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Phosphorus Loadings to the Groundwater from

Suburban Land Uses: A Preliminary Analysis 41 pp

Gold & Loomis (URI)

Narragansett Bay Estuary Program

Current Report

The Narragansett Bay Project

PHOSPHORUS LOADINGS TO THE GROUNDWATER FROM SUBURBAN LAND USES: A PRELIMINARY ANALYSIS

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FOREWORD

The United States Congress created the National Estuary Program in 1984, citing its concern for the "health and ecological integrity" of the nation's estuaries and estuarine resources. Narragansett Bay was selected for inclusion in the National Estuary Program in 1984 and designated an "estuary of national significance" in 1988. The Narragansett Bay Project (NBP) was established in 1985. Under the joint sponsorship of the U.S. Environmental Protection Agency and the Rhode Island Department of Environmental Management, the NBP's mandate is to direct a five-year program of research and planning focussed on managing Narragansett Bay and its resources for future generations. The NBP will develop a comprehensive management plan by December, 1990, which will recommend actions to improve and protect the Bay and its natural resources.

The NBP has established the following seven priority issues for Narragansett Bay:

* management of fisheries
* nutrients and potential for eutrophication
* impacts of toxic contaminants
* health and abundance of living resources
* health risk to consumers of contaminated seafood
* land-based impacts on water quality
* recreational uses

The NBP is taking an ecosystem/watershed approach to address these problems and has funded research that will help to improve our understanding of various aspects of these priority problems. The Project is also working to expand and coordinate existing programs among state agencies, governmental institutions, and academic researchers in order to apply research findings to the practical needs of managing the Bay and improving the environmental quality of its watershed.

This report represents the technical results of an investigation performed for the Narragansett Bay Project. The information in this document has been funded wholly or in part by the United States Environmental Protection Agency under assistance agreement #CX812680 to the Rhode Island Department of Environmental Management. It has been subject to the Agency's and the Narragansett Bay Project's peer and administrative review and has been accepted for publication as a technical report by the Management Committee of the Narragansett Bay Project. The results and conclusions contained herein are those of the author(s), and do not necessarily represent the views or recommendations of the NBP. Final recommendations for management actions will be based upon the results of this and other investigations.

This report is an interim report and should not be considered a comprehensive synthesis of the existing data on this subject. The interested reader is encouraged to investigate additional sources of information.

PHOSPHORUS LOADINGS TO THE GROUNDWATER FROM SUBURBAN LAND USES: A PRELIMINARY ANALYSIS

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ABSTRACT

Widespread suburbanization within Rhode Island has increased the use of individual sewage disposal systems (ISDS), and the acreage of manicured homelawns. Septic systems and homelawns receive considerable inputs of nitrogen and phosphorus. These nutrients have been shown to degrade surface water quality. The objective of this study was to conduct preliminary investigations on phosphorus (P) losses to the groundwater from ISDS and homelawns.

Phosphorus leaching to groundwater from both fertilized (41 kgP/hectare/yr) and unfertilized lawns appeared to be negligible and may be related to soil pH. Marked P reductions were observed in the septic systems studied. Phosphorus concentrations averaged 0.22 mg/l in effluent one meter below a relatively new conventional soil absorption trench. Acid soil conditions ($pH \leq 5.5$) may be responsible for the formation of relatively insoluble phosphate precipitates, removing phosphorus from percolating groundwater. With increasing use, this phosphorus removal capacity could dissipate.

The results of this study reflect P movement in groundwater on coarse-textured outwash derived soils typical of the western shore of the Narragansett Bay. Hydrologic soil groups, which can effect the mode of pollutant transport from a landscape, differ markedly from each side of

the Bay. Offsite transport of P could be dramatically different on the eastern shore and islands of the Bay, due to differences in soil type and runoff patterns.

INTRODUCTION

Nitrogen and phosphorus have been shown to be limiting nutrients for accelerated eutrophication of surface waters. While brackish waters have been shown to be nitrogen limited, upper reaches of estuaries such as Chesapeake Bay are considered phosphorus limited. To focus pollutant abatement programs for the Narragansett Bay it is necessary to quantify the losses of both nutrients from different sources in the Narragansett Bay watershed.

Nutrient losses are quite site and land use specific. Losses depend largely on nutrient loading, rates of transformation, attenuation mechanisms, and the hydrologic pathways for off-site transport. Generally, overland flow has been shown to contain relatively low concentrations of NO_3^--N, but high concentrations of sediment-bound nitrogen and phosphorus. Groundwater, however, often contains high concentrations of soluble nutrients, particularly NO_3^--N.

Suburbanization is rapidly occurring throughout Rhode Island and most of this new development will utilize individual sewage disposal systems (ISDS) for waste disposal and will replace existing vegetation with home lawns. The objective of this study was to conduct preliminary investigations on phosphorus losses to the groundwater from homelawns and ISDS.

METHODS

Water-borne losses from homelawns and on-site sewage disposal systems (ISDS) were monitored by the Natural Resources Science Department at URI. Field laboratories were constructed and instrumented to permit controlled sampling in the vadose zone from hydrologically isolated field plots. The field plots were located on a coarse-textured Merrimac sandy loam soil, of glacial outwash origin. These types of soils are found in the coastal pond watersheds and predominate along the western shore of Narragansett Bay.

On the ISDS study three types of treatment systems were monitored:

a) Leachate from a conventional septic tank soil absorption trench

b) Effluent from a RUCK sand filter

c) Effluent from a recirculating sand filter

The ISDS Field laboratory began operation in June, 1986. Three replicates of each system were monitored at six week intervals from September 1986 to September 1987. A description of the experimental design is given in Appendix B (Lamb et al., 1988). Due to the limited operation time the results from this study may not reflect long-term removal rates.

Sand filter effluent samples were collected from access ports located immediately after the sand filters. On the conventional systems soil water leachate was obtained from suction lysimeters placed 0.9m beneath each of the soil absorption trenches (Figure 1). Rhode Island ISDS regulations require a separation of 0.9m between trench bottom and the saturated zone. Phosphorus analysis of the soil water percolate from

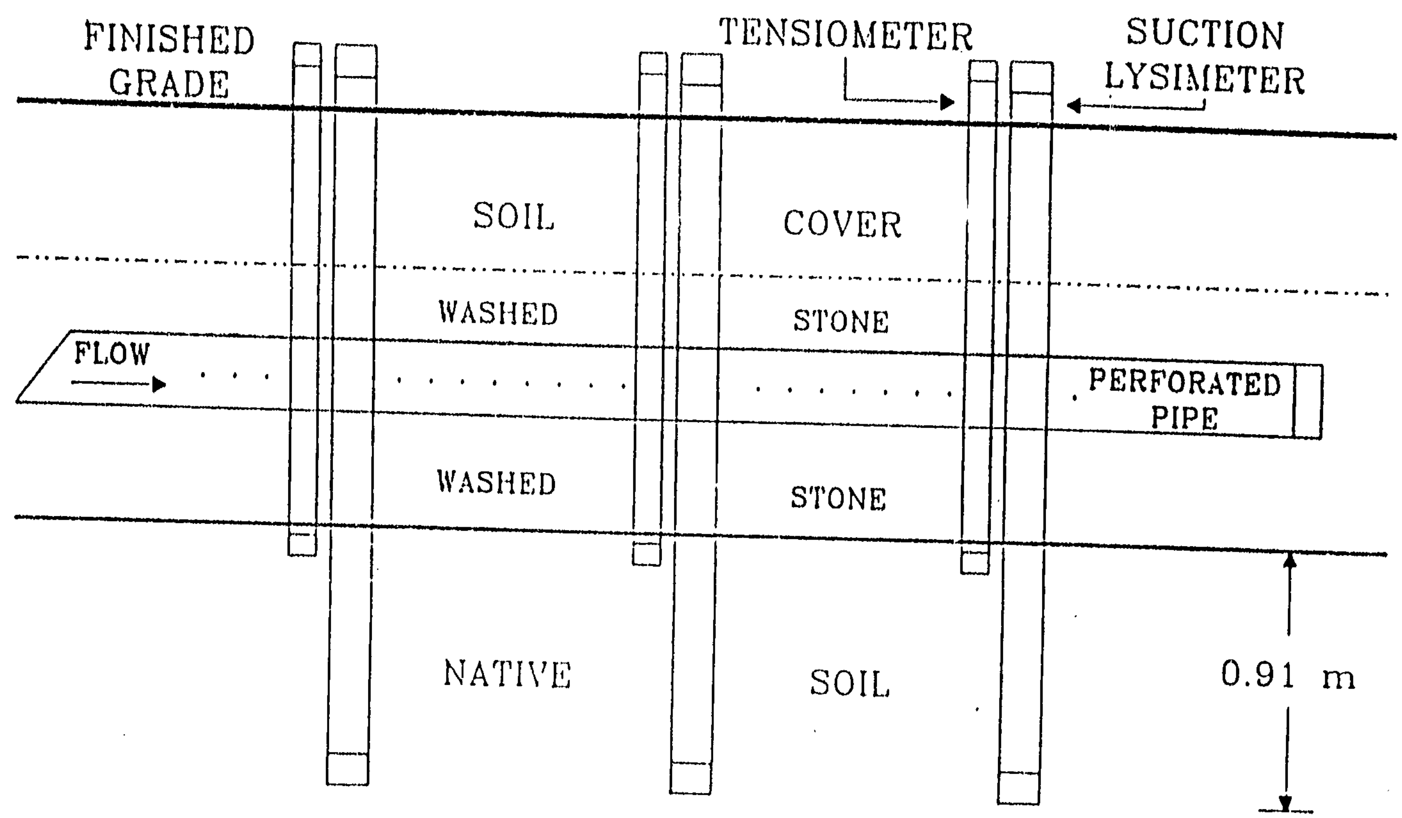

Figure 1. Cross-section of a soil absorption trench at the ISDS site showing suction lysimeter placement.

these lysimeters could indicate the potential for groundwater loading from properly constructed ISDS located on outwash derived soils.

Although the RUCK and recirculating sand filter systems are specifically designed to produce an aerobic environment for nitrification (Appendix B), they also have been suggested to enhance phosphorus and microbial treatment. Phosphorus analysis was performed on effluent from RUCK sand filters, recirculating sand filters and on septic tank effluent to observe the attenuation provided by the filters. Lysimeter samples, sand filter effluent and septic tank effluent were analyzed for orthophosphate with the ascorbic acid method (USEPA, 1979).

On the homelawn plots, suction lysimeter plates (30 cm in diameter) were installed 20 cm below the soil surface for leachate collection and to quantify percolate flux (Figure 2). The homelawn plots were established in 1983. Samples were obtained from irrigation events which generated leaching three times per week from the root zone for the 1986 growing season and should reflect potential P leaching from mature lawns. Fertilizer was applied at a rate of 41 kgP/ha/yr. Formulations and application methods were chosen to simulate the methods of commercial lawn care companies. Specifics on irrigation events, fertilizer treatments, formulations, and schedules are detailed in Appendix C (Morton et al., 1988). A hydrologic budget (Table 1 of Appendix C) was developed to predict percolation losses to the groundwater. Orthophosphorus analysis was performed on samples from these leachate events using the ascorbic acid method (USEPA, 1979). No mean separation statistical tests were performed between treatments, due to the preliminary nature of the results.

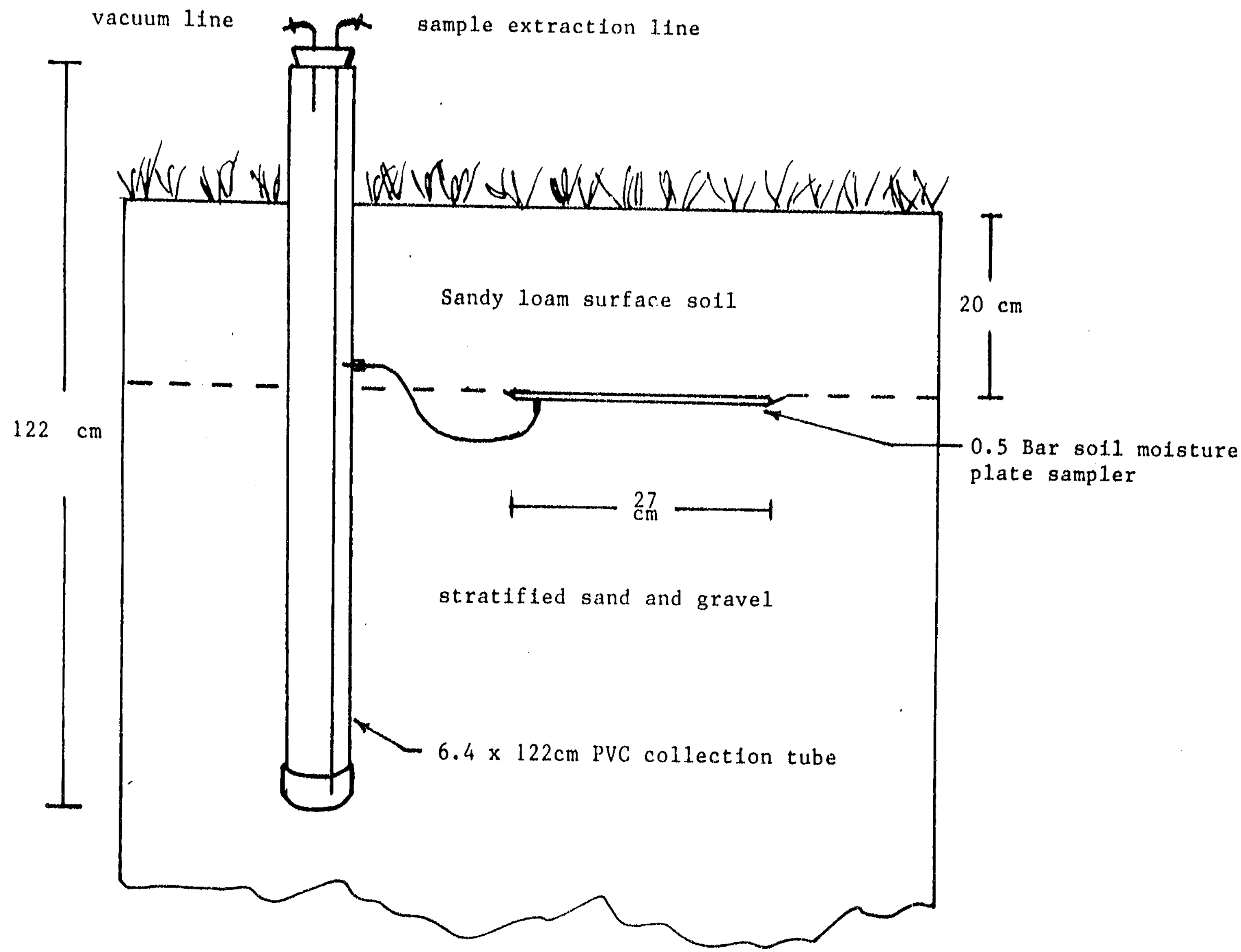

Figure 2. Cross-section showing suction plate lysimeter placement at the homelawn research site.

RESULTS AND DISCUSSION

Phosphorus Movement from Homelawns

Our preliminary analyses showed that phosphorus movement to the groundwater from both fertilized and unfertilized (control) lawns appeared to be negligible. Concentrations of phosphorus from control turf plots, which did not receive fertilizer applications, actually exceeded P concentrations from the high fertility plots (Table 1). The volume of water percolating through the root zone at the turf site was approximately 6,200 m^3/ha/yr. Using the mean soil water P concentration of 0.04 mg/l the estimated annual losses from the fertilized plots was 0.25 kg/hectare. This constitutes a net P loss of less than one percent. The amount of phosphorus lost from control plots was 1.3 kg/hectare and may represent leaching of in situ soluble phosphorus from the soil solum.

Phosphorus movement may be directly related to soil pH (Table 1). In acidic soils, phosphorus is expected to precipitate and be adsorbed to soil minerals by the formation of relatively insoluble phosphates of iron, aluminum and manganese (Figure 3). This fixation process occurs quite readily at soil pH 5.5 and below (Donahue, et al. 1983). Conversely, maximum solubility occurs at a pH of 6-7 for mineral soils (Brady, 1974).

Because of fertilizer application of urea and the subsequent nitrification process, soil pH in the high fertility plots was lower than the control plots, which have not received fertilizer inputs for several years. The nitrification process is summarized by the following equation:

$$NH_4^+ + 2O_2 \xrightarrow[\text{Nitrobacter Bacteria}]{\text{Nitrosomonas}} NO_3^- + 2H^+ + H_2O + \text{Energy}$$

Table 1. Soil pH and phosphorus levels beneath high fertility and unfertilized (control) turf plots.

PHOSPHORUS CONCENTRATIONS AT TURF PLOTS

PARAMETER	PLOT LOCATION	
	HIGH FERTILITY	CONTROL
Mean (mg P/l)	0.04	0.21
Range (mg P/l)	ND* - 0.29	0.07 - 0.38
No. of Samples (n)	23	23
Std. Deviation	0.07	0.09
Std. Error	0.01	0.02
Mean Soil pH	5.50 (n=3)	5.95 (n=3)

ND = No Detection below 0.01 mg/l

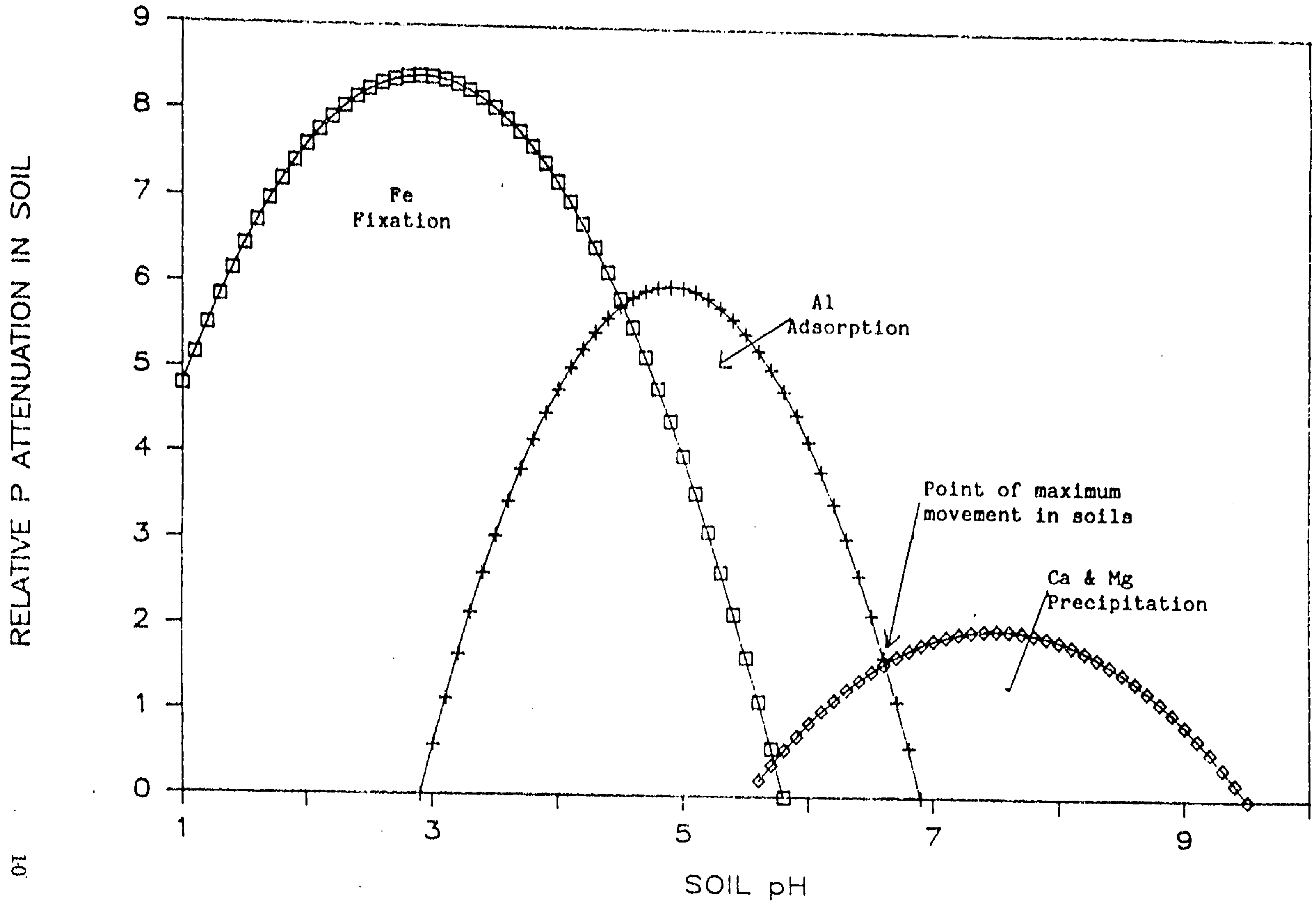

Figure 3. Relative phosphorus attenuation as a function of soil pH.
(Modified from Donahue et al., 1983.)

The oxidation of each NH_4^+ ion to NO_3^- releases $2H^+$, an important step in acidification of soils. The investigation of the chemical dynamics influencing P adsorption in the soil can warrant considerable attention and experimental effort beyond the scope of this study. However, after the results of our study were obtained we took several soil pH measurements to determine if pH might differ between treatments. Soil pH was approximately one-half pH unit higher in the control plots, which had not received any urea applications. This elevated soil pH level may cause additional soluble phosphorus to be present and leachable. Liming, a common recommendation in lawn care would raise the pH of a fertilized lawn and could be expected to increase P losses from the root zone of the acidic soils of Rhode Island. Although the underlying soil would be capable of absorbing much of the soluble P in the leachate, all soils have a finite phosphorus attenuation capacity and once reached, phosphorus breakthrough could occur.

Phosphorus - ISDS

Marked reductions in phosphorus concentrations were observed in the septic systems studied. Mean septic tank effluent P concentration was 4.18 mg/l while mean soil water percolate from the conventional septic system trench was 0.22 mg P/l (Table 2). This represents phosphorus removals of approximately 95% through one meter of unsaturated outwash sand and gravel soil material. This removal rate represents a new system (operating less than 2 years) and should not be equated with long term performance. Nagpal (1985) reported similar results in a soil column study comparing the differences between effluent P sorption in partially saturated and unsaturated gravelly silt loam soils. After 2.7 years of effluent loading, about 60 to 90 percent of the added phosphorus was being removed by the partially saturated and unsaturated soil columns, respectively.

Table 3 is a summary of other studies of P movement in groundwater from ISDS. Most of these studies had comparable results to this study; however the studies listed in Table 3 give P concentrations in the groundwater at varying distances from the leachfield and could include the effects of dilution and additional attenuation.

The phosphorus removal efficiency of the recirculating sand filter proved to be greater than that of the RUCK passive sand filter. Mean phosphorus removals in the sand filters were 24% for the RUCK sand filter and 75% for the recirculating sand filter. The RUCK sand filter, by design, is a single pass filter. Effluent moves rapidly through the filter limiting contact time for phosphate fixation. Conversely, in the recirculating sand filter, the effluent circulates through the filter several times per day before discharge. Retention time in the recirculating sand filter system is, therefore, several times greater than in the RUCK filter.

The results of this study suggest the importance of retention or contact time in phosphorus removal. This is supported by Nagpal (1985) who found that shorter contact time between effluent and soil attributed to poor sorption and rapid movement of phosphorus through soil. The native undisturbed soil was effective in removing P, while the sand filters which consist of coarse sands and gravels displayed less P removal. Continued use of the conventional system could exhaust the P adsorption capacity. More research on older systems and on systems in different soils might yield results that show more P movement from ISDS.

Table 2. Phosphorus concentrations at various locations throughout the ISDS site.

PARAMETER	LOCATION			
	STE	RUCK SF	Recirc. SF	Conv. Lys.
Mean (mg P/l)	4.18	3.18	1.02	0.22
Range (mg P/l)	3.26-6.10	1.15-5.00	0.16-2.68	ND*-2.61
No. of Samples (n)	9	28	28	26
Std. Error	0.33	0.20	0.12	0.12
Mean Effluent pH	7.3	4.3	4.4	4.0

* ND = No Detection below 0.01 mg/l
STE = Septic tank effluent
SF = Sand filter
Conv. Lys. = Conventional system lysimeter, sample obtained from soil water 0.9 meters below the trench bottom of a conventional system.

Table 3. Phosphorus movement below septic tank-soil absorption trench systems. (Modified from Brown, 1980)

Soil Texture	System age (yr)	Concentration P in effluent (mg/l)	Concentration P in groundwater (mg/l)	Depth to water table (m)	Horizontal distance moved (m)	Reference†
Sand	15	-	0.099	1.5-1.8	100	(a)
Loamy sand	8	11.5	11.6	0.9-1.2	9	(a)
Sand	5	27.1-33.8	0.04-0.05	3-4	6.1	(b)
Sand	8	-	0.65	4	-	(b)
Sand	9	-	up to 5.5	17.1	12.2	(b)
Sand	-	13.16	0.05-0.28	7.5	18	(b)
Sand	-	5.5	0.5	-	0.7	(c)
Sandy loam	-	10.8	0.01-0.55	-	10.4	(d)

†(a) Ellis and Childs, 1973.
(b) Dudley and Stevenson, 1973.
(c) Sawhney and Starr, 1977.
(d) Reneau, 1977.

SUMMARY AND FUTURE RESEARCH NEEDS

Annual P losses from both fertilized and control turf plots were 0.25 and 1.3 kg/ha/yr, respectively. On the fertilized turf plots less than one percent of the applied phosphorus leached. Acidic soil conditions, which favor the formation of rather insoluble iron, aluminum and manganese phosphates, may be responsible for soil attenuation of phosphorus.

Substantial P removals (95%) were seen in passage of septic tank effluent through one meter of undisturbed outwash soil on a relatively new septic system field. Lower removal efficiencies were seen with the two types of sand filters studied. Approximately 1/4 and 3/4 of the septic tank phosphorus load was removed by the RUCK and the recirculating sand filters, respectively. As a convention septic system is subjected to long term loading, P removal rates could decline markedly from those reported here.

The site selected for this study was comprised of soils derived from permeable glacial outwash parent material. These soils predominate along the western shore of Narragansett Bay. Due to their permeable nature, overland runoff does not commonly occur from soils composed of this type of soil parent material and most offsite transport of water and chemicals is expected to move to the groundwater through leaching.

In contrast, the soils on the islands and eastern shore of Narragansett Bay are developed from glacial till material. These soils have a lower permeability than do outwash soils and generate overland runoff more frequently. Offsite movement of phosphorus, originating from applied fertilizers or surfacing in effluent from a failed septic system, could be significantly greater in overland flow on till sites than what was observed leaching to the groundwater in this study. Sediment bound transport of P, also a likelihood in till soils, needs to be investigated.

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Appendix A. Raw phosphorus data for turf and ISDS sites.

Filename: SUBURBP.WKS

PROJECT DATA
Supplement to Final Report on
Phosphorous loadings to the groundwater from suburban land uses.
March 1988

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Kingston, RI 02881

ND = No Detection below 0.01 mg/l.

Sample collection date	Sample ID code	Sample type	Ortho-phosphate phosphorous (mg/l)	Effluent pH
09-Jun-86	C1	Unfertilized lawn plot	0.17	-
09-Jun-86	C4	Unfertilized lawn plot	0.30	-
09-Jun-86	P6	High fertility lawn plot	0.29	-
09-Jun-86	P8	High fertility lawn plot	0.10	-
09-Jun-86	P11	High fertility lawn plot	0.13	-
11-Jun-86	C4	Unfertilized lawn plot	0.34	-
23-Jun-86	C1	Unfertilized lawn plot	0.19	-
26-Jun-86	C4	Unfertilized lawn plot	0.26	-
22-Jun-86	P6	High fertility lawn plot	0.11	-
22-Jun-86	P8	High fertility lawn plot	ND	-
22-Jun-86	P11	High fertility lawn plot	0.03	-
01-Jul-86	C1	Unfertilized lawn plot	0.13	-
01-Jul-86	C4	Unfertilized lawn plot	0.22	-
01-Jul-86	C5	Unfertilized lawn plot	0.22	-
01-Jul-86	P6	High fertility lawn plot	0.03	-
01-Jul-86	P8	High fertility lawn plot	0.03	-
01-Jul-86	P11	High fertility lawn plot	ND	-
16-Jul-86	C1	Unfertilized lawn plot	0.14	-
15-Jul-86	C4	Unfertilized lawn plot	0.21	-
15-Jul-86	C5	Unfertilized lawn plot	0.28	-
15-Jul-86	P6	High fertility lawn plot	ND	-
15-Jul-86	P8	High fertility lawn plot	ND	-
15-Jul-86	P11	High fertility lawn plot	0.01	-
01-Aug-86	C1	Unfertilized lawn plot	0.09	-
01-Aug-86	C4	Unfertilized lawn plot	0.27	-
01-Aug-86	C5	Unfertilized lawn plot	0.38	-
01-Aug-86	P6	High fertility lawn plot	ND	-
01-Aug-86	P8	High fertility lawn plot	ND	-
04-Aug-86	P11	High fertility lawn plot	ND	-
15-Aug-86	C1	Unfertilized lawn plot	0.12	-
15-Aug-86	C4	Unfertilized lawn plot	0.19	-
15-Aug-86	C5	Unfertilized lawn plot	0.30	-
15-Aug-86	P6	High fertility lawn plot	ND	-
15-Aug-86	P8	High fertility lawn plot	0.01	-
01-Sep-86	C1	Unfertilized lawn plot	0.07	-
01-Sep-86	C4	Unfertilized lawn plot	0.11	-
01-Sep-86	C5	Unfertilized lawn plot	0.36	-
03-Sep-86	P6	High fertility lawn plot	ND	-

Sample collection date	Sample ID code	Sample type	Ortho-phosphate phosphorous (mg/l)	Effluent pH
01-Sep-86	P8	High fertility lawn plot	ND	-
01-Sep-86	P11	High fertility lawn plot	ND	-
17-Sep-86	C1	Unfertilized lawn plot	0.06	-
17-Sep-86	C4	Unfertilized lawn plot	0.13	-
17-Sep-86	C5	Unfertilized lawn plot	0.19	-
17-Sep-86	P6	High fertility lawn plot	ND	-
17-Sep-86	P8	High fertility lawn plot	ND	-
17-Sep-86	P11	High fertility lawn plot	0.01	-
02-Sep-86	BLACK	Septic tank effluent	3.72	7.39
02-Sep-86	RC1SF	Recirc. sand filter effluent	0.31	4.10
02-Sep-86	RC2SF	Recirc. sand filter effluent	0.16	4.04
02-Sep-86	RC3SF	Recirc. sand filter effluent	0.35	4.24
02-Sep-86	RU1SF	RUCK sand filter effluent	1.15	3.81
02-Sep-86	RU2SF	RUCK sand filter effluent	1.43	3.81
02-Sep-86	RU3SF	RUCK sand filter effluent	1.36	3.82
02-Oct-86	BLACK	Septic tank effluent	6.10	7.23
02-Oct-86	CO1LB	Conventional system lysimeter	ND	-
02-Oct-86	CO2LB	Conventional system lysimeter	ND	-
02-Oct-86	CO3LB	Conventional system lysimeter	ND	-
02-Oct-86	RC1SF	Recirc. sand filter effluent	0.73	3.92
02-Oct-86	RC2SF	Recirc. sand filter effluent	0.34	3.88
02-Oct-86	RC3SF	Recirc. sand filter effluent	0.53	3.95
02-Oct-86	RU1SF	RUCK sand filter effluent	1.76	3.54
02-Oct-86	RU2SF	RUCK sand filter effluent	3.10	3.56
02-Oct-86	RU3SF	RUCK sand filter effluent	2.46	3.23
12-Nov-86	BLACK	Septic tank effluent	5.75	6.97
12-Nov-86	CO1LA	Conventional system lysimeter	ND	-
12-Nov-86	CO2LC	Conventional system lysimeter	ND	-
12-Nov-86	CO3LD	Conventional system lysimeter	ND	-
12-Nov-86	RC1SF	Recirc. sand filter effluent	1.46	4.29
12-Nov-86	RC2SF	Recirc. sand filter effluent	0.63	3.98
12-Nov-86	RC3SF	Recirc. sand filter effluent	0.60	3.91
12-Nov-86	RU1SF	RUCK sand filter effluent	4.55	3.57
12-Nov-86	RU2SF	RUCK sand filter effluent	5.00	3.52
12-Nov-86	RU3SF	RUCK sand filter effluent	4.55	3.54
09-Dec-86	BLACK	Septic tank effluent	3.83	7.23
09-Dec-86	CO1LA	Conventional system lysimeter	ND	-
09-Dec-86	CO2LB	Conventional system lysimeter	ND	-
09-Dec-86	CO3LA	Conventional system lysimeter	ND	-
09-Dec-86	RC1SF	Recirc. sand filter effluent	1.34	4.91
09-Dec-86	RC2SF	Recirc. sand filter effluent	0.54	4.10
09-Dec-86	RU1SF	RUCK sand filter effluent	3.63	3.71
09-Dec-86	RU2SF	RUCK sand filter effluent	3.40	3.68
09-Dec-86	RU3SF	RUCK sand filter effluent	3.38	3.67
19-Jan-87	BLACK	Septic tank effluent	3.37	7.62
19-Jan-87	CO1LD	Conventional system lysimeter	ND	-
19-Jan-87	CO2LC	Conventional system lysimeter	ND	-
19-Jan-87	CO3LD	Conventional system lysimeter	1.44	-
19-Jan-87	RC1SF	Recirc. sand filter effluent	1.14	6.74
19-Jan-87	RC2SF	Recirc. sand filter effluent	0.32	5.66
19-Jan-87	RC3SF	Recirc. sand filter effluent	0.33	5.48

Sample collection date	Sample ID code	Sample type	Ortho-phosphate phosphorous (mg/l)	Effluent pH
19-Jan-87	RU1SF	RUCK sand filter effluent	2.68	3.78
19-Jan-87	RU2SF	RUCK sand filter effluent	1.80	4.36
19-Jan-87	RU3SF	RUCK sand filter effluent	1.78	5.50
11-Mar-87	BLACK	Septic tank effluent	4.14	7.25
11-Mar-87	CO1LD	Conventional system lysimeter	ND	-
11-Mar-87	CO3LD	Conventional system lysimeter	1.55	-
11-Mar-87	RC1SF	Recirc. sand filter effluent	2.68	7.55
11-Mar-87	RC2SF	Recirc. sand filter effluent	1.81	7.56
11-Mar-87	RC3SF	Recirc. sand filter effluent	1.57	7.46
11-Mar-87	RU1SF	RUCK sand filter effluent	3.33	7.15
11-Mar-87	RU3SF	RUCK sand filter effluent	3.26	7.07
20-Apr-87	BLACK	Septic tank effluent	3.36	7.14
20-Apr-87	CO1LD	Conventional system lysimeter	ND	-
20-Apr-87	CO2LB	Conventional system lysimeter	ND	-
20-Apr-87	CO3LB	Conventional system lysimeter	ND	-
20-Apr-87	RC1SF	Recirc. sand filter effluent	1.87	6.99
20-Apr-87	RC2SF	Recirc. sand filter effluent	1.66	6.59
20-Apr-87	RC3SF	Recirc. sand filter effluent	1.59	4.44
20-Apr-87	RU1SF	RUCK sand filter effluent	4.16	4.07
20-Apr-87	RU2SF	RUCK sand filter effluent	3.93	3.84
20-Apr-87	RU3SF	RUCK sand filter effluent	4.27	3.87
01-Jun-87	BLACK	Septic tank effluent	3.26	7.26
01-Jun-87	CO1LA	Conventional system lysimeter	ND	-
01-Jun-87	CO2LB	Conventional system lysimeter	ND	-
01-Jun-87	CO3LA	Conventional system lysimeter	ND	-
01-Jun-87	RC1SF	Recirc. sand filter effluent	1.54	3.89
01-Jun-87	RC2SF	Recirc. sand filter effluent	1.48	3.88
01-Jun-87	RC3SF	Recirc. sand filter effluent	0.91	3.81
01-Jun-87	RU1SF	RUCK sand filter effluent	2.87	3.71
01-Jun-87	RU2SF	RUCK sand filter effluent	3.66	3.50
01-Jun-87	RU3SF	RUCK sand filter effluent	2.90	3.50
30-Jun-87	BLACK	Septic tank effluent	4.12	7.33
01-Jul-87	CO1LD	Conventional system lysimeter	ND	-
01-Jul-87	CO2LB	Conventional system lysimeter	ND	-
01-Jul-87	CO3LD	Conventional system lysimeter	ND	-
01-Jul-87	RC1SF	Recirc. sand filter effluent	1.17	3.74
01-Jul-87	RC3SF	Recirc. sand filter effluent	0.50	3.79
01-Jul-87	RU2SF	RUCK sand filter effluent	3.43	3.43
01-Jul-87	RU3SF	RUCK sand filter effluent	2.81	3.45
08-Sep-87	CO1LC	Conventional system lysimeter	ND	-
08-Sep-87	CO2LB	Conventional system lysimeter	ND	-
08-Sep-87	CO3LD	Conventional system lysimeter	2.61	-
08-Sep-87	RC1SF	Recirc. sand filter effluent	1.44	3.69
08-Sep-87	RC2SF	Recirc. sand filter effluent	1.16	3.69
08-Sep-87	RC3SF	Recirc. sand filter effluent	0.52	3.66
08-Sep-87	RU1SF	RUCK sand filter effluent	3.81	3.47
08-Sep-87	RU2SF	RUCK sand filter effluent	4.65	3.39
08-Sep-87	RU3SF	RUCK sand filter effluent	3.98	3.41

Appendix B. Evaluation of nitrogen removal systems for on-site sewage disposal.

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EVALUATION OF NITROGEN REMOVAL SYSTEMS FOR ON-SITE SEWAGE DISPOSAL

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Leaching of nitrate-N from conventionally designed on-site sewage disposal systems threatens surface and groundwater quality in unsewered areas of the United States (Miller 1975). Nitrogen inputs to coastal ponds and estuaries may promote increased eutrophication and water quality degradation (Ryther and Dunstan 1971), while inputs to drinking water sources can raise nitrate-N concentrations above the 10 mg/l Federal drinking water standard (Preul 1966; USEPA 1976).

Conventional on-site sewage disposal systems are not specifically designed to promote nitrogen removal processes (Preul 1966). Nitrogen entering a conventional system is in the organic-N and ammonium-N forms. In the septic tank, settlement and ammonification occur, resulting in effluent containing primarily ammonium-N (USEPA 1980; Canter and Knox 1985). This ammonium-N is rapidly converted to nitrate-N in the subsequent aerobic soil absorption field (Andreoli et al. 1979). Soil cation adsorption mechanisms do not affect the soluble nitrate anion, allowing it to move freely with percolating effluent to the groundwater and nearby surface waters.

Biological denitrification, the reduction of nitrate-N to nitrogen gas, represents one of the most desirable nitrogen removal pathways for on-site sewage disposal. Denitrification requires: (i) the oxidation of ammonium-N to nitrate-N; (ii) the presence of a subsequent anaerobic zone; and (iii) an adequate carbon source for the denitrifying bacteria in the anaerobic zone. In a properly functioning soil absorption field, denitrification is limited by the absence of reduced conditions following nitrification and by the lack of an available carbon source.

The research described here presents the results of the first 16 months of a replicated field study comparing the performance of several alternative nitrogen-removal systems to the performance of a conventional system under the same field conditions. The overall goal of this research was to identify a practical nitrogen removal system for on-site sewage disposal. The specific objectives were: (i) evaluate the degree of nitrification provided by a buried multimedia filter (RUCK filter) and a free access recirculating sand filter; (ii) assess the ability of soil absorption trenches and rock tanks to provide the environment necessary for denitrification; and (iii) compare the effectiveness of household wastewaters (greywater and septic tank effluent) and methanol as carbon sources for denitrification.

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METHODS

An on-site sewage disposal field laboratory was designed and constructed at the University of Rhode Island in 1986. The soil parent material at the site consisted of well-drained stratified glacial outwash sands and gravels.

The waste source used for the project was from the South Kingstown Lift Station which services the University and surrounding community. Additional organic-N in the form of urea was added to the waste source by a metered nutrient pump to raise total-N levels to the 40 - 60 mg/l range typical of household wastewater (Canter and Knox 1985). Greywater was obtained from a nearby residential home approximately every two weeks and stored in a separate tank located on the site.

The field laboratory consisted of three 1/5th scale replicates of each of three types of on-site sewage disposal systems: a RUCK multimedia filter system, a recirculating sand filter system and a conventional septic tank/soil absorption trench system. The conventional systems served as standards for comparisons with the other two systems.

The RUCK systems incorporated buried multimedia filters to achieve nitrification (Laak 1986). Greywater was used as the carbon source and was mixed with filter effluent at a 1:3 ratio prior to entering the anaerobic zone. From July 1986 to June 1987, the RUCK systems relied on soil absorption trenches to provide the anaerobic environment. From June 1987 through October 1987, rock tanks with a three day retention period served as the sites for anaerobiosis.

The recirculating sand filter systems used free access sand filters to achieve nitrification. Rock tanks with a three day retention period provided the anaerobic environment throughout the study. From July 1986 to September 1987, a portion of the unfiltered septic tank effluent served as the carbon source, mixing with filter effluent at a ratio of 1:4 prior to entering the rock tanks. This ratio was chosen to maintain nitrogen levels of less than 10 mg/l following the denitrification stage. From September 1987 through October 1987, methanol was added to the rock tanks to provide the carbon source, at a ratio of 1 part methanol to 2000 parts filter effluent.

Each of the nine systems received 115 liters of septic tank effluent per day delivered in twelve equal increments. To ensure even wastewater distribution, pressure dosing was employed using specially designed orifice flow manifolds. The entire effluent delivery system was calibrated and checked for leakage after installation to ensure proper hydraulic loadings. Mercury float switches installed in dosing tanks and programmable electronic timers were used to activate and control scheduled effluent doses.

The field laboratory began operation in June 1986. Samples were taken approximately every three weeks from each component of the nine systems for the study period July 1986 through October 1987. Access ports installed at discrete points throughout the systems permitted sampling from: the septic tank, the greywater tank, sand filters and rock tanks. Porous suction cup lysimeters (Soil Moisture Equipment Co., Santa Barbara, CA) were placed in the vadose zone 0.9 meters below the base of each soil absorption trench to collect soil water leachate. Each trench contained 3 to 5 lysimeters. Suction cup lysimeters were also placed upgradient from the research site to provide background soil water leachate data. Soil water tension at the washed stone/undisturbed soil interface in each absorption trench was measured by tensiometers (Soil Moisture Equipment Co., Santa Barbara, CA).

Tank and filter samples were analyzed for temperature and pH prior to sample preservation and storage (APHA 1985). All samples were subsequently analyzed for ammonium-N by the colorometric salicylate-hypochlorite method (Bower and Holm-Hansen 1980); nitrate-N plus nitrite-N by cadmium reduction using an automated Technicon Auto-Analyzer (Lambert and Oviatt 1986); and chloride by the argentometric method (APHA 1985) and the specific ion electrode method (Orion Research Inc. 1984). Tank and filter samples were additionally analyzed for Total Kjeldahl Nitrogen (TKN) by the block digestor method (Eastin 1978; USEPA 1979). After digestion, ammonium-N was determined by the colorometric method mentioned above. Total organic carbon (TOC) was determined routinely on rock tank samples, septic tank effluent and greywater using an I.O. Corporation Model 700 Total Organic Carbon Analyzer (APHA 1985).

RESULTS AND DISCUSSION

Nitrification

Nitrification of septic tank effluent prior to the denitrification step is critical to the success of nitrogen removal on-site sewage disposal systems. Ammonium-N or organic-N which has not been nitrified may pass virtually unchanged through the anaerobic zone. The potential then exists for these nitrogen forms to be oxidized in the aerobic portions of the soil absorption trench, allowing the nitrate-N produced to leach into the environment.

Based on a paired comparison "t" test, the mean annual percent nitrification for the recirculating sand filters (66.0%, S.E.=2.6%) was not significantly different ($P>0.05$) from the RUCK multimedia filters (70.0%, S.E.=1.4%) for the period October 1986 to October 1987. However, seasonal trends in nitrification were observed for both types of filters (Fig. 1). Both filters achieved 70 to 90 percent nitrification at effluent temperatures greater than approximately 10°C. During the winter months, when effluent temperatures dropped below 10°C, nitrification decreased markedly. In the recirculating sand filters, percent nitrification dropped to a low of 24 percent, while in the RUCK filters it dropped to 44 percent. The mean effluent temperatures during the winter were 2.7°C for the recirculating sand filters and 4.3°C for the RUCK filters.

Other studies which have looked at the role of nitrification in wastewater treatment have indicated a correlation between temperature and nitrification (McCarty and Haug 1971; Kristiansen 1981b). In this study, the relationship between the log of effluent temperature and percent nitrification was highly significant ($P<0.01$) for both the RUCK and recirculating sand filters (RSF) in least-squares regression analysis. The regression equations computed were:

RUCK: Percent nitrification = 39.6 + 29.2log(temp) $R^2 = 0.61$ (1)

RSF: Percent nitrification = 27.5 + 40.2log(temp) $R^2 = 0.55$ (2)

The recirculating sand filters showed a more pronounced decrease in nitrification than did the RUCK filters in January, February and March (Fig. 1). In part this decrease may have been due to filter design: the free access recirculating sand filters were directly exposed to the air and were therefore more susceptible to lower ambient temperatures than were the buried RUCK filters. The differences in winter effluent temperatures and nitrification rates for the two types of filter may also have been due to maintenance problems associated with the recirculating sand filters. During the months of January and February, an ice layer developed on the surface of each recirculating sand filter which impeded effluent percolation and caused filter failure. Slight modifications of the filter design were subsequently

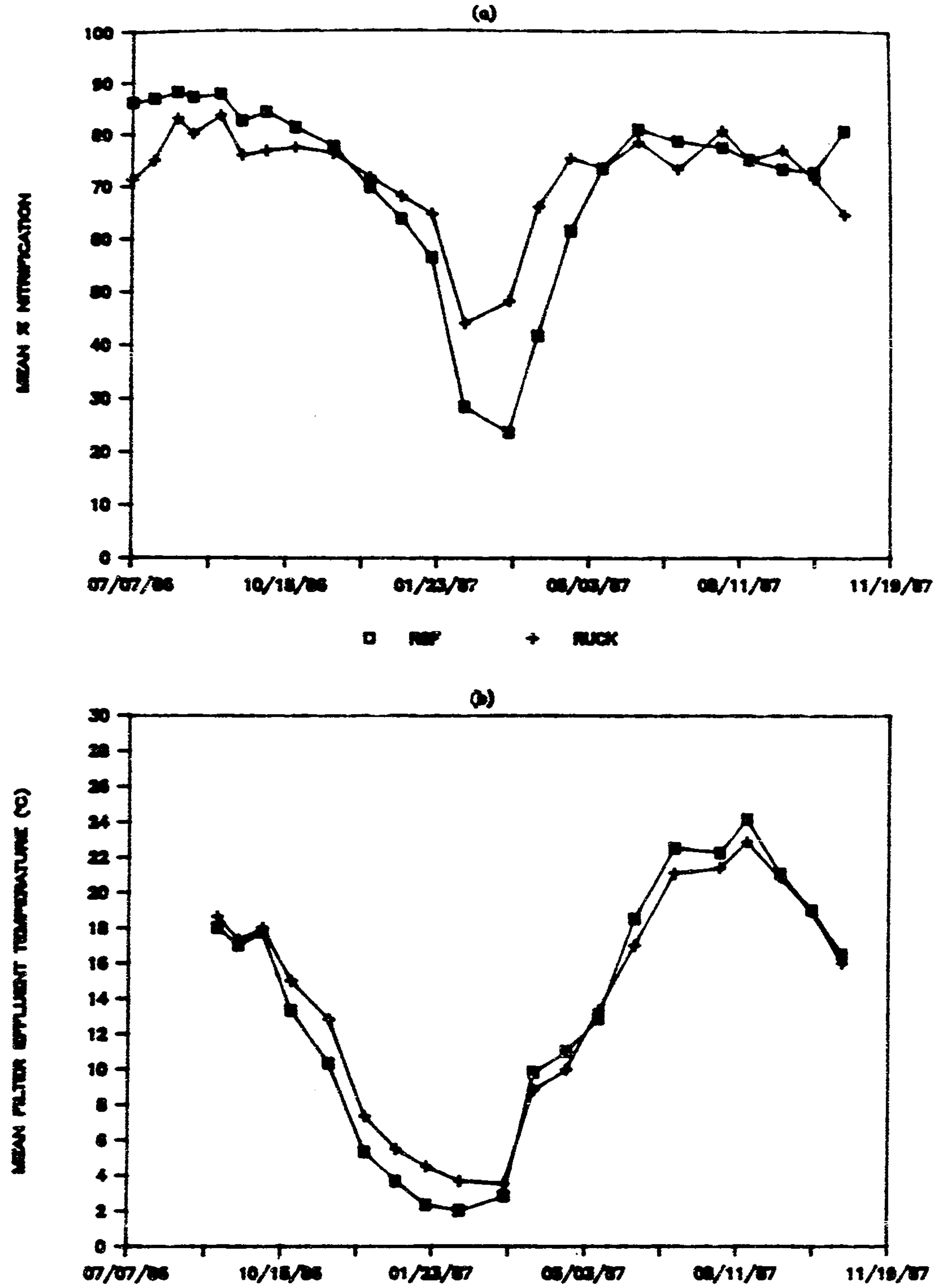

Fig. 1 Nitrification performance of recirculating sand filters (RSF) and RUCK multimedia filters. a) Mean percent nitrification. b) Mean filter effluent temperature.

made which should alleviate the problem of ice build-up in following winters.

In a comparison of the two aerobic filters, other design and maintenance features also need to be considered. The recirculating sand filter is a well-tested EPA filter design (USEPA 1980). However, it has several characteristics, such as its reliance on mechanical pumps, its free access status, and its need for routine maintenance of the filter surface that have precluded its acceptance by many states.

The RUCK filter is a relatively recent design (Laak 1986) which has not received the same extent or longevity of testing as the recirculating sand filter. The present design incorporates percolation indrains in a layered sand and gravel matrix. Due to the increased time and labor involved in construction, RUCK filters are more expensive than a single media filter. One of the greatest attractions of the RUCK filter for regulatory agencies is that the wastewater flows entirely by gravity beneath the ground's surface. There is no need for pumps or maintenance as required with the recirculating sand filter. The results presented here indicate that after 16 months, the performance of the RUCK filters was similar to the performance of the recirculating sand filters. Testing of the RUCK filters should continue, however, to assess its performance over longer periods of time.

Rock Tank Denitrification and Total System Nitrogen Removal

Total nitrogen removal in each system was calculated based on changes in the chloride:nitrogen ratio between incoming septic tank effluent and the percolate in the absorption trench lysimeters. Mean background soil percolate concentrations of chloride and total nitrogen were 3.8 mg/l and 0.4 mg/l respectively, while average concentrations in septic tank effluent were 45.9 mg/l chloride and 49.2 mg/l total nitrogen.

In the first nine months of the study, total nitrogen concentrations in the percolate waters below the conventional absorption trenches were routinely higher than in the septic tank effluent, suggesting the occurrence of nitrogen mineralization. During this period, the RUCK and recirculating sand filter systems both generated negative estimates of total system nitrogen removal, although less frequently than did the conventional systems. Estimates of total system nitrogen removal may be conservative for all systems during these first nine months.

Denitrification in the rock tanks was computed as the percent reduction between the nitrate entering and leaving the rock tanks. Although total system nitrogen removal included losses within the aerobic filters, rock tanks and soil absorption trenches, the percent total system nitrogen removal was generally less than the percent denitrification. This difference was due to the addition of nitrogen from the carbon source, the lack of complete nitrification in the aerobic filters, and nitrogen mineralization within the trenches.

For the first 12 months of the study, the RUCK systems used the soil absorption trenches as the anaerobic environment and greywater as the carbon source. Previous studies (Kristiansen 1981a) indicated that an anaerobic clogging mat could develop in an absorption trench after 6 to 15 months of continuous effluent applications, creating an environment in which denitrification might occur. If the absorption trench provided a suitable anaerobic environment, the cost and complexity of alternative systems could be significantly reduced.

Using the soil absorption trenches as the anaerobic site for denitrification, the mean total system nitrogen removal for the RUCK systems from July 1986 to June 1987 was 6 percent (Table 1). During this period, total nitrogen removal in the RUCK systems was significantly higher ($P<0.05$) than in the conventional systems on only 17 percent of the sampling dates. Tensiometer data and trench excavation did reveal the development of clogging mats in the absorption trenches by the spring of 1987. Average total system nitrogen removal in the spring ranged from 8 to 38 percent. Nitrogen removal might have increased with time as a thicker mat developed, but it was felt that a system which had provided a maximum of only 38 percent total nitrogen removal after 12 months of operation was not an effective alternative system.

In June 1987, rock tanks were added to the RUCK systems to replace the soil absorption trenches as the anaerobic environments. With greywater still used as the carbon source, the RUCK systems performed significantly better ($P<0.05$) than conventional systems on every sampling date (Table 1). Based on changes in the chloride:nitrogen ratios between the septic tank effluent and the absorption trench lysimeters, mean total system nitrogen removal for the RUCK systems during this second phase was 50 percent. Rock tanks achieved an average of 51 percent denitrification during this period.

Table 2 presents the comparison of the total system nitrogen removal and denitrification for the recirculating sand filter systems and the conventional systems. Using septic tank effluent as the carbon source, total system nitrogen removal for the two types of systems was significantly different ($p<0.05$) during only 19 percent of the sampling dates. Rock tanks in the recirculating sand filter systems achieved an average of 25 percent denitrification during this period.

With the introduction of methanol as the carbon source for the recirculating sand filter systems, immediate differences in denitrification and total system nitrogen removal were seen. Rock tanks achieved an average of 97 percent denitrification, with a mean total nitrogen removal for the systems of 84 percent (Table 2). The conventional systems achieved an average of 5 percent total system nitrogen removal for this period.

These results illustrate the importance of the carbon source in the process of denitrification. A carbon:nitrate ratio of at least 2:1 to 3:1 is usually considered necessary for optimal denitrification (Bremner and Shaw 1958). To minimize total nitrogen losses from a system, the carbon source should also have negligible concentrations of TKN since it is introduced into the rock tank without prior nitrification. Any TKN present in the carbon source has the potential to be later nitrified in the soil absorption trench.

Methanol performed the best of the carbon sources studied; the TKN concentration was 0 mg/l and the TOC concentration could be easily calculated to achieve the optimal carbon:nitrate ratio desired. Of the two types of wastewater studied, greywater appeared to be superior to septic tank effluent as a carbon source. Greywater had lower mean TKN concentrations (15 mg/l) and higher mean TOC concentrations (62 mg/l) than did septic tank effluent (49 mg/l TKN and 35 mg/l TOC). Using septic tank effluent, the carbon:nitrate ratio entering the rock tank was approximately 0.7:1; using greywater it was approximately 1:1. These ratios suggest that the amount of TOC in both greywater and septic tank effluent may have been a limiting factor for denitrification. The greywater used in this study was stored approximately 10 days before mixing with aerobic filter effluent and may have had a lower TOC value than greywater coming directly from a home. There is a need for additional research to better characterize the carbon content of greywater and the factors that affect it.

Table 1. RUCK System versus Conventional System (CONV): Total Nitrogen (N) Removal Comparisons

Sampling Period	System	Anaerobic Zone	Mean Total System N Removal	Mean Denitrification in Rock Tanks	No. of sampling dates	% of dates[a] RUCK>CONV	% of dates[a] CONV>RUCK
			(%)	(%)			
July 1986 - June 1987	RUCK	Absorption Trench	6 %	-----	18	17 %	0 %
	CONV	-----	<1 %	-----			
June 1987 - Oct. 1987	RUCK	Rock Tank	50 %	51 %	6	100 %	0 %
	CONV	-----	6 %	-----			

[a] Significantly different at $P<0.05$ in accordance with Fisher's Least Significant Difference (LSD) test.

Table 2. Recirculating Sand Filter System (RSF) versus Conventional System (CONV): Total Nitrogen (N) Removal Comparisons

Sampling Period	System	Carbon Source	Mean Total System N Removal	Mean Denitrification in Rock Tanks	No. of sampling dates	% of dates[a] RSF>CONV	% of dates[a] CONV>RSF
			(%)	(%)			
July 1986 - Sept. 1987	RSF	Septic Tank Effluent	<1 %	25 %	21	19 %	0 %
	CONV	-----	<1 %	-----			
Sept. 1987 - Oct. 1987	RSF	Methanol	84 %	97 %	3	100 %	0 %
	CONV	-----	5 %	-----			

[a] Significantly different at $P<0.05$ in accordance with Fisher's Least Significant Difference (LSD) test.

SUMMARY AND RECOMMENDATIONS

This study demonstrated that the nitrification process can be a limiting factor for successful total system nitrogen removal. The two aerobic filters studied nitrified a significant portion of the influent wastewater, although both appeared to be limited by temperature. Further observations and research are needed to assess: (i) the continued performance of the relatively untested RUCK filter as the system matures; (ii) the performance of the recirculating sand filter under winter conditions; and (iii) the seasonal variations demonstrated in both filters.

The soil absorption trenches did not appear to provide a suitable anaerobic environment for denitrification after one year of continuous use. Based on the results of this study, rock tanks are recommended as a more effective anaerobic environment.

Methanol was the most successful of the three carbon sources studied in this project. With methanol, total system nitrogen removal in the recirculating sand filter system was limited only by nitrification. Almost 100 percent denitrification was achieved in the rock tank. Use of methanol in alternative systems is complicated, however, by the need for the operation and maintenance of a methanol feed system. The use of household wastewaters as a carbon source does not require such a feed system. Greywater showed more promise as a suitable carbon source of the two types of wastewater studied, although further research needs to be done to better characterize its carbon and nitrogen content. Even with optimal TOC concentrations, total nitrogen removal in a greywater system will still be limited by both the degree of nitrification and by the presence of nitrogen in the greywater.

The results presented here provide a preliminary assessment of alternative denitrification on-site sewage disposal systems. The last four months of the study suggest that significant nitrogen removal can be achieved using a rock tank as the anaerobic zone and either methanol or greywater as the carbon source. However, these successful designs have only been tested during the summer months when there is optimal biological activity. Continued monitoring of controlled, replicated systems is needed to assess the long-term performance of these designs under a greater variety of environmental conditions.

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Appendix C. Influence of overwatering and fertilization on nitrogen losses from home lawns.

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Influence of Overwatering and Fertilization on Nitrogen Losses from Home Lawns

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Influence of Overwatering and Fertilization on Nitrogen Losses from Home Lawns

T. G. MORTON, A. J. GOLD,* AND W. M. SULLIVAN

ABSTRACT

Fertilized home lawns represent a potential source of NO_3-N contamination to groundwater and surface waters. The waterborne losses of inorganic N from Kentucky bluegrass (*Poa pratensis* L.) turf subjected to three levels of N fertilization (0, 97, and 244 kg N ha^{-1} yr^{-1} as urea and methylene urea) and two irrigation regimes (scheduled by tensiometer and overwatering with 3.75 cm of water per week in addition to rainfall) were measured. The site was located on a Merrimac sandy loam (sandy, mixed, mesic Typic Dystrochrept). Soil-water percolate was collected by suction plate lysimeters placed below the root zone. Surface runoff was quantified with orifice flow splitters. Soil-water percolate flux comprised >93% of the total water and inorganic-N discharged from all treatments. Mean annual flow weighted concentrations of inorganic N in soil-water percolate were below the U.S. drinking water standard on all treatments and ranged from 0.36 mg L^{-1} on the overwatered, unfertilized, control treatment to 4.02 mg L^{-1} on the overwatered, high N treatment. Annual losses ranged from 32 kg ha^{-1} on the overwatered high N rate treatment to 2 kg ha^{-1} on the scheduled irrigation, unfertilized, control treatment. Overwatering in conjunction with fertilization generated significantly higher annual flow weighted concentrations and mass loss than the unfertilized controls. Nitrogen loss and concentrations from the scheduled irrigation treatments were not significantly different from the controls.

Additional Index Words: Nitrate-nitrogen, Turfgrass, Groundwater pollution, Water quality, Irrigation scheduling.

Since 1970, pesticide and fertilizer use on home lawns has steadily increased (Watschke, 1983). The growth of chemical use suggests the possibility for an increase in off-site losses and subsequent environmental contamination. Lawn care chemicals may be applied in close proximity to impervious zones with high potential for surface runoff. Miller et al. (1974), in their study of groundwater contamination in the northeastern USA, stressed the need for long-term studies to determine if home lawn agrichemicals have penetrated the soil zone and entered the groundwater system.

Several researchers have described conditions under which they found substantial NO_3-N leaching from fertilized cool season turfgrass (Owen and Barraclough, 1983; Rieke and Ellis, 1974). Nitrate-N is a drinking water contaminant with a U.S. drinking water standard of 10 mg L^{-1} (USEPA, 1976). Leaching of NO_3 is of particular

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concern on Long Island, NY, and the southern New England states where permeable, outwash soils overlie unconfined drinking water aquifers. Coastal estuaries and bays have been found to be N limited, and may be degraded by concentrations of NO_3-N much less than the drinking water standard of 10 mg L^{-1} (Ryther and Dunstan, 1971). Other researchers have shown little increase in NO_3-N leaching from fertilized turfgrass (Starr and DeRoo, 1981; Snyder et al., 1984). Starr and DeRoo (1981) monitored the fate of N applied to cool season turfgrasses in southern New England and found low concentrations of NO_3-N in leachate when moderate rates of N were applied and no supplemental irrigation water was used.

Irrigation has been shown to significantly increase NO_3-N leaching (Snyder et al., 1984; Endelman et al., 1974; Timmons and Dylla, 1981; Rieke and Ellis, 1974). Home lawns are typically watered with little regard for soil moisture status or the water holding capacity of the soil. Where irrigation is automatically controlled, rates are often selected to meet maximum evaporative demands, resulting in routine overwatering (Snyder et al., 1984). Excessive watering will increase antecedent soil moisture, thereby promoting additional leaching and surface water runoff from natural storm events or from the supplemental water alone.

The goal of this study was to quantify N losses from turf subjected to the range of fertilization and watering practices generally used on home lawns. Commercial applicators often employ several practices that past studies have shown to minimize off-site transport of N (Rieke and Ellis, 1974; Brown et al., 1982). Nitrogen is frequently applied in the form of urea in combination with some form of slow release materials, rather than in immediately available forms. The fertilizer is usually applied in small increments throughout the growing season, which is thought to minimize high N concentrations in the root zone.

However, commercial home lawn care companies often apply greater annual amounts of N than individual homeowners. A survey of 460 households on Long Island found that homeowners applied an average of 122 kg N ha^{-1} yr^{-1} to their lawns (Koppelman, 1978). Many commercial operations apply 220 to 293 kg N ha^{-1} yr^{-1} (J.F. Wilkinson, Old Fox Lawn Care, Inc., 1985, personal communication). The objective of this study was to determine the effects of varying N fertilization rates and irrigation regimes on waterborne losses of inorganic N from home lawns.

MATERIALS AND METHODS

Site Description

Hydrologically isolated plots were established at the Univ. of Rhode Island, Kingston, to monitor surface and subsurface water loss from turfgrass. The soil at the site was classified as a Merrimac sandy loam. Twelve plots (2.1 by 15.2 m) were instrumented for monitoring overland flow. Eighteen plots were instrumented for subsurface collection. Runoff plots had 2 to 3% slope toward surface collection units and no cross slope. Monitoring occurred from October 1984 through October 1986.

Soil-water percolate from six treatments, consisting of three rates of N, coupled with two irrigation regimes, was evaluated. Overland runoff was evaluated for two rates of N and two irrigation regimes. Three replications of each combination of irrigation and fertilizer treatment were established in a completely randomized design.

A mixture of 90% Kentucky bluegrass and 10% red fescue (*Festuca rubra* L.) was planted during the fall of 1980. The turfgrass was maintained at a 5.0- to 7.5-cm height and the clippings remained on the plots.

Instrumentation

A subsurface sprinkler head system was used to irrigate the study site. Half and quarter circle, flat spray nozzles were employed to ensure controlled applications. In addition, plots receiving different irrigation rates were separated by 1.5-m sod buffer strips. Application rate was 5.0 cm h^{-1}. Uniformity of application was measured at >90%.

Ceramic lysimeter plates obtained from Soil Moisture, Inc. (Santa Barbara, CA) were used to measure the flux and quality of soil-water percolate. Each lysimeter was 27 cm in diameter with a 0.05-MPa air entry value. Soil-water percolate samples were collected and temporarily stored in a polyvinyl chloride (PVC) vacuum reservoir stand pipe inserted to a depth of 70 cm. Prior to installation the plates and PVC collection reservoirs were cleaned as described by Miller (1977).

The soil moisture plates were placed 20 cm below the thatch layer where the sandy loam solum abruptly changed to a coarse gravelly sand deposited as glacial outwash. The depth of root penetration was observed to extend to 15 cm. Concentrations of inorganic N that reached 20 cm were assumed to approximate concentrations that would travel to the groundwater. Tensiometers located in the study area revealed that a potential of −0.01 MPa corresponded to the field capacity of this soil 48 h after saturation. To simulate undisturbed drainage, the plates were maintained at a suction of −0.01 MPa after any event intense enough to produce drainage.

Following any precipitation or irrigation event that generated surface runoff or soil-water percolate, samples were removed from the collection system at 24-h intervals for flow quantification and chemical analysis. Any samples not analyzed immediately were frozen at −10 °C. Samples were analyzed for NH_4-N by the steam distillation method and then reduced with DaVarda's alloy and analyzed for NO_2-N and NO_3-N (Bremner, 1965). To ensure quality control, standards of NH_4 and NO_3 were routinely analyzed.

Surface water flow was collected by means of an orifice flow splitter (McLeod and Hegg, 1984) produced by the Engineering Instrument Shop at the Univ. of Rhode Island. Flow splitters were individually calibrated. They directed 9.6 to 11.3% of the total flow to 230-L collection barrels. Runoff water samples were analyzed in the same fashion as the soil-water percolate.

Fertilizer Application

Chemical applications began in June 1984, 4 months before monitoring was initiated. The N application rates investigated were 0, 97, and 244 kg N ha^{-1} yr^{-1} for the control, low, and high N treatments, respectively. Nitrogen was applied at a rate of 48.5 kg ha^{-1} in June and September to the low and high N treatment. The high N treatments received additional applications in July and August (24 kg ha^{-1} per application) and in November (97 kg ha^{-1}) to simulate the commercial home lawn care application rates. Nitrogen was applied in a liquid form as 50% urea and 50% Fluf® (flowable liquid ureaform, manufactured by W.A. Cleary, Somerset, NJ). The low N treatments received 17 kg P ha^{-1} yr^{-1} and 17 kg K ha^{-1} yr^{-1}, whereas the high N treatments had applications of 42 kg P ha^{-1} yr^{-1} and 42 kg K ha^{-1} yr^{-1}.

Irrigation Schedules

Plots were irrigated at two rates. These rates were (i) a scheduled rate to avoid drought stress and to prevent percolation from the root zone, and (ii) a rate to simulate overwatering. On scheduled irrigation plots, irrigation was initiated when the soil-water potential measured by two out of three tensiometers attained −0.05-MPa tension. To wet the soil to field capacity (−0.01 MPa) while preventing drainage from the root zone, 1.2 cm of water was applied. This irrigation depth was based on a soil moisture characteristic curve obtained in the laboratory from undisturbed soil cores of the Ap horizon (Richards, 1965). The rate chosen to simulate overwatering was three applications per week of 1.25 cm per application (3.75 cm wk^{-1}), regardless of rainfall. This irrigation depth represents the mean maximum weekly evapotranspiration in Rhode Island during the summer months (NOAA, 1982).

Soil-water Percolate Flux

For most of the rainfall and irrigation events, the plates collected all the expected percolate. However, on large storm events the plates were unable to adequately quantify percolate flux (volume plate per area per event). During the study, several storms ≥8.0 cm d^{-1} generated no overland runoff from irrigated turfgrass, while the lysimeters collected approximately 2.5 cm of percolate. Since soil storage above the collection plates could not account for the additional water, a mass balance model was created to correct the percolate estimates from large storm events. Hergert (1986) also used suction lysimeters to quantify flux volume. He found similar discrepencies during large flux events and used a water balance to improve flux estimates

Daily percolation (P_i) was computed by the equation of Kincaid et al. (1979)

$$P_i = PPT_i - ET_i + SM_{i-1} - SM_i$$

where P_i = water percolating from root zone on a given day (cm), PPT_i = precipitation or irrigation on a given day (cm), ET_i = evapotranspiration on a given day derived from the modified Penman equation (cm), SM_{i-1} = soil moisture content on the previous day (cm), and SM_i = soil moisture content on a given day (cm).

Following the approach of Smith and Williams (1980), leaching was assumed to occur whenever the soil moisture of the root zone exceeded field capacity. All of the precipitation was assumed to infiltrate into the soil. Potential evapotranspiration was computed using the modified Penman equation (Doorenbos and Pruitt, 1977). Meterological data was obtained from the Rhode Island Agric. Exp. Stn. weather station located 1200 m from the study site.

The predictive mass balance model was run for the entire study period. On events where the observed percolation differed by 1.0 cm or more from the predicted percolate flux, the corrected percolate flux value was used and chemical losses calculated from that value. For all other events, the observed flux was used in calculation of N loss.

Statistical Analysis

Data were subjected to analysis of variance procedures using the Statistical Analysis System (SAS Inst., Inc., 1982). Significant differences in means were tested using the least significant difference test at the 0.05 level. Beginning in the winter of 1986, rodent activity was observed in the turfgrass directly over one of the lysimeters on the high N, scheduled irrigation treatments. A corresponding increase in inorganic-N concentrations from this lysimeter was observed. Inorganic-N concentrations rose in excess of 40 mg L^{-1} for the remainder of the study, while the other replicates of this treatment averaged <3 mg L^{-1}. Results from this replication were excluded from the data set during the second year of study.

RESULTS AND DISCUSSION

Soil-water Percolation

An annual and seasonal hydrologic balance for the periods studied is presented in Table 1. Since there were no significant differences in the observed quantity of soil-water percolate between N treatments, the observed and corrected flux estimates were averaged over the N treatments. Water losses estimated from the sum of evapotranspiration and percolation estimates derived solely from the plate lysimeters did not account for 13 to 56% of the total water inputs. Correcting the lysimeter flux for storms where the observed percolation differed by 1.0 cm or more (large storms) from the flux predicted by the hydrologic model markedly improved the mass water balance. The discrepency between seasonal inputs and losses dropped to a range of 5 to 33% with the corrected estimates of percolation. The percolation estimates corrected for large storms were used in computing mass N loss estimates and seasonal flow weighted N concentrations.

Table 1. Annual and seasonal hydrologic balance for irrigated and nonirrigated treatments.

Components of hydrologic balance	Nonirrigation period Oct. 1984–June 1985		Irrigation period July–Oct. 1985		Nonirrigation period Oct. 1985–June 1986		Irrigation period June–Oct. 1986		Mean annual values	
	Schd.†	Over.‡	Schd.	Over.	Schd.	Over.	Schd.	Over.	Schd.	Over.
	cm									
Inputs										
Precipitation	63.6	63.6	50.3	50.3	71.6	71.6	37.5	37.5	111.5	111.5
Irrigation	0.0	0.0	2.5	50.8	0.0	0.0	0.0	64.8	1.3	57.8
Total	63.6	63.6	52.8	101.1	71.6	71.6	37.5	102.3	112.8	169.3
Losses										
ET	28.9	28.9	18.3	27.1	26.2	26.2	26.6	36.0	50.0	59.1
Percolation	26.2	25.7	26.8	68.4	21.8	22.0	9.2	44.9	42.0	80.5
	(3.8)§	(3.0)	(2.4)	(4.0)	(6.9)	(5.9)	(3.3)	(7.2)		
Runoff	2.0	2.0	0.2	0.2	0.0	0.0	0.0	0.0	1.1	1.1
Total loss	57.1	56.6	45.3	95.7	48.0	48.2	35.8	80.9	93.1	140.7
% Unaccounted	10.2	11.0	14.2	5.3	33.0	32.7	4.5	20.9	17.5	16.9

† Sched. = scheduled irrigation treatments. ‡ Over. = overwatered treatments. § ( ) = standard deviation of nine plots.

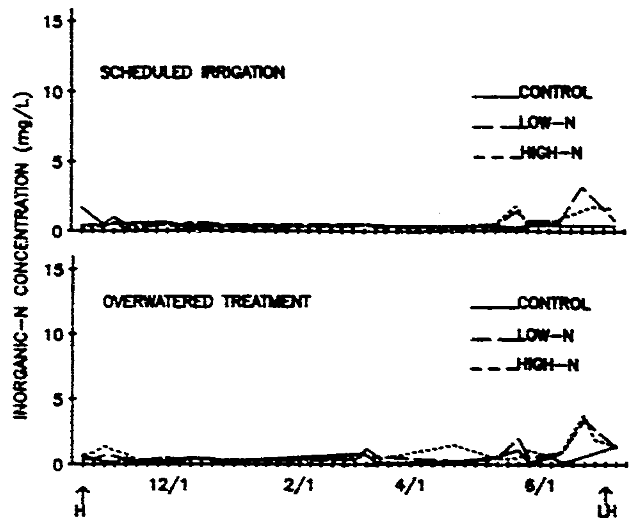

Fig. 1. Inorganic-N concentrations by event, 1 Oct. 1984 to 30 June 1985. ↑ = fertilization date; H = fertilizer application to high N treatment; and L = fertilizer application to low N treatment.

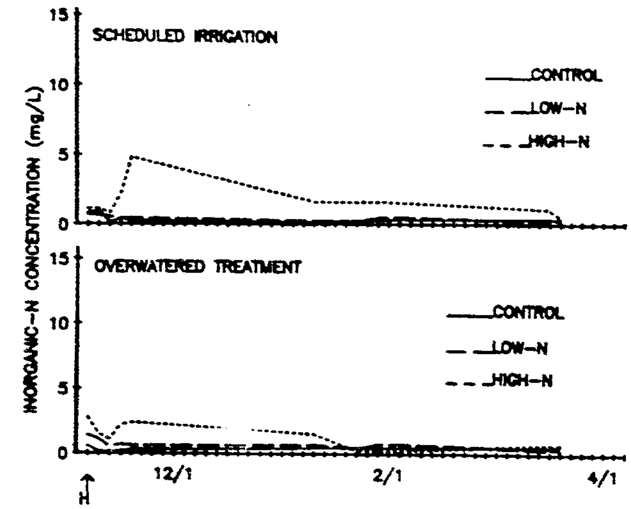

Fig. 3. Inorganic-N concentrations by event, 9 Oct. 1985 to 1 June 1986. ↑ = fertilization date; H = fertilizer application to high N treatment; and L = fertilizer application to low N treatment.

Corrected soil-water percolation averaged 42 cm yr^{-1} on the scheduled irrigation treatments compared to 81 cm yr^{-1} on the overwatered treatments (Table 1). Approximately 63% of the soil-water percolate from the root zone occurred during the nonirrigated periods (October–June) on the scheduled treatments vs. 29% for the overwatered treatments. During the two irrigation periods studied, soil-water percolate from the overwatered treatments was approximately three times that generated by the scheduled irrigation plots.

The potential for off-site chemical losses depends on both the frequency and quantity of percolation from the root zone. On the overwatered treatments an average of 45 events per summer generated percolate below the root zone. In contrast, percolation events from the scheduled treatments averaged nine per summer.

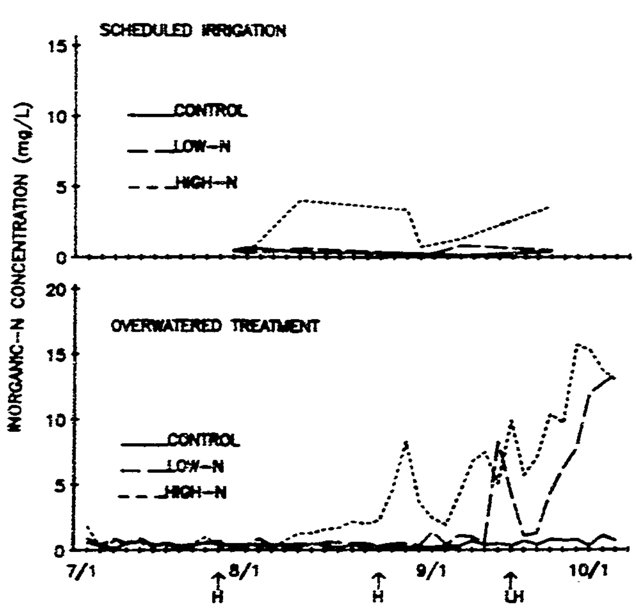

Fig. 2. Inorganic-N concentrations by event, 1 July to 8 Oct. 1985. ↑ = fertilization date; H = fertilizer application to high N treatment; and L = fertilizer application to low N treatment.

Soil-water N Concentrations

The safest approach to assess the potential for NO_3 contamination of groundwater from samples of soil-water percolate is to assume that all forms of inorganic N that leached from the root zone would eventually convert to NO_3-N. Nitrate-N comprised 77% of the total inorganic N in leachate from all plots during the study. Mean NH_4-N concentrations were <0.52 mg L^{-1} on all

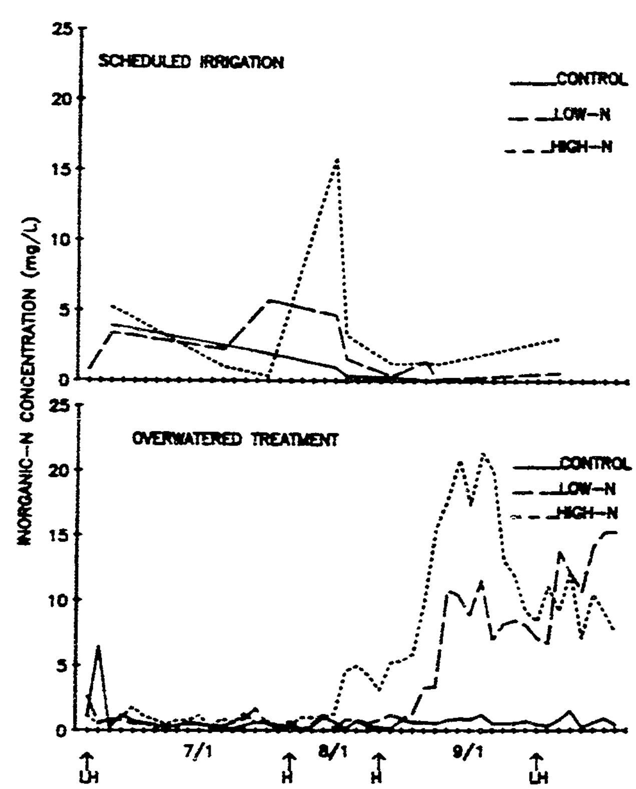

Fig. 4. Inorganic-N concentrations by event, 2 June to 1 Oct. 1986. ↑ = fertilization date; H = fertilizer application to high N treatment; and L = fertilizer application to low N treatment.

treatments (data not shown). The following results and discussion will not differentiate between inorganic-N fractions.

Mean inorganic-N concentrations for each event that generated percolate below the root zone are depicted in Fig. 1 to 4. The control treatments maintained percolate concentrations <2 mg L^{-1} during all events. The only percolation events from the controls with concentrations in excess of 1 mg L^{-1} occurred in late spring.

Although the high N treatments received 97 kg N ha^{-1} each fall, the percolate concentrations of inorganic N never exceeded 5 mg L^{-1} (one-half of the drinking water standard of 10 mg L^{-1}) throughout the succeeding fall, winter, or spring (Fig. 1 and 3). Dramatic increases in percolate concentrations of inorganic N were observed in late summer on the overwatered fertilized plots (Fig. 2 and 4). Percolate concentrations on the high N, overwatered treatments began to rise above control concentrations following an application of 28 kg N ha^{-1} in mid-July. Inorganic-N concentrations in excess of 10 mg L^{-1} were observed for the first time each summer following the August application of fertilizer on the high N treatments. Elevated concentrations were maintained through the September application each year.

The low N, overwatered treatment also generated elevated inorganic-N concentrations in late summer, although no fertilizer had been applied since late spring. Following the September application (49 kg N ha^{-1}), concentrations rose steadily until the end of each irrigation period (Fig. 2 and 4). Cisar (1986), through analysis of growth and N content of clippings, found that N uptake by Kentucky bluegrass declined in August and September. Applying N fertilizer when plant uptake is reduced could generate excess soluble N in the root zone and enhance the possibility for waterborne losses of N.

Soil-water percolation from the scheduled irrigation treatments occurred from random, large precipitation events, producing a spotty record of inorganic-N concentrations in percolate during each irrigation period. Generally, the high N, scheduled irrigation treatment generated percolate with higher inorganic N than the low and control treatments, although concentrations never approached those generated from the overwatered treatments (Fig. 2 and 4). The low N, scheduled irrigation treatment did not exhibit elevated concentrations in any of the events that occurred throughout August and early September. No percolate events were generated by precipitation in the month following the September fertilizer application, precluding evaluation of its effect on percolate concentrations on the scheduled treatments.

Event based analysis is not sufficient to evaluate the impact of N percolation from home lawns. Mean flow weighted concentrations are given in Table 2. Largely as a result of elevated concentrations during the irrigation periods, the fertilized overwatered treatments had significantly higher mean annual flow weighted concentrations than the controls. No significant differences were observed between the scheduled, fertilized treatments and the controls during any period. On all treatments, mean seasonal and annual flow weighted concentrations were well below the drinking water standard of 10 mg NO_3-N L^{-1}, suggesting that fertilizer practices in common use for home lawns do not pose a threat to potable water supplies.

Table 2. Mean flow weighted, inorganic-N concentrations in soil-water percolate.

N treatment†	Irrigation	Nonirrigation period 1 Oct. 1984–30 June 1985	Irrigation period 1 July–8 Oct. 1985	Nonirrigation period 9 Oct. 1985–1 June 1986	Irrigation period 2 June–1 Oct. 1986	Mean annual
		mg L^{-1}				
Control	Scheduled	0.50 (0.03)‡	0.24 (0.14)	0.41 (0.08)	1.49 (0.94)	0.51 (0.35)
Control	Overwatered	0.38 (0.00)	0.35 (0.03)	0.36 (0.07)	0.46 (0.19)	0.36 (0.09)
Low	Scheduled	0.46 (0.05)	0.20 (0.04)	0.52 (0.11)	3.47 (4.69)	0.87 (1.01)
Low	Overwatered	0.56 (0.15)	1.61 (0.40)	0.93 (0.36)	3.08 (1.74)	1.77 (0.87)
High	Scheduled	0.51 (0.09)	1.05 (1.68)	1.48 (1.33)	2.96 (0.55)	1.24 (0.96)
High	Overwatered	0.66 (0.20)	4.85 (0.99)	1.75 (1.31)	5.60 (1.49)	4.02 (1.01)
LSD ($P \leq 0.05$)		0.30	1.19	1.25	4.06	0.95

† Control = 0.0 kg N ha^{-1} yr^{-1}; Low = 98 kg N ha^{-1} yr^{-1}; High = 244 kg N ha^{-1} yr^{-1}.
‡ ( ) = standard deviation of three replicates.

Table 3. Annual and seasonal percolation losses of inorganic N.

N treatment†	Irrigation	Nonirrigation period 1 Oct. 1984–30 June 1985	Irrigation period 1 July–8 Oct. 1985	Nonirrigation period 9 Oct. 1985–1 June 1986	Irrigation period 2 June–1 Oct. 1986	Mean annual
		kg ha^{-1}				
Control	Scheduled	1.31 (0.10)‡	0.63 (0.39)	0.89 (0.11)	1.37 (1.10)	1.88 (1.01)
Control	Overwatered	0.98 (0.00)	2.39 (0.19)	0.79 (0.30)	2.08 (0.41)	2.79 (0.33)
Low	Scheduled	1.21 (0.36)	0.54 (0.13)	1.13 (0.10)	3.19 (3.95)	3.04 (2.91)
Low	Overwatered	1.43 (0.30)	11.00 (1.73)	2.06 (0.98)	13.81 (7.12)	13.65 (5.06)
High	Scheduled	1.34 (0.20)	2.80 (3.41)	3.24 (2.89)	2.73 (0.51)	4.87 (3.23)
High	Overwatered	1.69 (0.53)	33.17 (6.74)	3.86 (2.56)	25.15 (8.26)	31.94 (7.67)
LSD ($P \leq 0.05$)		0.77	5.64	2.74	9.30	5.04

† Control = 0.0 kg N ha^{-1} yr^{-1}; Low = 98 kg N ha^{-1} yr^{-1}; High = 244 kg N ha^{-1} yr^{-1}.
‡ ( ) = standard deviation of three replicates.

Nitrogen Percolation Losses

Mean annual and seasonal losses of N in soil-water percolate are displayed in Table 3. Mean annual losses ranged from approximately 2 kg ha^{-1} for the scheduled irrigation control treatment to 32 kg ha^{-1} for the high N, overwatered treatment.

Nitrogen losses from the fertilized plots with scheduled irrigation were not significantly different from control plots, although mean N losses from the high N, scheduled irrigation treatment were more than double the losses from the control treatment. Overwatering, however, did significantly increase N losses from fertilized plots. The low and high N overwatered treatments generated losses five- and 11-fold greater than the overwatered control plots, respectively. Most of the additional N lost from the overwatered, fertilized plots occurred during the summer irrigation period. Summer losses accounted for 88 and 91% of the annual N lost from the overwatered low and high N treatments, respectively.

Nitrogen Runoff Losses

Overland runoff occurred on only two storm events during the 2 yr of monitoring. Runoff depths are summarized in Table 1. Both of the runoff events resulted from unusual climatic conditions. Surface runoff in one of the events was generated by rainfall on frozen ground with snow cover. Extremely wet conditions preceeded the other storm (12.5 cm) that generated runoff. Although a total of 26.4 cm of precipitation occurred within 1 wk, depth of runoff was <0.2 cm.

The sandy loam soil of the study site has a high infiltration rate (SCS hydrologic group A) and is characteristic of the majority of soils overlying prime aquifers in Rhode Island. Using the SCS curve number method (SCS, 1972), a 24-h storm would have to exceed 10 cm to generate any runoff on these turf covered sites. Storms of this magnitude in Rhode Island have a return interval of 5 yr (Hershfield, 1961); therefore, overland runoff is not expected to be a major pathway for water loss from home lawns on the outwash soils of Rhode Island. On more impermeable soils, overland runoff could be expected to generate more substantial N losses.

For the two overland runoff events, concentrations of inorganic N for all the treatments ranged from 1.1 to 4.2 mg L^{-1}. Mean annual losses from overland flow ranged from 0.1 to 0.4 kg ha^{-1} hr^{-1} and comprised <7% of total waterborne loss of inorganic N from any treatment during the study.

CONCLUSIONS

Leaching losses of inorganic N from turf subjected to fertilization and watering practices associated with home lawn care do not appear to pose a threat to drinking water aquifers. Although individual events occurred that exceeded the U.S. drinking water standard, seasonal and annual flow weighted mean concentrations were always less than one-half the standard. In coastal watersheds, fertilized home lawns may contribute to the degradation of bay and estuarine water quality since substantial increases in N loadings can result from overwatered fertilized lawns.

The results of the scheduled irrigation treatments or the overwatered treatments should not be equated with the expected losses from home lawns. The irrigation regimes examined in this study represent the two extremes of home lawn water management. Currently homeowners can not be expected to schedule irrigation as carefully as this study. In practice, some degree of overwatering will occur due to the lack of homeowner knowledge regarding soil moisture status. In areas concerned with NO_3 contamination of groundwater, late summer applications could be minimized and homeowners encouraged to limit the quantity and frequency of watering. Overland runoff from turfgrass on permeable soils was not a major contributor to total inorganic-N losses.

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