

in the effluent. The drop at the beginning of week 7 is explained by the soil added to seal the effluent collection system, which eroded and contaminated the effluent samples.

The increase in effluent BOD_5 concentration when system operation was resumed following weekend drying can be attributed in part to decreased microbial population due to dessication and an increase in suspended solids. The increase in suspended solids concentration is probably due to the scouring of dried (easily broken) litter which accumulated on the slopes near the effluent troughs as shown in Figure 61. BOD_5 removal efficiencies recovered to levels obtained prior to harvest within 2 days at the highest hydraulic loading. The 20 cm/wk slopes took only 1 day to recover. Volatile suspended solids (VSS) comprised the major percentage of effluent suspended solids. The VSS fraction decreased at the start of week 7, due to the soil contamination. Biochemical oxygen demand and suspended solids (7-day average) effluent discharge limits were not exceeded due to harvesting in the second year.

Phosphorus

After harvesting during the first year, total phosphorus in the effluents the second day after cropping are similar to the Tuesday effluent levels obtained for weeks 5 and 7 (Figure 62). Effluent phosphorus data for the first day after cropping are missing.

Effluent orthophosphate concentrations, however, substantially increased after harvesting (Figure 62). The effluent orthophosphate concentrations (8.2 and 8.8 mg/l) were more than double the influent concentration (3.7 mg/l). Part of the increase was the result of

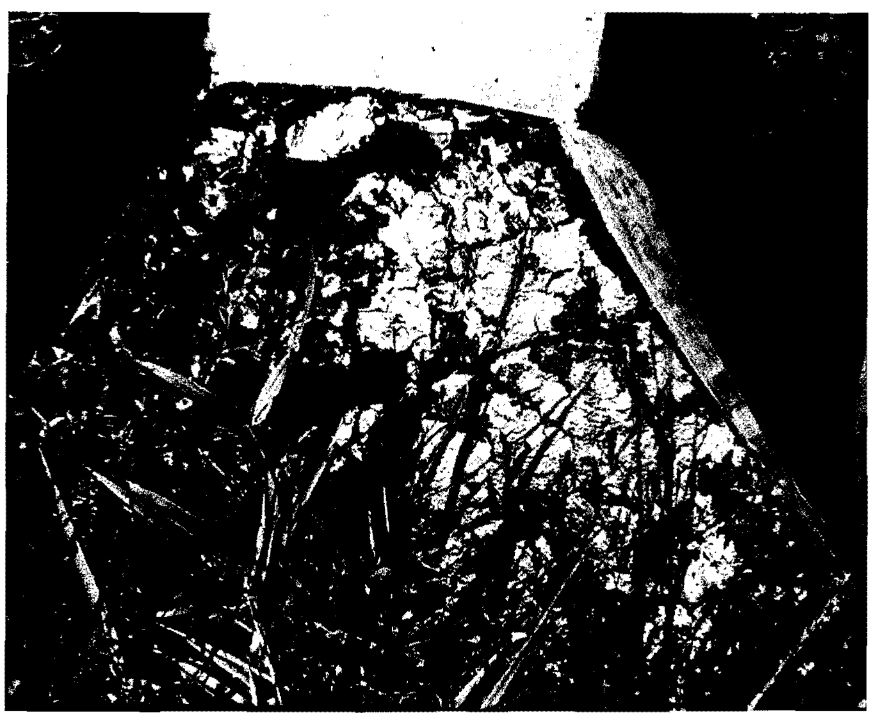


Figure 61. Dessicated solids became dry and brittle during test periods.

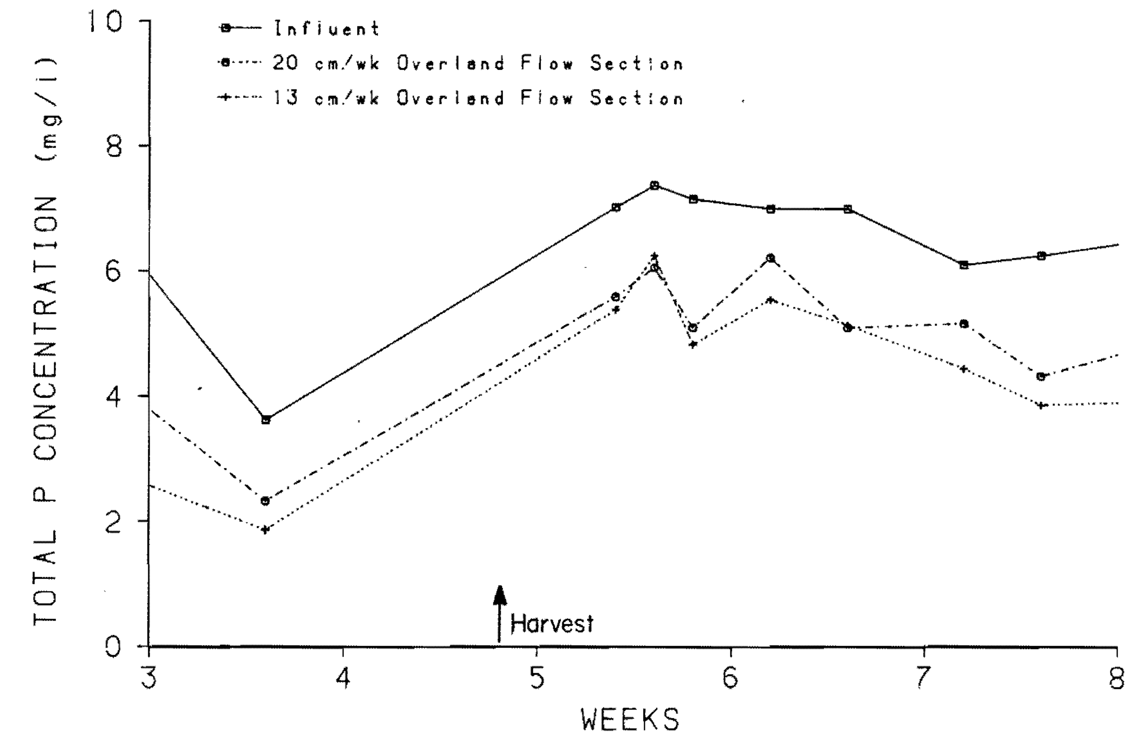


Figure 62. Influent and effluent total phosphorus concentrations following cropping of the overland flow sections the first year of operation (after Zirschky 1980).

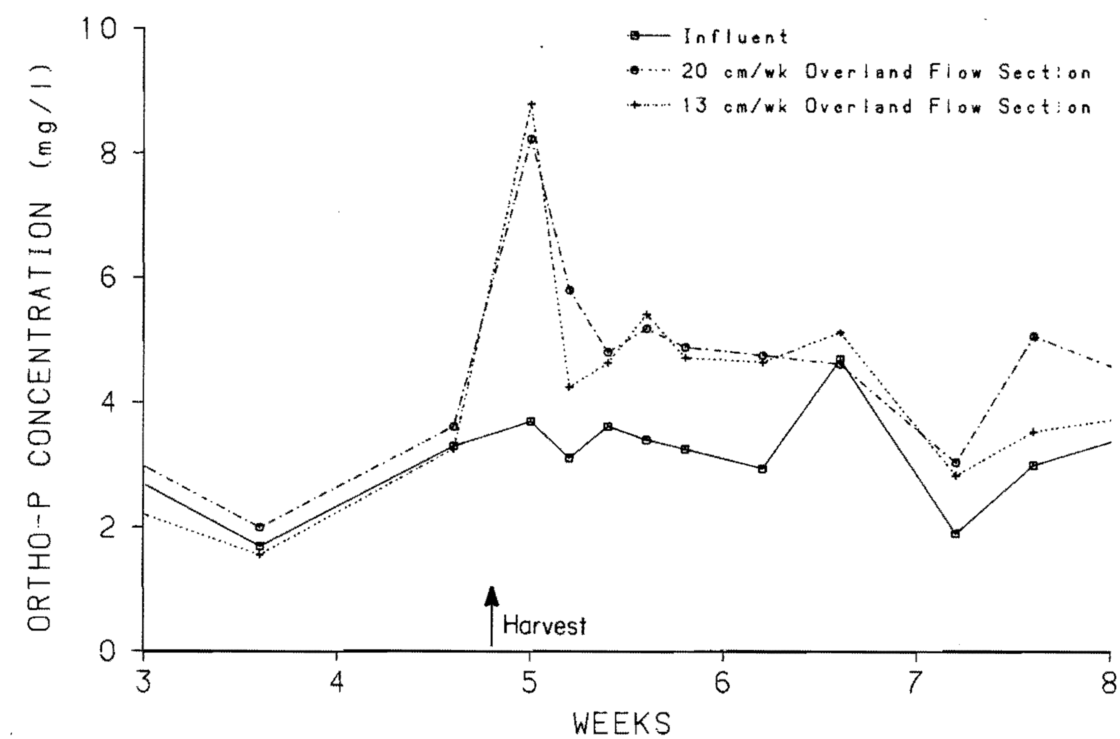


Figure 63. Influent and effluent orthophosphorus concentrations following cropping of the overland flow sections the first year of operation (after Zirschky 1980).

residual inorganic phosphorus leached from the soil and microbial mineralization of organic phosphorus. Reddy et al. (1978) reported that the effluent-treated soils they studied had considerable NH_4-Cl extractable inorganic phosphorus in the top 5 cm after cropping, reflecting both residual soluble inorganic phosphorus and microbial mineralization of the added organic phosphorus. Some of the increase in orthophosphate phosphorus could have been derived from the grass residues since the grass assimilates approximately 10 percent of the applied phosphorus (EPA 1977) and can be washed out (Husted 1974).

For the second year, total phosphorus removals during the weeks before and after harvesting are shown in Figure 64. Average total phosphorus removal for the whole growing season (Thursday data only) for the 23 cm/wk loadings was 29 to 30 percent. After cropping, phosphorus removal efficiencies dropped to 23 to 27 percent during week 7 and further dropped to 11 to 16 percent during week 15.

The slopes loaded at 57 cm/wk achieved an average of 24 percent total phosphorus removal. After harvesting, the removal efficiency decreased to only 2 to 10 percent. All slopes experienced a gradual increase in total phosphorus removal as the application cycle progressed both before and after harvesting. Data are missing for day 5 after both croppings due to difficulties with analyses.

Orthophosphate phosphorus concentrations were greater in the runoff from all slopes than the concentration present in the influent wastewater (Figure 65). The percent orthophosphate increase was higher at the beginning of each cycle after the weekend shutdown for aeration, but highest after harvesting. The increase in orthophosphate in week 5 prior to harvesting was probably due to the conversion of the phosphorus retained in the soil to soluble phosphorus, causing a shift in the dynamic equilibrium between soil phosphorus and wastewater phosphorus that led to desorption from the soil. The larger orthophosphate increases after harvest may be explained as washout of soluble phosphate from plant material. Husted (1974) reported from the findings of Gburek and Broyan (1974) that soluble phosphorus could be leached from cut or living plant material. Furthermore, living plants can supply leachable phosphorus for 2 to 3 days.

Nitrogen

Ammonia-nitrogen (NH_3-N) removals generally decreased immediately after harvesting (Figure 66 and 67). Ammonia can be released from herbage and decaying wastes (Viets 1974, Scott and Fulton 1978). However, during weekly application cycles, both before and after harvesting, ammonia removal efficiencies increased through the cycle, except the overland flow slope loaded at 57 cm/wk. Degradation of effluent quality from this slope was so severe that NH_3-N was leached from the slope on the second day of week 7.

The decrease in ammonia-nitrogen removal efficiencies at the beginning of the application cycle can be attributed to accumulated ammonia diffused out of the decaying wastes and the stress placed on the microbial population caused by the drying. The generation times for Nitrobacter agilis are 8 to 12 hours (Morrill and Dawson 1967) or about 1 day of wastewater application. The extended drying time necessary to complete harvesting added to this decrease and was intensified by the release of ammonia from the severed grass.

The improvement in ammonia removal efficiencies as the application period progressed may have resulted from the recovery of the microbial populations responsible for NH_3-N mineralization to NH_4-N and subsequent nitrification. The initial nitrite accumulation (negative

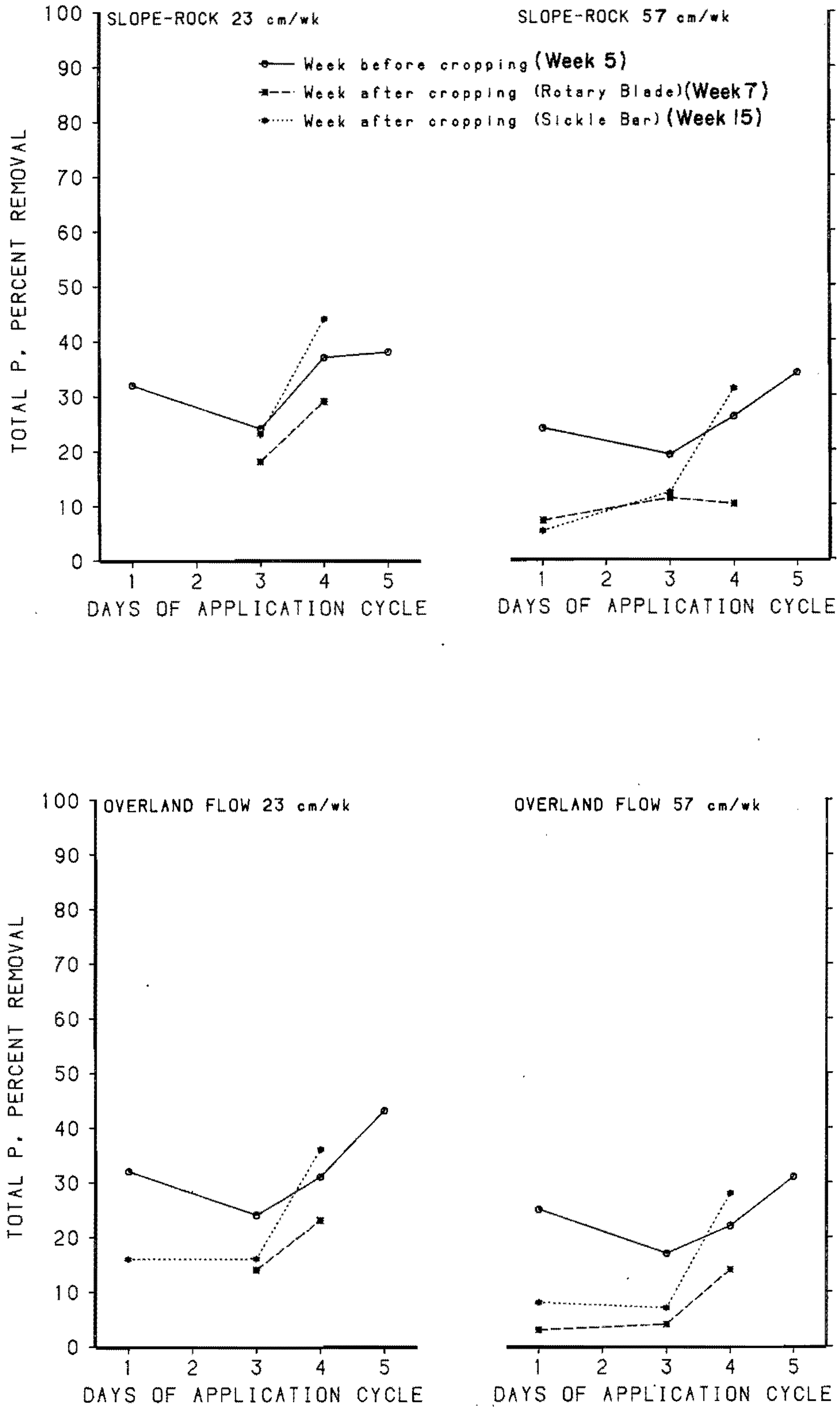


Figure 64. Total phosphorus removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

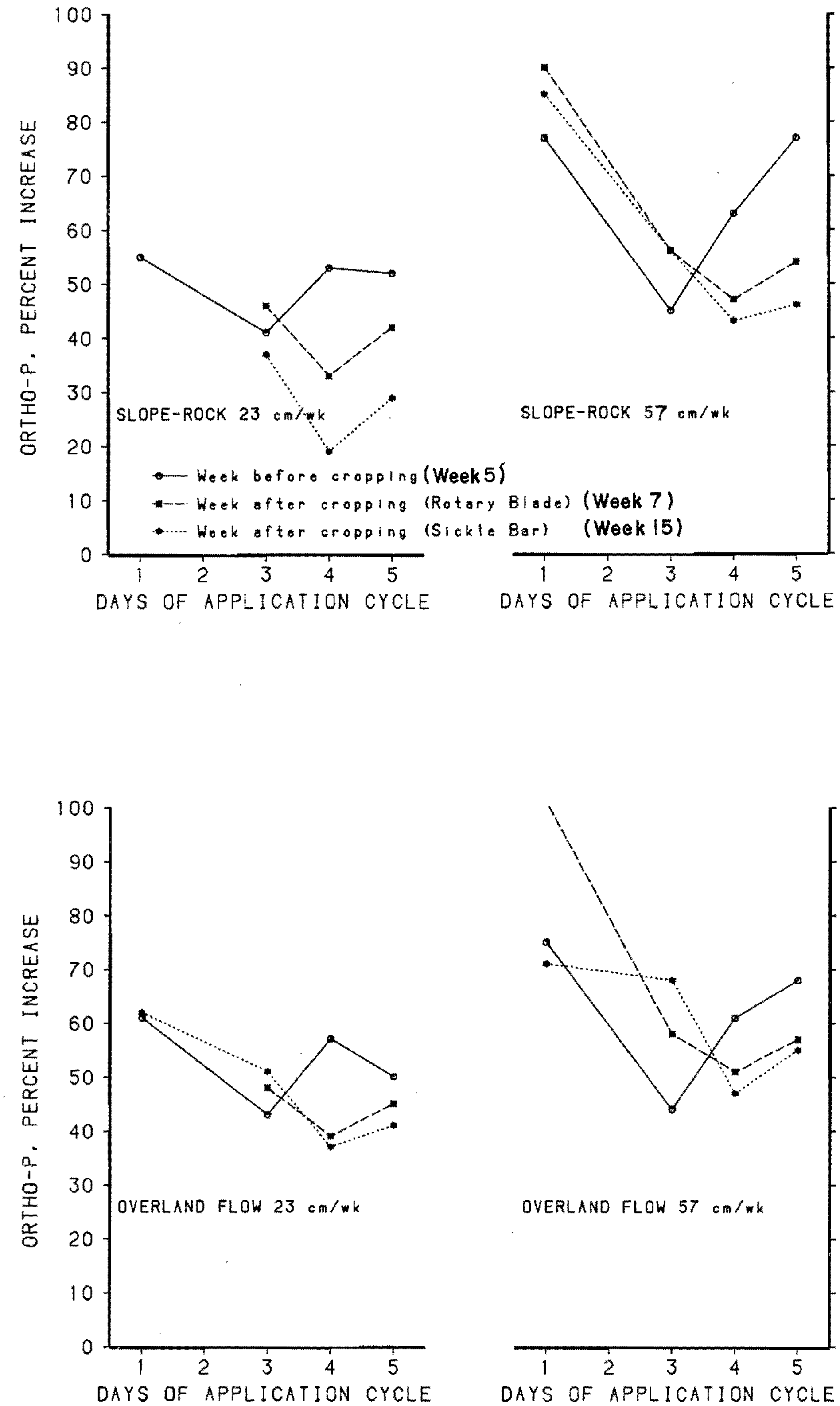


Figure 65. Orthophosphorus removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

removal) after wastewater application was resumed may have been caused by the lag in proliferation of Nitrobacter spp., which convert nitrite to nitrate after the Nitrosomonas spp. convert ammonia to nitrite.

During the first year, nitrite and nitrate concentrations were suppressed after cropping (Figure 68 and 69), with effluent values lower than influent levels. Nitrification processes and microorganisms were, therefore, affected by harvesting. The increased organic loading may have inhibited nitrification due to the heterotrophic organisms out competing the chemo-autotrophs (nitrifiers) for the micronutrients. The unconventional harvesting procedure used the first year of this study (i.e., not allowing the fields to dry prior to harvest and the lawnmower mulching the clippings as it cut the grass) made it difficult to remove the residue. Thus, these results may not be indicative of full scale systems.

Nitrite-nitrate (NO_2-N) removals before and after harvesting are shown in Figure 70 for the second year of operation. An initial increase in effluent NO_2-N concentration (-104 to -4 percent removal) lasted 1 or 2 days. Effluent NO_3-N levels, however, continued to exceed influent levels (Figure 71).

Nitrite oxidation is inhibited by free ammonia in liquid systems, particularly when the pH is alkaline. Soil adsorption of ammonium prevents this inhibition from developing in most circumstances (EPA 1977). The inhibitory effect of NH_3 on Nitrobacter spp. giving rise to NO_2-N accumulation in soils is evident primarily during the growing initiation phase (Morrill and Dawson 1967). Nitrite oxidation can occur

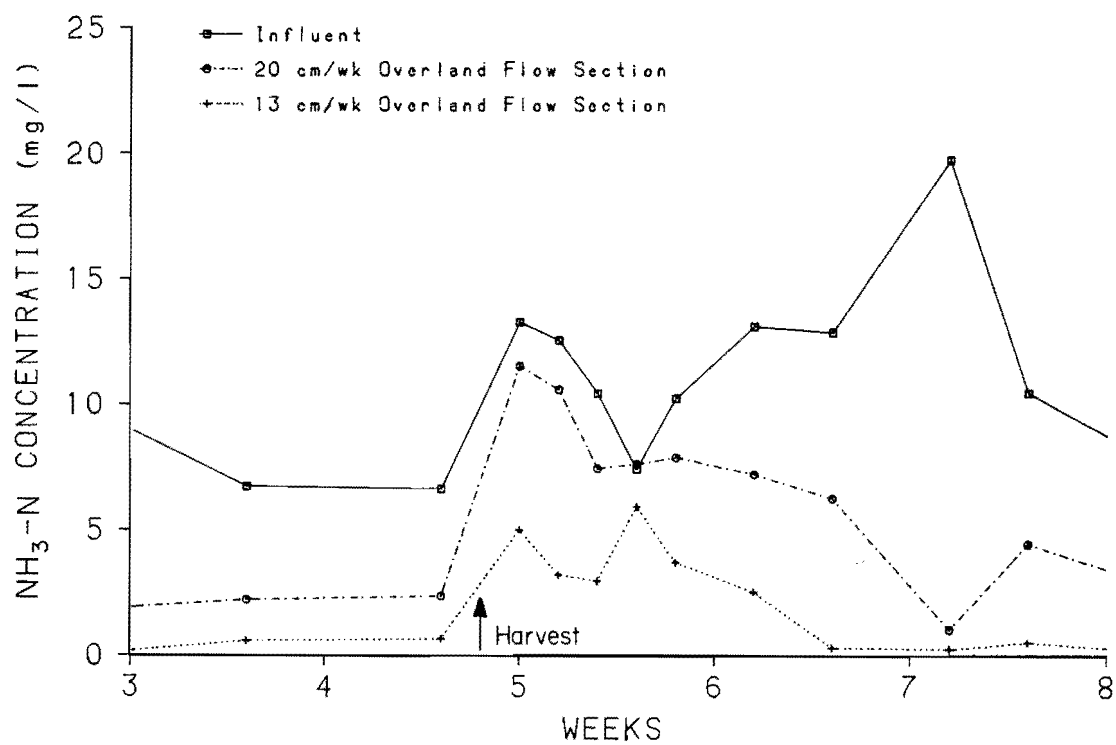


Figure 66. Influent and effluent ammonia-nitrogen concentrations following cropping of the overland flow sections the first year of operation (Zirschky 1980).

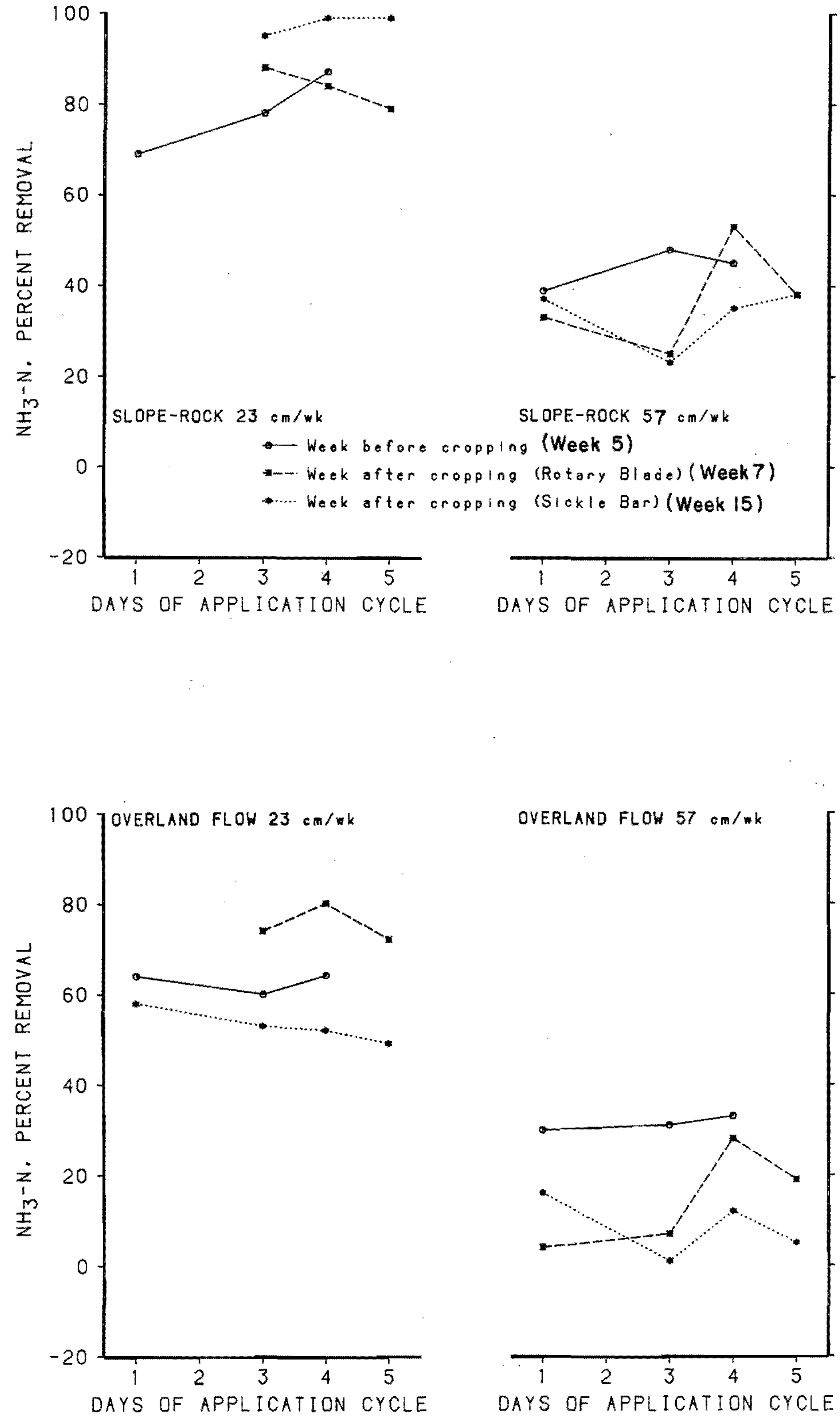


Figure 67. Ammonia-nitrogen removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.

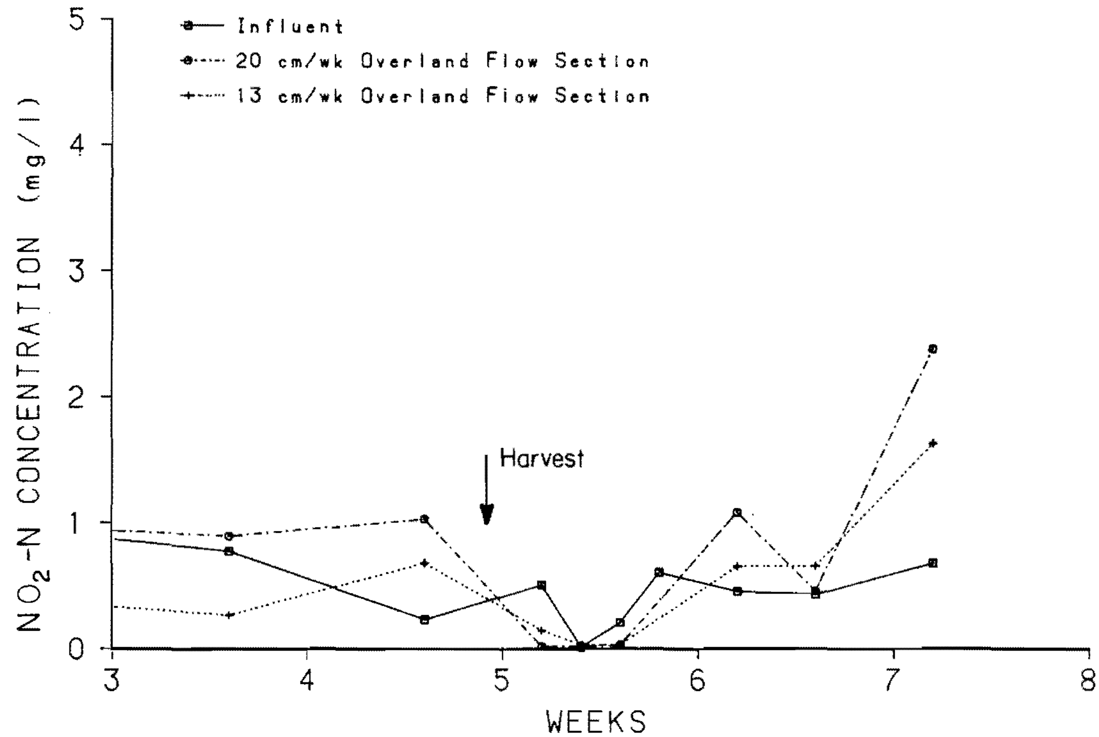


Figure 68. Influent and effluent nitrate-nitrogen concentrations following cropping of the overland flow sections the first year of operation.

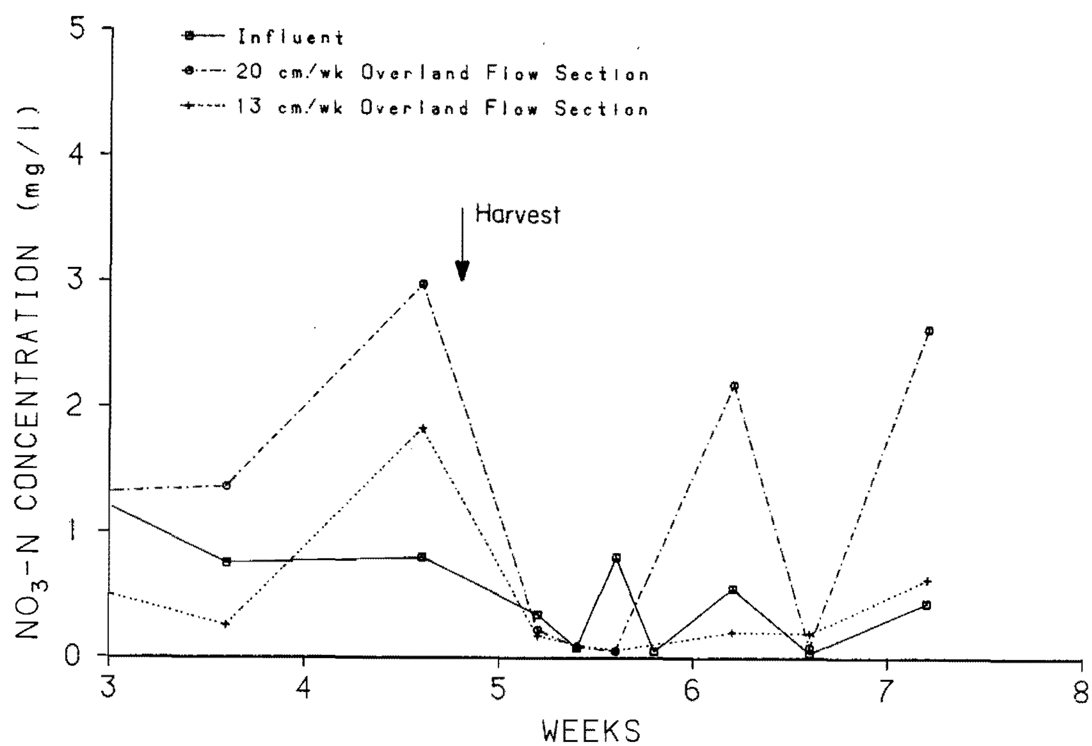
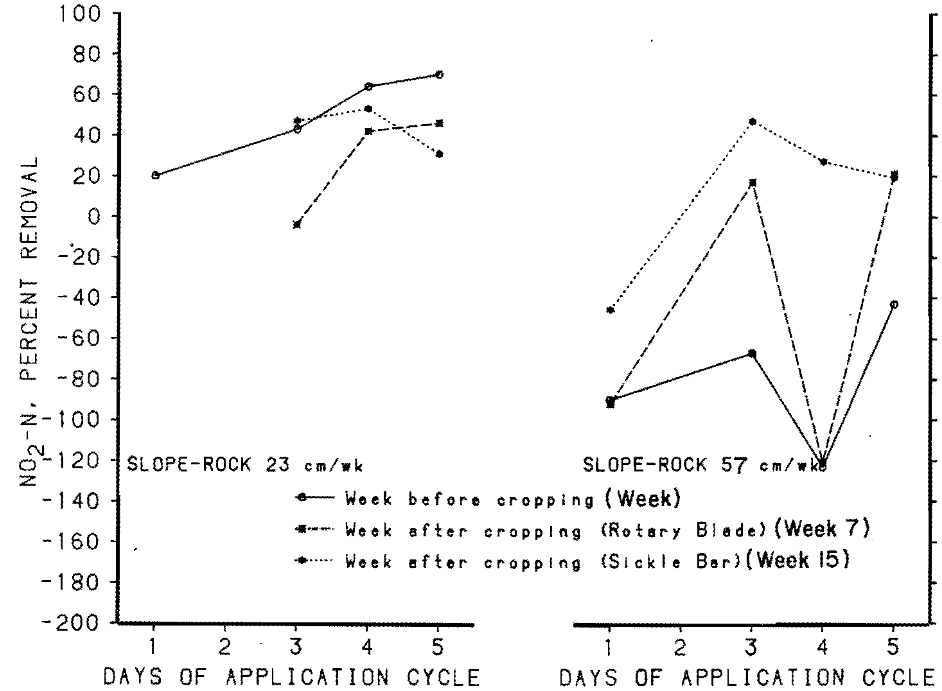
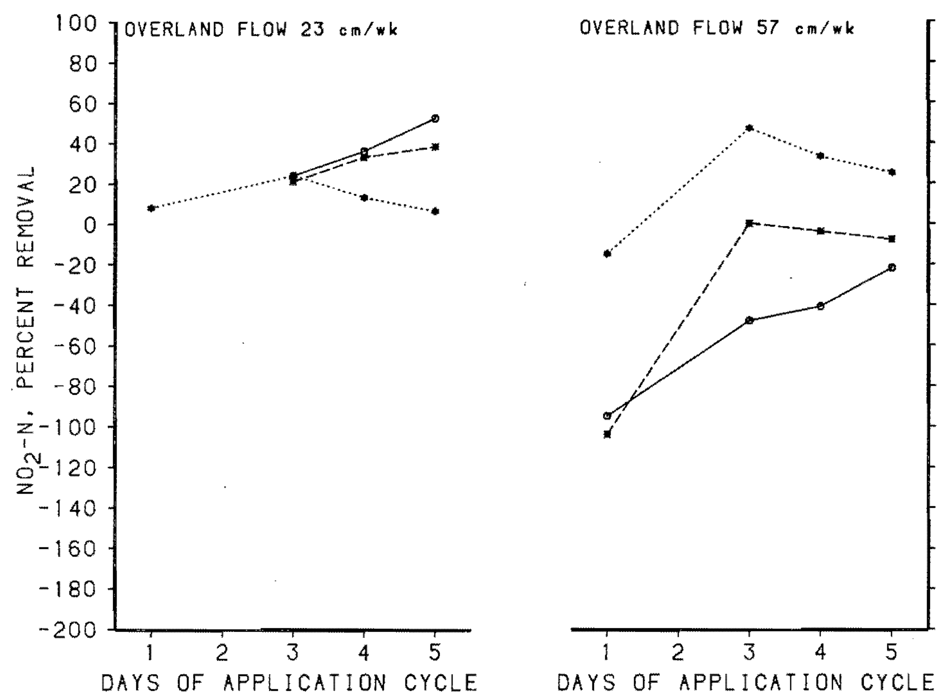


Figure 69. Influent and effluent nitrite-nitrogen concentrations following cropping of the overland flow sections the first year of operation.

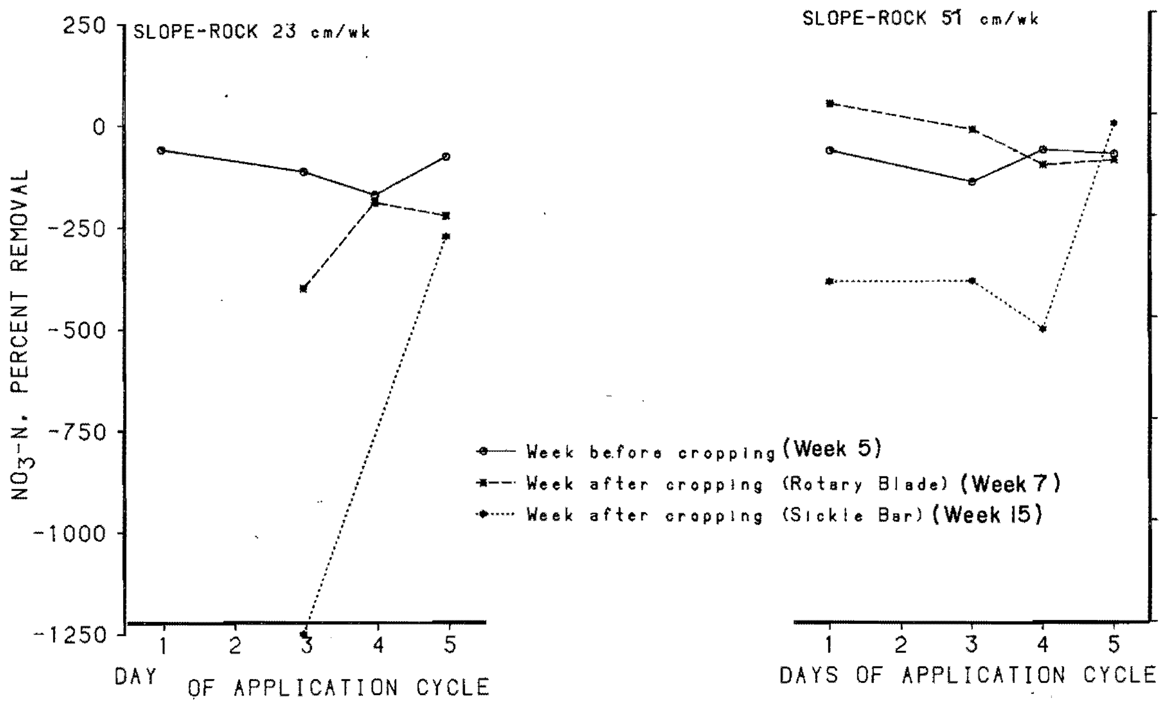


a) Slope rock

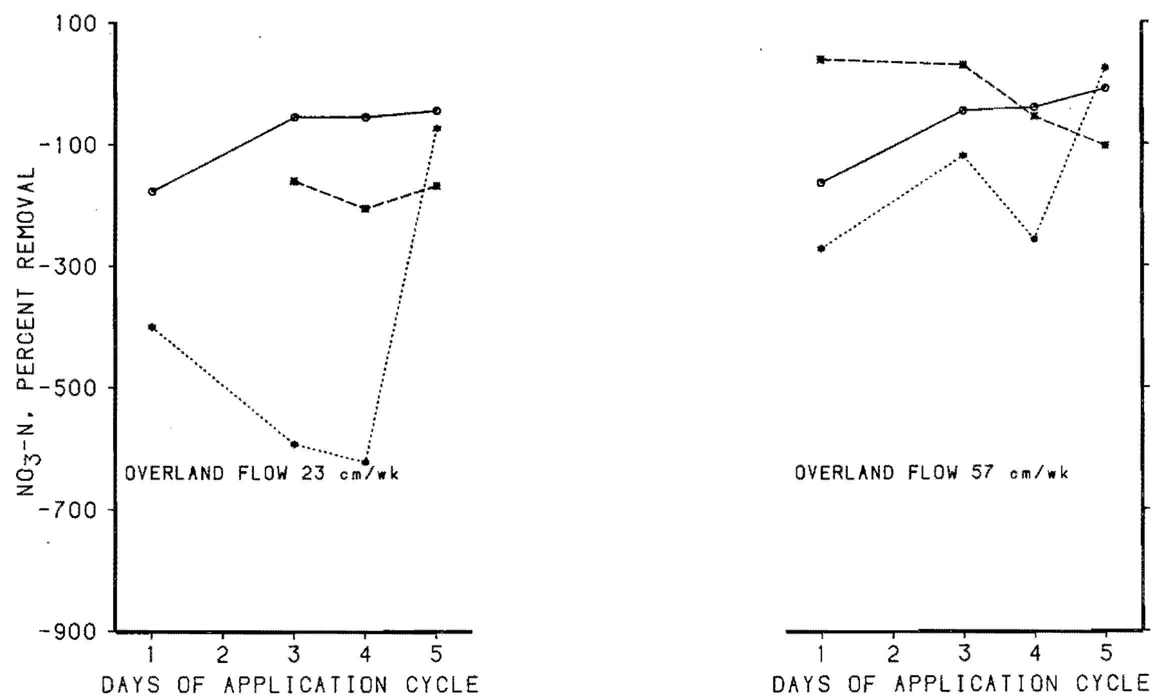


b) Overland flow

Figure 70. Nitrite-nitrogen removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for (a) slope-rock sections and (b) overland flow sections.



a) Slope rock



b) Overland flow

Figure 71. Nitrate-nitrogen removal efficiencies after harvesting the second year of operation with comparison to weekly startup without harvesting for a) slope rock sections and b) overland flow sections.

rapidly (in a few minutes), after *Nitrobacter* spp. have been allowed to regenerate (Morrill and Dawson 1967, Hoeppel et al. 1974). NO_2-N concentrations are rarely above 1.0 mg/l (Culp, undated).

The nitrate-nitrogen (NO_3-N) increases which occurred after cropping were consistent with the findings of Law et al. (1970), Bouwer (1973), and Hoeppel et al. (1974), who observed high nitrate in overland flow effluent due to the intermittent nature of nitrification during the drying cycle. Shutdown changes the oxygen status to inhibit denitrification, and substantial nitrate-nitrogen are present in the runoff when flooding is resumed (Law et al. 1970, Bouwer 1973, Thomas et al. 1976, and Gilbert et al. 1979). The delayed nitrate-nitrogen peak could be attributed to the migration patterns of the soluble nitrate salts formed. Nitrate-nitrogen was formed in the soil during aeration. The nitrate salts solubilized upon subsequent flooding and moved with the soil water. During the short rest periods, evaporation could have caused upward movement of the salts toward the soil surface (Thomas 1972). The salts may have been washed down the slope with the surface applied water.

The gradual increase in effluent NO_3-N concentrations coincided with increased ammonia and nitrate concentrations in the influent. Well-aerated wastewater and soil conditions which inhibited denitrification could have produced a high effluent nitrate concentration in the slope-rock 23 cm/wk after the seventh week of operation. The decrease and (at several points) removal of NO_3-N from the applied wastewater on the 57 cm/wk slopes can be attributed to denitrification. Soil saturation created reducing conditions which promoted the denitrification of NO_3 to nitrous oxide or N_2 gas. The ammonia reduction was greater on the treatment fields hydraulically loaded at 23 cm/wk than on the 57 cm/wk field. The thinner film of water passing over the slopes at the lower hydraulic loading allowed greater soil water contact and more oxygen diffusion conducive to nitrification.

Two-year summary

Cutting and harvesting the cover crop on an overland flow wastewater treatment site is an important component of system management. At least three times a year, the grass should be cut and removed from the fields. Harvesting requires drying to minimize damage to wet slopes from heavy cropping equipment and to allow moisture content of the crop to drop to where it will harvest freely. Further drying of the crop may be necessary after harvest unless silage is being produced.

Wastewater application must be scheduled to permit harvesting. Alternate disposal or storage areas must be available to accept the wastewater while fields dry. From the results of this study, increasing the loading rates on adjacent fields may "shock" the treatment fields sufficiently to degrade the runoff from these slopes. The rain of week 13 increased BOD_5 levels substantially. Increasing the application rate from 5 cm/wk to 20 cm/wk during the initial stages of system start-up also resulted in degradation of effluent quality. In contrast, Gilde et al. (1971) stated that overland flow systems maintain excellent effluent quality in spite of shock organic loads and long shutdowns. More research is needed.

A lawnmower was unsatisfactory for harvesting because it mulched the grass and made it impossible to remove all the grass cuttings. Harvesting equipment should minimize mulching effects. Using a thrasher and baler should maximize crop removal. Heavy machinery should be equipped with high flotation tires to prevent damage to the slope. The equipment should cut across the slope, rather than up and down, to check channeling if damage does occur. The most efficient mower used in this study was a tractor-driven sickle-bar cutter. A baling machine would work well with this type of cutter bar on a thrasher.

The degradation in treatment efficiency with harvesting depended on the degree of crop removal and the hydraulic loading rate. The treatment efficiency decreased after harvesting at high hydraulic loading rates. However, BOD_5 and SS (7-day average) effluent State of Utah discharge limits were not exceeded. System recovery to effluent levels achieved prior to harvest took only 2 days. The 20 cm/wk slopes took only 1 day to recover.

The grass should be dry and standing erect prior to harvesting to achieve the best results. The grass should be cut at regular intervals and removed to extend the life of the treatment fields.

Operational Difficulties

The major operating problems encountered in this study were head losses caused by large solids at pump intakes and flow valves, solids deposition on the gravel layer, and channeling and short-circuiting due to differential settling and compaction. Minor problems included mosquitoes, weeds, rainfall at harvest time, and freezing pipes at the end of the season.

The buildup of solids at the pump intake created large head losses. A screen and an oversized pump, which pumped wastewater to a constant head tank to be gravity fed to the test field, alleviated the problem during the first year. The screen was cleaned daily and the pump was cleaned fortnightly. A small submersible pump which pumped the wastewater directly to the top of the slopes was used the second year. To maintain relatively constant flows, a bar rack and three 4.8 cm (3/4 inch) mesh screens were placed before the pump. These screens were cleaned every 2 hours.

Clogging of the gate valves to each distribution trough occurred the first year, but the problem was remedied by reducing the level of wastewater in the constant head tank, which allowed the gate valves to remain open wider. Ball valves, combined with the higher flow rates used the second year of operation, completely eliminated the problem.

Solids deposited on the gravel layer would accumulate in the first 2 m to such an extent that ponding developed. The rocks in the first 2 m were rearranged to produce channels for the wastewater to overcome the head loss (Figure 14). The solids which deposited in the channel were organically removed once a week (Figure 25).

On the conventional overland flow slopes, the combination of high grass and solids deposition near the region of distribution created head losses at the high flow rates (41 and 57 cm/wk) which caused the wastewater to flow away from the slopes. Cutting and removing the tall grass remedied the problem.

Uneven settling and differential compaction around the perimeter of the test area resulted in uneven wastewater distribution on the outer slopes loaded at 41 cm/wk. Channeling and short-circuiting adversely affected the wastewater renovation capacity of the overland flow 41 cm/wk slope. Water losses off to the side of the slope-rock 41 cm/wk plot and down boundary stake holes resulted in a lower wastewater loading.

Mosquitoes and flies were observed near the influent troughs and amongst the grass. Stagnant water in nearby irrigation canals may have been the source of the mosquitoes. Water ponding on the test sections was not observed. The moist environment, tall grass, and organic matter on the test slopes, however, harbored the pests.

Weeds encroached on the gravel zone and proliferated on the berms dividing the plots. The gravel layer was periodically weeded. Poison hemlock was among the weeds which would make the crop nonpalatable.

CONCLUSIONS

The scope of this research was to use a small-scale overland flow treatment system to evaluate the effectiveness of adding a gravel wastewater distribution and treatment zone at the top of a conventional overland flow slope. The test area, 24 m wide by 36 m long (80 ft by 120 ft), was divided into six plots. A 9 m long by 7.6 cm deep gravel layer was placed at the top of three of the treatment slopes (slope-rock plots). Raw (screened, degritted) wastewater from Hyrum, Utah, was applied to the experimental slope-rock sections and to the conventional overland flow slopes at application rates of 13 and 20 cm/wk during the first year of the study and 23, 41, and 57 cm/wk the second year. The conclusions were:

System Hydrology

1. The gravel layer, placed at the top of the treatment slopes, increased infiltration and, therefore, decreased the amount of wastewater recovered. Forty-one to 78 percent of the applied raw wastewater was recovered compared with 61 to 87 percent recovered from conventional overland flow slopes.

2. The gravel layer increased wastewater detention time on all but the slope loaded at 13 cm/wk. Detention times on the slope-rock treatment slopes were 13 to 38 minutes longer than the detention times on the conventional overland flow slopes.

3. Mean detention time was inversely correlated to flow rate for both slope-rock and overland flow systems with correlation coefficients (r2) of 0.56 to 0.69.

4. Hydraulic irregularities cause channeling and short circuiting and shorten wastewater detention time (treatment time) on the slope. Channeling and short circuiting resulted in the 41 cm/wk overland flow slope exhibiting the shortest mean detention time (89 minutes). Detention times on the other slopes ranged from 98 minutes on the 57 cm/wk overland flow slope to 154 minutes on the 13 cm/wk overland flow slope.

System Performance

5. In general, the presence of a rock layer had no significant effect on overland flow effluent quality.

6. BOD_5 removals for the test sections averaged 87 to 93 percent. BOD_5 effluent averages ranged from 6 to 12 mg/l. BOD_5 removal was not significantly affected by application rate (from 13 to 57 cm/wk).

7. Effluent suspended solids values ranged from 5 to 9 mg/l with 91 to 95 percent removals. Suspended solids removal was not significantly affected by application rate.

8. The treatment slopes with the highest application rate (57 cm/wk) achieved mean effluent concentrations meeting the 15 mg/l BOD_5 and 10 mg/l SS (30-day average) Utah State 1985 discharge limits.

9. Total phosphorus removals were only 20 to 33 percent.

10. On a mass basis, greater phosphorus reductions (39 to 72 percent) were obtained by the slope-rock sections than by the overland flow slopes (35-59 percent).

11. Orthophosphate levels increased (17 to 46 percent) on all treatment slopes as conversion of phosphorus forms and leaching occurred on the slopes.

12. Ammonia reductions were significantly decreased at 57 cm/wk application rates. The percent removal dropped from 69 to 93 percent at the 13, 20, and 23 cm/wk application rates to 33 to 43 percent at the 57 cm/wk loading rate.

13. Ammonia removal was a function of application rate, effluent flow rate, and mean detention time for both the slope-rock and overland flow treatment sections.

14. Nitrification occurred at all application rates from 13 to 57 cm/wk, as evidenced by the increase in nitrite- and nitrate-nitrogen species in the effluent (85 to 747 percent increase).

15. Ammonia-nitrogen was more effectively removed than nitrate-nitrogen.

16. Total nitrogen removals ranged from 49 to 79 percent on the 13, 20, and 23 cm/wk slopes but dropped to 36 to 38 percent on the 57 cm/wk treatment slopes.

17. On a mass basis, greater nitrogen reductions (50 to 87 percent) were obtained by slope-rock sections than by overland flow slopes (44 to 79 percent).

18. Total nitrogen removal was linearly correlated with application rate and effluent flow rate with correlation coefficients ($r2$) of 0.61 to 0.79 respectively for both slope-rock and overland flow slopes. Total nitrogen removal versus application rate for the overland flow slopes yielded a higher r^2 equal 0.92 when a hyperbolic function was used.

19. Fecal coliform removal efficiencies were high, ranging from 88 to 99 percent. However, the effluent contained more than 10^4 organisms/100 ml, which amount would necessitate disinfection before discharge.

20. Harvesting should be done by cropping the grasses in long pieces and completely removing them from the slopes. The 1985 Utah State BOD_5 and SS effluent discharge limits were not exceeded on a weekly average due to harvesting operations. However, effluent values after harvesting did exceed 20 mg/l BOD_5 and 12 mg/l SS for one day at the higher (57 cm/wk) loading rate.

21. Mulching the grass clippings and failing to remove the residues from the slopes resulted in the deterioration of effluent quality. BOD_5 effluent values (118 mg/l) exceeded influent levels (82 mg/l). Both BOD_5 (20 mg/l, 7-day average) and SS (12 mg/l, 7-day average) limits were exceeded by the conventional overland flow treatment slopes. Only the SS effluent limit was exceeded by the slope-rock treatment.

System Age

22. Commencing overland flow wastewater application in the spring resulted in, at most, a 7-week (48 days) acclimation period before effluent quality stabilized.

ENGINEERING SIGNIFICANCE

Based on the operating experience and results obtained during this study, placement of a gravel layer at the upstream end of a conventional overland flow treatment slope did not significantly improve effluent quality. However, the gravel increased wastewater detention times and infiltration. Gravel layers could be used intermittently down the slope to increase detention times. Also, gravel could be used instead of aluminum baffles or lateral troughs to distribute flow more evenly in trouble spots. If the gravel zone is located near the influent distribution area, primary pretreatment necessary because of solids accumulation.

Under the operating conditions of this study, the treatment slopes were able to obtain excellent BOD_5 and SS effluent quality even at the highest hydraulic loading rate of 57 cm/wk. When effluent nitrogen concentrations are not important, land requirements for overland flow could be reduced by half.

Preparation and maintenance of plot surfaces are most important. Grading irregularities, differential compaction, and incomplete sealing should be avoided. A dense sod-forming grass cover should be established before wastewater application begins to prevent erosion. Favorable local conditions included a healthy, dense, sod-forming vegetative cover (mixture of rye, fescue, and bluegrass), high clay content subsoil and clay loam topsoil, a semiarid growing season, and smooth topography on most of the slopes.

A water application schedule of 8 hours on and 16 hours off for 5 days cycled with 2-day rest periods was effective in the semiarid climate and operating conditions of this study. Nitrogen removal proved to be the factor controlling loading rates. Hydraulic loading rates as high as 57 cm/wk can be used if nitrogen removal is not an important consideration. Otherwise, application rates up to 23 cm/wk are recommended.

The overland flow treatment system presented in Figure 72 can potentially meet 1985 Utah State effluent discharge limits, treating screened wastewater at hydraulic loading rates of 13 to 57 cm/wk. To minimze clogging and aerosols, troughs or gated pipes should be used to distribute the wastewater. A minimum of 39 cm of a clay loam topsoil over a highly impermeable (high clay content subsoil) is recommended.

A mixture of rye grass, fescue grass, and bluegrass should be sown for high density growth. Effluent can be collected in clay lined or gravel lined ditches. An option to recycle should be provided for system startup and acclimation and any other event which causes treatment efficiency to be less than desired.

Grass should not be allowed to grow higher than 1 m, especially near the influent troughs. Proper harvesting operations, if done properly, will not substantially deteriorate effluent quality. Grass should be cut by a sickle type mower, and the clippings completely removed from the fields.

Pilot-scale research may not reveal the full treatment capacity of a proposed overland flow treatment process because treatment efficiency improves as the application season progresses. Pilot-scale research is necessary, however, because treatment efficiency, in terms of BOD_5 and SS, has been shown to be site specific. Through field investigations, it was concluded that overland flow, under the operating conditions of this experiment, can achieve BOD_5 and SS effluent concentrations below 15 mg/l BOD_5 and 10 mg/l SS at hydraulic loading rates up to 57 cm/wk. Hydraulic loading rate has no appreciable effect on effluent BOD_5 and SS concentration. Nitrogen reductions, however, decreased as application rate increased. If nitrogen concentrations in the effluent were not of local concern, overland flow land requirements could be substantially reduced, thus reducing the capital cost significantly. Operating under the design parameters of this study at a hydraulic loading rate of 57 cm/wk, 3.4 hectares (8.3 acres) of land would be necessary to satisfactorily treat a 1-mgd raw sewage influent. Approximately 14.6 hectares (36 acres) would be required for a 1-mgd facility at a 13 cm/wk application rate. Therefore, approximately four times more land would be required at the 13 cm/wk loading rate than at the higher rate of 57 cm/wk.

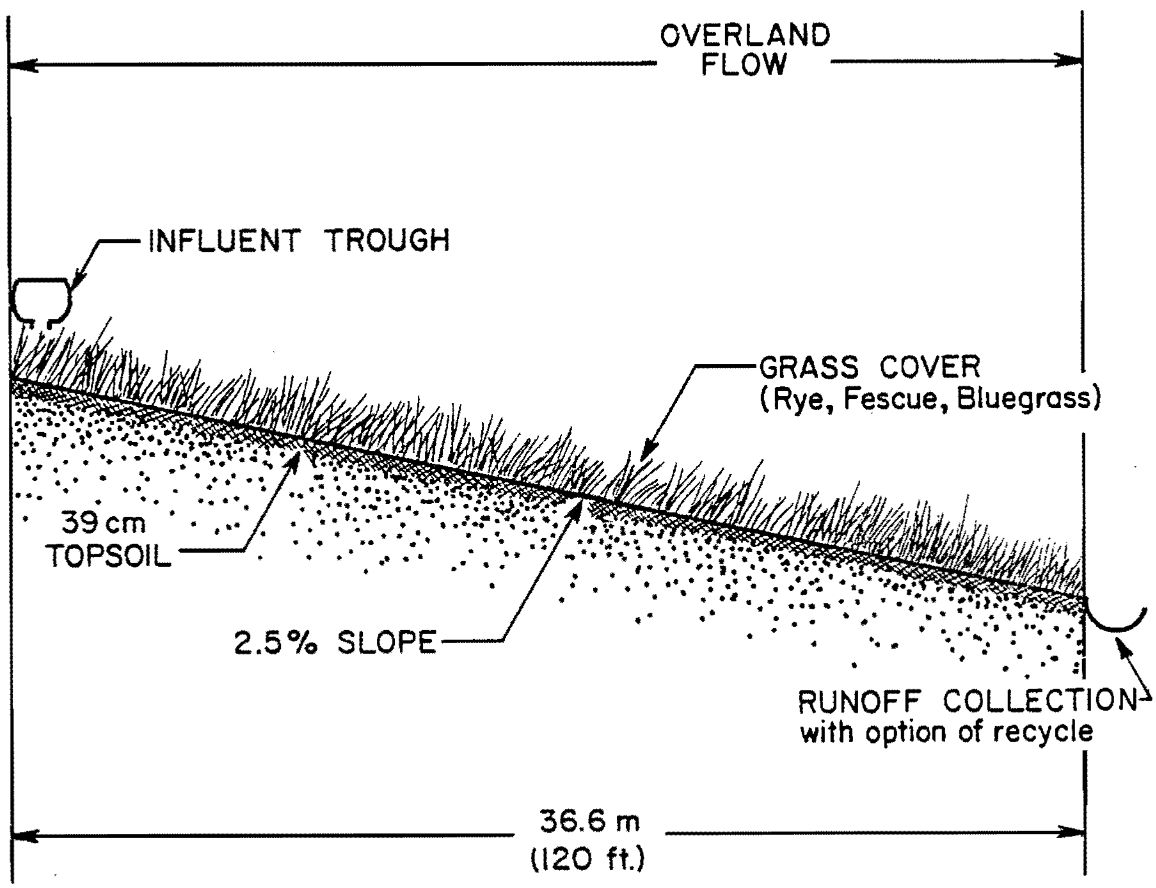


Figure 72. Schematic diagram of proposed overland flow treatment slope.

RECOMMENDATIONS FOR FUTURE RESEARCH

Overland flow treatment of municipal wastewater is in its developmental stages. Limited information about design and operation is available. Research is needed to support the conclusion that overland flow is a viable wastewater treatment alternative. Recommendations for future research are:

1. A study to determine the effects of temporarily increasing hydraulic loading rates on effluent quality from overland flow wastewater treatment slopes.

2. Further research at high hydraulic application rates to determine maximum loading limits.

3. Research at high hydraulic application rates at other locations to test the validity of the results obtained from this study.

4. Evaluation of the use of gravel zones at other locations on the overland flow slope to increase wastewater detention time and/or distribute the flow more evenly in trouble spots.

5. A study of ammonia removal as a function of distance down the treatment slope.

6. Further evaluation of the effects of harvesting.

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