AN ASSESSMENT OF THE IMPACT OF SEPTIC LEACH FIELDS, HOME LAWN FERTILIZATION AND AGRICULTURAL ACTIVITIES ON GROUNDWATER QUALITY

Prepared for the
NEW JERSEY PINELANDS COMMISSION

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ABSTRACT

The available information on the potential for movement of nutrients, pathogens and organic compounds to the groundwater from septic leach fields, fields irrigated with secondary treated sewage effluent, and fertilized lawns and agricultural fields under the New Jersey Pinelands soils was reviewed and utilized to develop predictions of pollution hazards associated with these practices.

The high permeabilities, low pH and low phosphorus adsorption capacities of the soils minimize their ability to prevent migration of inorganic pollutants to the groundwater. Additionally, many of the soils have high water tables which would allow pathogenic organisms to enter the aquifer and are thus unsuitable for septic fields.

For septic leach fields, the soils with rapid infiltration belonging to hydrological group A cannot be used because of the potential for pathogen migration to the groundwater. In the New Jersey Pinelands, the soils belonging to hydrological groups C and D cannot be used because of seasonally high water tables, and thus only soils in hydrological group B with groundwater tables greater than 6 ft from the surface may be acceptable. The potential for denitrification is low in these soils and only small amounts of nitrogen may be taken up by vegetation. Therefore, much of the nitrogen is likely to eventually move to the groundwater. For drinking water use, the U.S. Public Health Service requires that nitrate nitrogen be less than 10 ppm, while the New Jersey State Department of Environmental Protection has set an acceptable limit for the Pinelands area of 2 ppm. The average nitrate nitrogen content of the existing groundwater is 0.17 ppm. To prevent the average annual nitrate nitrogen concentration in the groundwater from rising more than 2.0 ppm, septic field spacing for a typical family would require lot sizes of 3.0 acres. The 0.17 ppm leachate nitrogen limit would require 10.5 acres. Phosphorus will likely break through the soils in group B which have permeabilities of between 2 and 6 inches per hour within a year after septic systems are put into use. For soils in group B with permeabilities of 0.2 to 2 inches per hour, the

breakthrough may require several years. In either case, procedures need to be employed, including perhaps the use of alum to precipitate the phosphorus.

Home lawn fertilization would be expected to add large amounts of nitrogen to the groundwater, and even larger acreages than are required for septic fields would need to be set aside to allow dilution of the leach water to acceptable levels. It may be necessary to consider a prohibition on home lawns and instead encourage the use of native vegetation which would not require fertilization.

Irrigation with secondary treated sewage effluent could provide effective disposal without elevating the groundwater concentration above 2 ppm only if large acreages were set aside for the purpose.

Fertilization of the limited number of acres used for agricultural production is expected to contribute only small amounts of nitrogen to the groundwater if fertilizers are applied in split applications and if management practices are designed to meet, but not exceed, the crop needs.

Information is lacking and needed on (1) the groundwater recharge volumes under soils in the different hydrological groups, (2) the phosphorus adsorption capacity of the subsoil materials in soils in hydrological group B, (3) the influence of septic effluent on the soil pH and that of the water entering the groundwater, and (4) the concentrations and mobilities of biocides found in septic effluent.

1.0 INTRODUCTION AND OBJECTIVES

The question of housing development in the New Jersey Pinelands has raised the issue of the suitability of native soils and hydrogeology for domestic waste disposal. While the principles of onsite septic systems rely on the soil as the final treatment in wastewater renovation, indiscriminant utilization of these systems may pose a potential for degeneration of groundwater quality through excessive nutrient and contaminant leaching. To provide a viable development strategy for the Pinelands region, an understanding of the sources, interactions, and environmental fate of domestic waste constituents is essential. Additional modifiers of the system including lawn and garden fertilization, household pest management, fertilization of agricultural fields and irrigation disposal of secondary treated sewage effluent must also be considered.

Water has been said to be the outstanding resource of the Pinelands and most groundwater recharge originates from direct percolation of rainwater on an area-wide basis (Rhodehamel, 1970). The region is also generally characterized by acid sandy soils overlying shallow water tables. Therefore, one would intuitively suspect that careful home and waste management would be a necessary requirement for groundwater protection, particularly since water supply wells would be located in the same aquifer.

The "standard" septic system forms the core of this investigation. The basis for such choice stems from the wide preference for this method of waste treatment in a variety of soils and climates and from the extensive experimental and regulatory information available on septic tank-absorption trench systems. An understanding of the advantages and restraints of septic system use may aid in determining the proper approach to waste treatment in the Pinelands.

From the above considerations, this report is an exploration of available information regarding the movement and fate of septic tank effluent constituents in soils and groundwater with particular respect to the Pinelands area. After additional review of other possible contaminant sources such as lawn fertilization, a conceptual model is specified which delineates the suitability of soils for onsite septic tank-absorption trench systems. Management and alternative technology recommendations are proposed which may resolve any shortcomings of the "standard" septic system.

2.0 SEPTIC SYSTEM PRINCIPLES

In areas where there is no access to centralized sewage collection and treatment, the conventional method of domestic waste disposal is the septic tank-soil absorption system. The U.S. EPA (1973) reported a 1970 survey which estimated that 32 million people, or about 15% of the U.S. population were served by such onsite sewage treatment. A study of domestic waste management practices in the Northeast by Miller et al. (1974) revealed that approximately 1,300,000 people in New Jersey were served by these systems. Consequently, developments in an unsewered region such as the Pinelands look to the septic system as the first alternative for waste treatment.

2.1 The "Standard" Septic System

The term "standard" septic system as used herein refers to a septic system constructed according to criteria established by the New Jersey State Department of Environmental Protection (1978). These standards consider the disposal trench to be the preferred septic tank effluent distribution system for most locations. A comparison of New Jersey regulations with recommendations set by the U.S. Public Health Service (1972) for absorption trench area calculations (Table 2.1) indicates that the New Jersey criteria are less stringent for more permeable soils with the exception that extremely permeable soils necessitate special evaluation and agency approval. The resultant system constructed by New Jersey standards provides a smaller effective soil volume for sewage effluent renovation than the federal government criteria would provide.

Extensive studies of onsite waste management in Wisconsin have led to design recommendations which account for reduced infiltration rates due to soil clogging (Otis et al., 1977). Comparison of the Wisconsin criteria with New Jersey standards (Table 2.2) indicates that calculation of leach field size using actual water flux in an operative system require larger trench bottom areas than specified by the conventional New Jersey methods.

TABLE 2.1 Comparison of U.S. Public Health Service and New Jersey criteria for determination of required areas for disposal trenches.

Percolation Rate (min./inch)		Minimum Bottom Trench Area (ft^2/bedroom)	
U.S.	New Jersey	U.S.	New Jersey
$\leq$ 1	-	70	-
2	-	85	-
3	$\leq$ 3	100	Special Approval
4	-	115	-
5	$\leq$ 5	125	70
10	6-10	165	95
15	11-15	190	120
30	26-30	250	190
45	41-45	300	265
60	56-60	330	340
> 60	> 60	Unsuitable	Unsuitable

2.1.1 Function of the System

With the objective of renovating wastewater while minimizing surface and groundwater contamination, the three basic components of a "standard" system are the septic tank, the effluent distribution field and the native soil (Figure 2.1). Initial treatment by the septic tank functions to remove most of the suspended solids by sedimentation while trapping any low density floating scum by means of baffles within the tank. Anaerobic conditions also initiate chemical and biological alteration of sewage constituents. Partially renovated effluent is then discharged to the soil via the drainfield which functions to distribute the effluent load as evenly as possible over the entire area. The soil acts as the final treatment of the wastewater before loss to either evapotranspiration or deep percolation. Miller and Wolf (1977) have listed the functions of the soil to be (1) physical filtering of suspended solids and microorganisms, (2) sorption by ionic exchange, chelation and pH induced precipitation, and (3) chemical and biological oxidation.

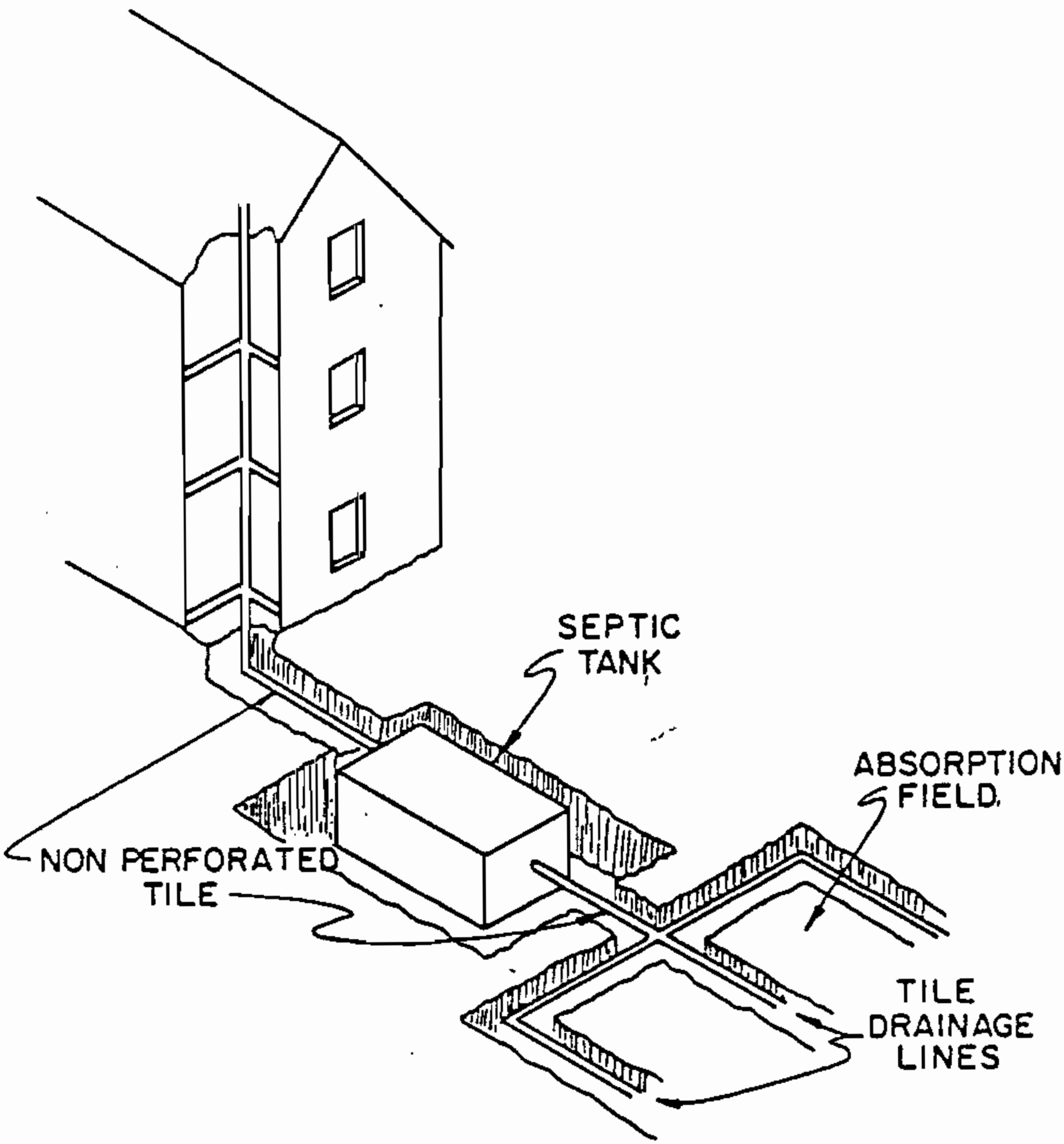

FIGURE 2.1 Layout of the principle components of a standard septic tank-soil absorption system (U.S. Public Health Service, 1972).

TABLE 2.2 Comparison of New Jersey criteria with recommendations by Otis et al. (1977) for determination of required areas for disposal trenches.

Percolation Rate (min./inch)		Minimum Bottom Trench Area (ft^2/gal/day)	
New Jersey	Otis (estimated)	New Jersey	Otis
5	-	0.49	-
6-10	10	0.64	0.83
11-15	-	0.80	-
16-20	-	0.96	-
21-25	-	1.12	-
26-30	10-30	1.28	1.67
31-35	-	1.44	-
36-40	-	1.60	-
41-45	30-45	1.75	2.00*+
46-50	-	1.95	-
51-55	-	2.15	-
56-60	-	3.35	-
>60	45-90	Not Acceptable	2.22*+

* Shallow trench only.
\+ Should not be applied to soils with expandable clays.

2.1.2 Design Limitations

Two important shortcomings of the "standard" septic system are related to the conventional methods used to design the required absorption field area. The design involves performing a percolation ("perc") test at the site to determine the hydraulic loading capacity of the soil which receives the effluent. Evidence from numerous investigators (Sawhney and Starr, 1977; Bouma et al., 1972; Walker et al., 1973) has revealed that within a few months of service an organic slime or "crust" forms at the interface of the trench fill material and the undisturbed soil profile which limits absorption to well below the soil's capacity. The crust forms anaerobically as a result of continuous inundation of infiltrative surfaces, and an absorption field area which does not account for crusting could result in ponding of effluent in the trenches with possible discharge on the land surface. However, crusting can be beneficial since the organic layer serves as an effective degradative filter to suspended and dissolved organic matter and because aerobic unsaturated flow conditions are encouraged in the soil beside and below the disposal trench (Walker et al., 1973). Laak (1977) has shown that a long-term equilibrium permeability is eventually established and is regulated by pretreatment level and pollutant load.

A second inadequacy of the "perc" test as used in absorption field design is that only the saturated hydraulic conductivity of the soil is taken into account (U.S. Public Health Service, 1972). As previously noted, flow below a crusted absorption trench is usually unsaturated, but probably of greater importance is that such a design does not consider the effectiveness of various soils as treatment media. Bouma (1975) has indicated that soils vary in their capacity to remove pollutants from the percolating effluent, with sands being relatively poor purifiers (Figure 2.2). Thus, in the case of a sandy soil, the "perc" test design criteria calls for relatively small field area based on saturated hydraulic conductivity. In fact, the limited capacity of a sand to renovate wastewater may dictate a large absorption field to optimize pollutant attenuation, particularly where a permanent or fluctuating shallow water table limits the depth of unsaturated soil available for wastewater renovation.

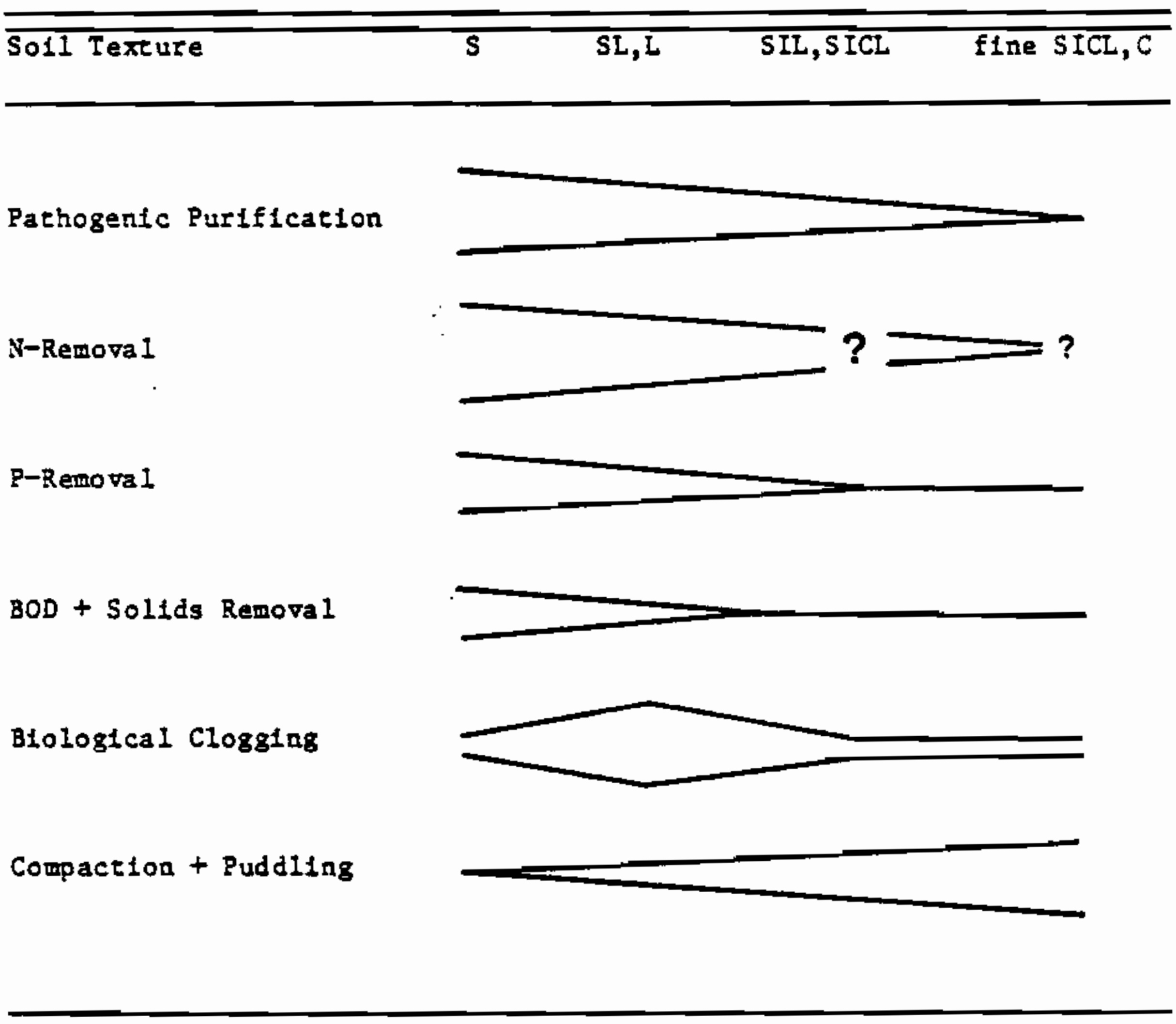

FIGURE 2.2 Suitability and limitations of various soil types for renovation of septic tank effluent, where limitations and potential problems increase with band width (Bouma, 1975).

2.1.3 Flow in Soils Beneath Leaching Fields

A key to the effectiveness of the "standard" septic system is the formation of the previously mentioned crust or "mat." Reports indicate that crust development in a new absorption trench can occur within about 100 days of the initial system use (Sawhney and Starr, 1977). McGauhey and Krone (1967) observed that the infiltration capacity of these crusts generally appears to be unrelated to soil properties. Aerobic conditions due to crusting have been observed in sandfill within 15 cm below a crusted absorption trench (Couture, 1978). In an in situ study of several septic systems, Walker et al. (1973) documented soil conditions near leaching fields after crust formation. Water flow in the soils, sands and loamy sands, was unsaturated with soil moisture tensions of 20 to 27 mb within 30 cm of the trenches, and a flow rate of 8 cm day^{-1}. These values corresponded to a phase distribution of 13% water and 25% air. Oxygen measurements showed that conditions were suitable for nitrification of effluent nitrogen. Unsaturated conditions can be maintained in sandy soils under crusted trenches where water tables are at least 0.9 m below the trenches (Bouma et al., 1972). Thus, noted Miller and Wolf (1977), the transmission and renovation capability of a soil is dependent upon its interaction with waste effluent constituents under an unsaturated flow regime.

The advantages of unsaturated flow of wastewater are the increased contact time and closer proximity of waste constituents with soil particles, organic matter, and microorganisms. An adequate supply of oxygen is also provided for plant root survival and uptake of nutrients. The effect of contact time and proximity is to enhance the purification of the sewage effluent since attenuation of pollutants is achieved through filtration, sorption, and oxidation.

2.2 Characteristics of Septic Tank Effluent

Septic tank effluent is subject to a wide variability in composition dependent upon several factors. Table 2.3 illustates a typical analysis of raw sewage entering a septic tank. The sewage may differ according to system usage (ie. business or school versus home), personal habits, and management practices. The size and design of the septic tank, the degree of sludge and scum accumulation in the tank, tank-water detention time, and waste flow rates determine the effectiveness with which contaminants are removed from the waste stream. Miller and Wolf (1977) have listed the major components of septic effluent to include:

(1) Biological contaminants (eg. bacteria, virus)
(2) Chemical contaminants
    (a) Organics (eg. suspended, dissolved)
    (b) Inorganics (eg. N, P, K, salts)

Table 2.4 lists concentrations of constituents in septic tank effluent as found by several investigators.

TABLE 2.3 Characteristics of raw sewage entering septic tanks.

Constituent (mg/1*)	Viraraghavan & Warnock (1974) Range	Viraraghavan & Warnock (1974) Average	Baumann et al. (1978)
pH	6.60-9.10	7.63	-
Total suspended solids	100-372	161	300
Volatile suspended solids	32-344	117	250
BOD, 5-day @ 20°C	397-582	484	200
TOC	120-1040	335	-
Total phosphates as PO_4	2.5-45.0	18.9	-
Nitrogen, total	-	-	50
free ammonia	6.3-226.6	71.3	30
organic	-	-	20
nitrate	0.00-0.12	0.12	0.20
Chlorides	21.0-296.0	162.7	100
Coliform (MF count per 100 ml)	$1100-22 \times 10^6$	6.27×10^6	
Fecal coliform (MF count per 100 ml)	$600-6 \times 10^6$	1.05×10^6	
Fecal streptococci (MF count per 100 ml)	$360-0.6 \times 10^6$	0.12×10^6	
Pseudomonas aeruginosa (MPN/100 ml)	14-92000	15484	

*Except for pH and where noted.

TABLE 2.4 Characteristics of septic tank effluent as reported by various investigations. (All paramaters expressed in mg/l except pH.)

Reference	NH_4-N	NO_3-N	Organic N	Total N	PO_4	Total P	pH
Andreoli (1979)	41.0	0.1	20.2	61.3	-	-	-
Aulenbach (1972)	-	-	-	-	22.0 22.8	25.4 25.8	-
Biggar (1969)	25.0	0.15	10.0	-	20.0	-	-
Bennett (1969)	-	-	-	-	-	10.4	-
Brown (1977)	-	-	-	-	6.94	8.18	-
Bouma et al. (1972)	-	-	-	80.0**	-	10.00	-
Bouwer (1973)	20-40	0-3	2-6	-	7-12	-	-
Dudley and Stephenson (1973)	35.0	0.5	10.0	45.5	20-37	5.4-61	-
Hickey and Duncan (1966)	37.0	-	3.4	40.4*	-	-	7.26

TABLE 2.4 continued.

Reference	NH_4-N	NO_3-N	Organic N	Total N	PO_4	Total P	pH
Hill (1972)	-	5.0	-	-	-	8.0	-
Lake George (1971)	-	-	-	-	-	8.2	-
Magdoff et al. (1974)	-	-	-	42.0	-	21.0	7.50
Otis and Boyle (1976)	35.6	0.5	-	52.2	11.6	15.2	-
Otis et al. (1977)	19.2	0.3	4.4	23.9	8.7	10.2	-
Popkin and Bendixon (1968)	24.6	0.21	5.6	30.4	-	-	-
Preul (1967)	25.0	0.1	10.0	35.1	-	20.0	7.25
Robeck et al. (1964)	22.0	0.12	5.4	27.5	-	-	7.80
Reneau (1979)	-	-	-	-	-	10.8	-
Sanborn	-	-	-	-	20.8	26.4	-

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TABLE 2.4 continued.

Reference	NH_4-N	NO_3-N	Organic N	Total N	PO_4	Total P	pH
Sauer et al. (1976)	20.0	0.3	-	20.3*	9.8	-	-
Sawhney (1977)	-	-	-	-	-	5.5	-
Silbermann (1977)	27.0	0.2	8.0	35.2	-	18.0	7.1
Stewart (1979)	-	-	-	-	-	9.5	-
Thomas and Bendixon (1969)	25.4	< 0.1	7.9	33.4	-	-	7.7
Viraraghaven and Warnock (1976)	97.0	0.026	-	97.03*	-	11.6	6.9
Walker et al. (1973)	66.3	-	13.7	80.0**	-	-	6.4
Mean Values	35.3	0.65	8.6	44.6	16.0	16.4	7.0

*Calculated Total N, NH_4-N and Organic N where NO_3 & NO_2-N were not reported and are assumed to equal 0.

**Total-N reported by Bouma et al. (1972) and Walker et al. (1973) are, apparently, derived from the same data base, hence, their values were considered as a single value in the calculation of mean Total-N.

Phosphorus concentrations cited in the table vary widely dependent upon septic system usage. Sikora and Corey (1976) noted that the phosphorus in effluent is largely from detergents which contain phosphorus and from human excreta, and the use of low phosphorus detergents can reduce P concentrations by as much as one-half. Anaerobic digestion within septic tanks is responsible for converting most effluent phosphorus to soluble orthophosphate. Two investigations by Magdoff et al. (1974) and Otis et al. (1975) found more than 85% of the effluent P in the orthophosphate form and total P concentrations of 15.6 to 24.5 mg/l and 11.0 to 31 mg/l, respectively.

The major sources of nitrogen in raw sewage are feces and urine which Witt et al. (1975) characterized to contain urea, uric acid, ammonia, undigested proteinaceous foodstuffs, and bacterial cells. Soluble ammonium is the predominant form, with amounts reported in Table 2.4 averaging 80% and ranging from 67 to 100% of the total N. Organic N accounts for most of the remainder while nitrate and nitrite each constitute less than two percent of the total. Walker et al. (1973), investigating nitrogen transformations in and around septic systems, found that much of the organic N was retained by the crust interface of the absorption trench and soil. Consequently, practically all nitrogen entering the soil from absorption trenches initially appears as ammonium.

3.0 SOILS

The potential for the movement of pollutants to the groundwater from the septic leach fields, treated sewage effluent irrigation systems, home lawn fertilization, and cropland fertilization will depend greatly on the properties of the soils to which the materials are added.

Pertinent properties of the soils mapped in the New Jersey Pinelands are summarized in Table 3.1. The seasonal water tables in all the soils in hydrological groups C and D rise to within six feet of the surface. Additionally, two group A soils (Lakehurst and Pemberton), three group B soils (Hammonton, Klej, and Nixonton), and three B/D soils (Pasquotank, St. Johns, and Weeksville) have seasonally high water tables. The possibility of movement of microbial pathogens through these soils eliminates them from consideration for the installation of standard septic leach fields. Typically soils in hydrological group A do not have restrictive layers with permeabilities less than 2 inches per hour, while surface permeabilities are typically >6 inches per hour. Soils in hydrological group B typically have restrictive layers with permeabilities not less than 0.2 inches per hour, and the surface horizons typically have permeabilities greater than 0.6 inches per hour. Textures of the restricting horizons in these soils include sandy loam, sandy clay and loams. Percolation tests in the field may indicate the acceptability of soils in either of these hydrological groups for standard septic tanks based on hydrological considerations alone.

Soils in hydrological group A which can potentially accommodate septic systems include Evesboro, Fort Mott, Fripp, Galestown, Lakeland, Lakewood, and Tinton.

Potentially acceptable soils in hydrological group B include Aura, Collington, Colts Neck, Downer, Freehold, Phalanx, Sassafras, Westphalia, and Woodmansie.

Typical soil pH at the 2 ft to 2-1/2 ft depth where absorption trenches are installed ranges from 3.6 to 6.5 with the exception of the Fripp soil which in some cases has been reported to be as high as pH 7.8.

The high sand and low clay contents in these soils result in very low cation exchange capacities. Although specific data on the soils is lacking, data on similar soils from other locations indicates that the cation exchange of the sandy soil in hydrological groups A and B should range from a low of 1.5 meq/100 g for the sand layers to a high of 4 meq/100 g for the layers containing more fine textured material.

TABLE 3.1 Characteristics of the soils mapped in the New Jersey Pinelands.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Adelphia	P	C	1.5-4	0-14	SL,FCL,SIL	3.6-5.0	0.2-6.0
			apparent	14-37	SCL,L	3.6-5.0	0.2-2.0
			Jan-Apr	37-60	SR,LS,SL	3.6-5.0	0.6-6.0
Alluvial Land		D	0-1.0	0-10	L,SIL,SL	>4.5	0.6-6.0
				10-30	SIL	4.5-5.0	0.6-2.0
				30-60	Stratified SL,L,SIL	4.5-5.0	0.6-6.0
Atsion		D	0-1.0	0-18	S,FS	3.6-5.0	6.0-20
			apparent	18-24	LS,S,SL	3.6-5.0	2.0-6.0
			Nov-Jan	24-40	S,LS	4.5-5.0	6.0-20
				40-60	SR,S-SIL	4.5-5.0	6.0-20
Aura	P,S	B	>6.0	0-22	SL,LS,L	3.6-5.0	0.2-6.0
				0-22	GR-SL	3.6-5.0	0.2-6.0
				22-59	GR-SL,SCL, GR,SCL	3.6-5.0	0.2-2.0
				59-72	S,GR-S,SCL	3.6-5.0	0.2-6.0
Bayboro		D	0-0.5	0-14	FSL	4.5-5.5	2.0-6.0
				0-14	L,CL	4.5-5.5	0.6-2.0
				14-64	CL,SC,C	4.5-5.5	0.06-0.2
Berryland	U	D	0-0.5	0-12	S,LS	3.6-4.4	6.0-20
			apparent	12-20	S,LS	4.5-5.0	2.0-6.0
			Oct-Jan	20-30	S	4.5-5.0	2.0-6.0
				30-40	S,LS	4.5-5.0	2.0-20
				40-72	SR,S,SL	4.5-5.0	2.0-20

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Colemantown	P	D	0-1.0	0-10	FSL,L,SCL	3.6-5.5	0.2-2.0
			perched	10-30	SC,C,CL	4.5-5.5	0.06-0.2
			Oct-Jun	30-60	L,CL,SL	4.5-5.5	02.-0.6
Collington	P,S	B	>6.0	0-13	SIL,FSL,SL	3.6-5.5	0.6-2.0
				0-13	LS	3.6-5.5	6.0-20
				13-22	SL,SCL,CL	3.6-5.5	0.2-20
				32-60	SR-SCL-S	3.6-5.5	0.2-6.0
Colts Neck	P,S	B	>6.0	0-10	SL,LS,L	4.5-6.0	0.6-2.0
				10-35	SL,SCL,L	4.5-6.0	0.6-2.0
				35-60	LS,S,SL	4.5-6.0	2.0-20
Donlonton	P	C	1.5-2.0	0-12	FSL,L,LFS	4.5-5.5	0.2-20
			apparent	12-50	SC,SCL	4.5-5.5	0.06-0.2
			Nov-May	50-60	FSL,SCL,CL	4.5-5.5	0.2-2.0
Downer	P,S	B	>6.0	0-18	LS,SL	3.6-5.0	0.6-6.0
				18-30	SL,GR-SL	4.5-5.0	0.6-6.0
				30-40	SR-S-GRLS	4.5-5.0	>2.0
				40-60	SR-GR-S-SCL	4.5-5.0	>2.0
Elkton	S	D	0-1.0	0-10	SIL,SL,SICL	3.6-5.5	0.6-2.0
			apparent	10-36	SIC,CL,C	3.6-5.5	<0.2
			Jan-Apr	36-60	SICL,FSL,C	3.6-5.5	0.2-6.0
Evesboro		A	>6.0	0-40	S,LS	3.6-5.0	6.0-2.0
				40-72	S,GR-S,SL	4.5-5.0	>2.0

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Fallsington	S	D	0-1	0-11	SL,FSL,L	3.5-5.5	0.6-6.0
			apparent	11-27	SL,L,SCL	3.6-5.5	0.6-2.0
			Dec-May	27-60	LS,S,SL	3.6-5.5	2.0-6.0
Fill Land		A	variable		S,GRS	4.0-5.0	6.0+
Fort Mott	P	A	>6.0	0-30	LS,S	3.6-5.5	6.0-20
				30-49	SL,SCL	3.6-5.5	0.6-6.0
				49-60	SR-S-LS	3.6-5.5	6.0-20
Freehold	P,S	B	>6.0	0-12	L,SL,FSL	3.6-4.4	0.2-6.0
				0-12	LS,LFS	3.6-4.4	6.0-20
				12-35	SL,SCL,L	4.5-5.0	0.6-2.0
				35-60	SR-LS-SL	4.5-5.0	0.6-6.0
Fripp		A	>6.0	0-5	FS,S	5.1-7.8	6.0-20
				5-80	FS,S	5.6-7.8	6.0-20
Galestown		A	>6.0	0-40	LS,S	3.6-5.5	>6.0
				40-60	S,LS,GR-S	3.6-5.5	>6.0
Hammonton	P	B	1.5-4.0	0-18	LS,SL	3.6-4.4	2.0-6.0
			apparent	18-36	SL,GR-SL	4.5-5.0	0.6-6.0
			Jan-Apr	36-60	SR-GR-S-SL	4.5-5.0	>2.0

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Holmdel	P	C	0.5-4.0	0-10	L,SL,FSL	3.6-4.4	0.6-2.0
			apparent	0-10	LS,LFS	3.6-4.4	2.0-20
			Dec-May	10-34	SL,SCL,L	4.5-5.0	0.6-2.0
				34-60	SR-FSL-S	4.5-5.0	0.6-2.0
Howell		C	>3.0	0-8	FSL,L,SIL	3.6-5.0	0.2-2.0
			apparent	8-14	SCL	3.6-5.0	0.2-2.0
			Nov-May	14-46	C,SICL,SIC	3.6-5.0	0.2-0.6
				46-60	C,SIL,GR,SCL	3.6-5.0	0.2-0.6
Keansbury		D	0-0.5	0-10	FSL,SL,L	<4.5-5.5	0.6-2.0
			apparent	10-30	SL,SCL,FSL	4.5-5.5	0.2-2.0
			Oct-Jun	30-60	SR-LS-SL	4.5-5.5	2.0-6.0
Keyport	P,S	C	1.5-4.0	0-10	SIL,L	3.6-4.4	0.2-0.6
			perched	0-10	SL,FSL	3.6-4.4	0.6-2.0
			Nov-May	10-60	SICL,CL,C	4.5-5.0	<0.2
Klej	S	B	1.5-2.0	0-39	LS,FS,LFS	3.6-5.0	6.0-20
			perched	39-47	S,FS	3.6-5.0	6.0-20
			Dec-Apr	47-60	SL,SCL,SC	3.6-5.0	0.06-0.6
Kresson		C	1.0-1.5	0-14	FSL,SL,L	3.6-5.5	0.2-6.0
			perched	0-14	LS	3.6-5.5	2.0-6.0
			Dec-May	14-30	C,CL,SC	3.6-5.5	0.06-0.2
				30-60	SR-SL-C	3.6-5.5	0.06-0.2

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Lakehurst		A	1.5-3.5	0-15	S,FS	3.6-5.0	6.0-20
			apparent	15-30	S,FS,LS	3.6-5.0	6.0-20
			Jan-Apr	36-60	S,GR-S,SL	4.5-5.0	6.0-20
Lakeland		A	>6.0	0-43	S,FS	4.5-6.0	>20
				43-90	S,FS	4.5-6.0	>20
Lakewood		A	>6.0	0-10	S,FS	3.6-5.0	6.0-20
				10-36	S,FS,LS	3.6-5.0	6.0-20
				36-60	S,GR-S,SL	3.6-5.0	0.6-20
Lenoir-Keyport		D	1.0-2.5	0-8	L,SIL,VFSL	4.5-5.5	0.6-2.0
			apparent	8-75	C,SIC,CL	4.5-5.5	0.06-0.2
			Dec-May				
Manahawkin Muck	P,U	D	+1-0	0-39	SP	3.6-5.5	0.2-6.0
			apparent	39-60	S,GR-S	4.5-5.0	2.0-6.0
			Oct-Jun				
Marlton	P,S	C	2.0-5.0	0-10	FSL,SL,L	<4.0-5.0	0.2-2.0
			perched	10-40	SC,SCL,C	4.5-5.5	0.06-0.2
			Nov-May	40-60	SR-SL-C	4.5-5.5	0.06-0.2
Matawan	P	C	2.0-3.0	0-20	LS	4.5-5.5	0.6-6.0
			apparent	0-20	SL,FSL	4.5-5.5	0.6-6.0
			Jan-Apr	20-38	SCL,SL,CL	3.6-5.5	0.06-0.6
				38-60	SR-S-CL	3.6-5.5	0.06-20

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Mullica		C	0-0.5	0-10	SL,L	3.6-5.0	0.6-2.0
			apparent	0-10	GR-SL,GR-L	3.6-5.0	0.6-6.0
			Dec-May	10-28	SL,SCL,GR-SL	3.6-5.0	0.6-2.0
				28-60	SR-GR-S-SCL	3.6-5.0	0.2-20
Nixonton	P	B	3-5	0-34	VFSL,L,SIL	5.1-6.5	0.6-2.0
			apparent	34-80	LFS,LS,FS	5.1-6.5	2.0-6.0
			Dec-Mar				
Pasquotank		B/D	0-1.0	0-14	FSL	<4.5-5.0	0.6-2.0
				14-30	VFSL,FSL	4.5-5.0	0.2-2.0
				30-60	SR-LS-S-FSL	4.5-5.0	0.6-2.0
Pemberton		A	1.0-4.0	0-24	S,LS	3.6-5.0	2.0-6.0
			apparent	24-34	SL,FSL,SCL	3.6-5.0	2.0-6.0
			Dec-May	34-60	SR-S-C	3.6-5.0	0.6-6.0
Phalanx		B	>6.0	0-6	LS,S	3.6-5.0	2.0-6.0
				6-22	SL,LS,CN-SL	4.5-5.5	0.6-6.0
				22-46	CN-SL,FL-LS, SCL	4.5-5.5	0.6-2.0
				46-72	S,LS,FL-S	4.5-5.5	2.0-6.0
Pocomoke		D	0-0.5	0-28	SL,FSL,LS	3.6-5.5	0.6-2.0
			apparent	28-40	LS,S	3.6-5.5	2.0-6.0
			Dec-May	40-60	SCL,SL,S	3.6-5.5	0.6-6.0

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Sassafras	P,S	B	>6.0	0-9	SL,L,FSL	3.6-5.5	0.6-6.0
				0-9	LFS	3.6-5.5	0.6-6.0
				9-40	L,SCL,SL	3.6-5.5	0.6-2.0
				40-70	GR-SL,FSL,S	3.6-5.5	0.6-20
Shrewsbury		D	0-1.0	0-14	FSL,SL,L	3.6-5.0	0.6-2.0
			apparent	14-32	SL,SCL,CL	3.6-5.0	0.2-2.0
			Oct-Jun	32-60	SR-LS-SL	3.6-5.0	2.0-20
St. Johns	U	B/D	0-3	0-22	SAND	4.5-5.5	>20
				22-42	SAND	4.5-5.5	0.6-2.0
				42-72+	SAND	4.5-5.5	>20
Tidal Marsh		D	0+	0-96	SIL,M	6.6-7.3	6.0+
				0-36	SIL,M	6.6-7.3	6.0+
				36-96+	S,GR-S	6.6-7.3	6.0+
Tinton		A	>6.0	0-26	S,FS,LS	3.6-5.0	0.6-6.0
				26-40	FSL,SL,SCL	3.6-5.0	2.0-6.0
				40-60	SR-S-SL	3.6-5.0	0.6-6.0
Weeksville		B/D	0-1	0-42	SIL,VFSL,L	4.5-5.5	0.6-2.0
			apparent	42-56	FSL,SL	4.5-5.5	0.6-2.0
			Dec-Mar	56-72	S,LS,LFS	4.5-5.5	2.0-6.0
Westphalia	P,S	B	>6.0	0-10	FSL,VFSL	3.6-5.5	0.6-2.0
				10-28	FSL,VFSL,LFS	3.6-5.5	0.6-2.0
				28-72	FS,LFS	3.6-5.5	0.6-6.0

TABLE 3.1 continued.

Series	Importance*	Hydro. group**	Depth to H_2O (ft)	Horizon depth (in)	Texture***	pH	Permeability (in/hr)
Woodmansie	P,S	B	>6.0	0-17	S,LS	3.6-4.4	6.0-20
				17-30	SL,GR-SL,SCL	3.6-5.5	0.6-6.0
				30-60	SR-GR-S-SCL	3.6-5.5	0.6-20
Woodstown	P	C	1.5-2.5	0-11	SL,FSL,L	3.6-5.5	0.6-6.0
			apparent	11-29	SCL,L,SL	3.6-5.5	0.6-2.0
			Feb-Apr	29-60	SL,LS,GR-S	3.6-5.5	0.6-6.0

* P = Prime farmland soil; S = statewide importance; U = unique.

** A = High infiltration rate, high water transmission rate, deep, well to excessively drained, coarse texture (low runoff potential).
B = Moderate infiltration rate, moderate water transmission rate, moderately deep to deep, moderately well to well drained, moderately fine to moderately coarse texture.
C = Slow infiltration rate, slow water transmission rate, moderately fine to fine texture or a layer which impedes water movement.
D = Very slow infiltration rate, very slow water transmission rate, and consists of clay soil of high swelling potential, soil with a permanent high water table, soil with a claypan, soil with a clay layer at or near the surface, or shallow soil over nearly impervious material (high runoff potential).

*** C = Clay; CN = channery; FL = flaggy; FS = fine sand(y); GR = gravel(ly); L = loam(y); P = peat; S = sand(y) SI = silt(y); SR = stratified; V = very.

4.0 INTERACTIONS OF POTENTIAL CONTAMINANTS WITH SOILS AND GROUNDWATER

The major contributors to the potential degradation of water resources in a developing area are effluents from domestic waste disposal and leaching and runoff losses from agricultural, commercial, and home land management practices. In view of the characteristics of septic tank effluent, parameters of interest are nitrogen, phosphorus, virus, bacteria, organics, pH, soluble salts, and other nutrients. Additional pollutant hazards may result from the use of agricultural chemicals, particularly nitrogen, phosphorus, lime and pesticides. An understanding of each of these potential contaminants will allow the development of a conceptual model of the environmental fate of domestic waste and will identify the process-limiting constituents.

4.1 Nitrogen

Nitrogen is an element of primary importance in the renovation of a sewage effluent because excessive concentrations in groundwater may be a public health hazard and can contribute to eutrophication of surface water. Additionally, certain forms of nitrogen are highly mobile in soils.

Nitrite and, indirectly, nitrate in drinking water may lead to methemoglobinemia which can impair oxygen transport in the blood (Lee, 1970), particularly in infants where gastrointestinal upsets can encourage reduction of nitrate to nitrite. U.S. Public Health Service (1962) standards set acceptable drinking water limits at 45 mg/l nitrate (10 mg/l NO_3-N) for infant feeding. A nitrate study group from the National Academy of Sciences (1972) compiled a listing of reported cases of methemoglobinemia (Table 4.1) which indicates the importance of contaminated groundwater in the occurrence of the disease. Although the preponderance of clinical cases of methemoglobinemia have been reported to be associated with NO_3-N concentrations of greater than 22 mg/l, two studies were noted as having found the disease where exposures were as low as 9 and 11 mg NO_3-N (National Academy of Sciences, 1972). One investigation was also noted where possible subclinical effects of nitrogen ingestion at concentrations below the drinking water standards were manifested by impaired motor reflexes.

4.1.1 The Modified Nitrogen Cycle in Soils Receiving Septic Tank Effluent

The fate of nitrogen in the environment is complex and is the result of a variety of physical, chemical, and biological mechanisms which are in turn greatly influenced by environmental conditions such as temperature, moisture, and atmospheric composition. With regard to septic effluent, considerable nitrogen is introduced into the soil below an absorption trench, and one can view the cycle of nitrogen in terms of sewage as the primary source (Figure 4.1). As discussed previously, about 75% of septic effluent-N is in the form of ammonium while the remainder is largely organic-N most of which is sorbed and transformed to ammonium in the crusted zone of the absorption field. Nitrogen then enters the soil from septic systems almost totally as ammonium.

TABLE 4.1 Cases of methemoglobinemia reported in the U.S. and Europe (National Academy of Sciences, 1972).

Number of Cases	Number of Fatalities	Reported Cause	Years
United States			
278	39	Well-water nitrate	1945-1950
40	0	Well-water nitrate	1952-1966
10	0	Well-water nitrate	1960-1969
12	1	Nitrate added to meat	1955
3	1	Nitrate added to fish	1959
Europe			
1,000	80	Well-water nitrate	1948-1964
15	1	Nitrate and nitrate in spinach	1959-1965

4.1.1.1 Plant uptake. The plant available forms of nitrogen in soils are ammonium (NH_4^+) and nitrate (NO_3^-); therefore, ammonium discharged as septic effluent may be immediately subject to removal by crops. Lance (1975) has reported that corn silage and reed canary grass supplied with excess nitrogen from secondary sewage effluent could remove 180 and 450 kg-N/ha/yr, respectively. The use of perennial grasses along with overseeded small grains was suggested as the best

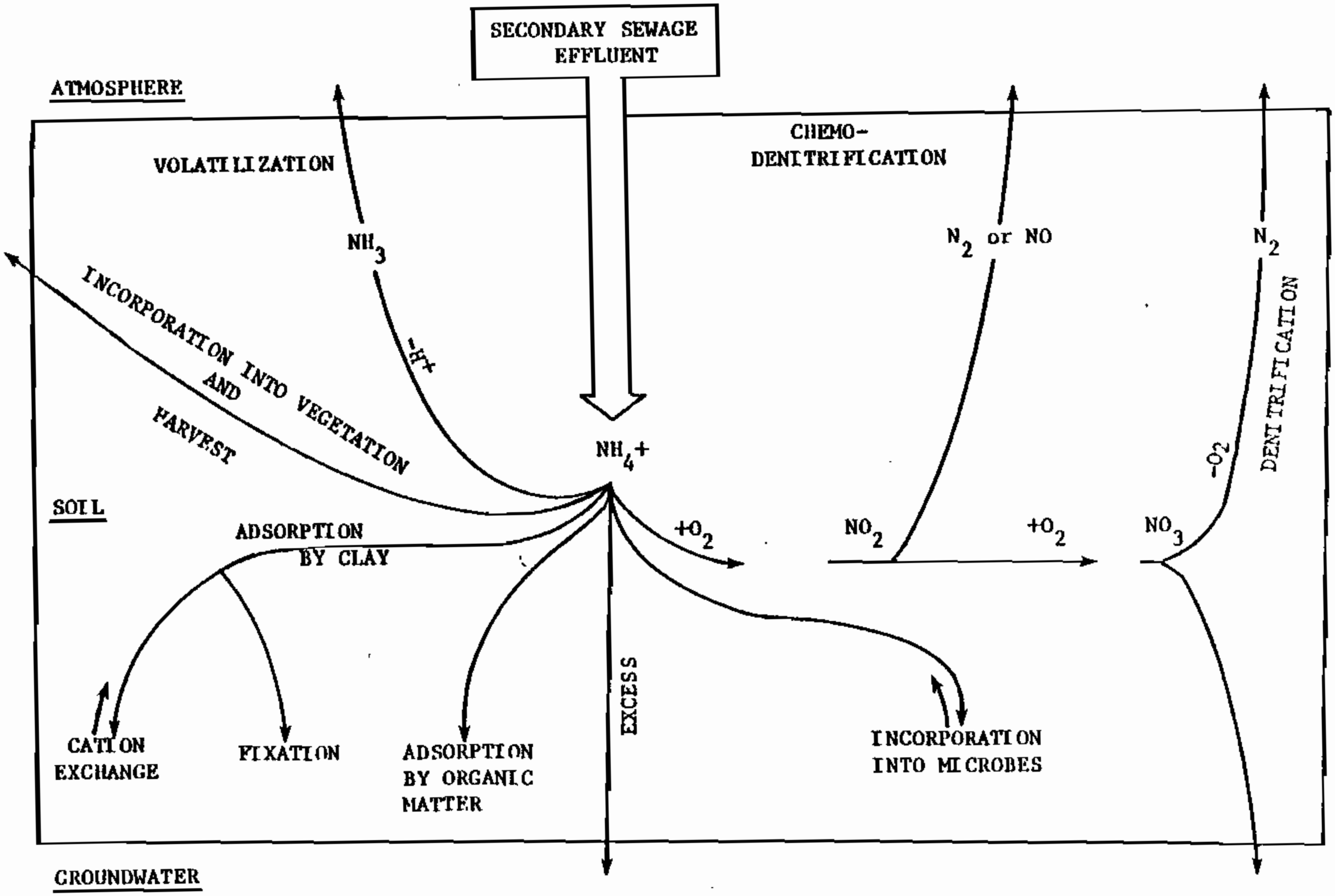

FIGURE 4.1 Nitrogen cycle below a septic field (Lance, 1972).

crop cover for southern states. However, data by Brown et al. (1977) show that for a sandy soil nitrogen additions from septic systems were 2170 kg N/ha/yr. Plant uptake would at best account for only 20% of applied nitrogen. For a septic system, particularly in New Jersey soils, this percentage will be considerably lower, because much of the nitrogen is introduced in the lower part of the root zone. System design for sands results in greater N applications per unit area, while the New Jersey climate limits the growing season. In fact, Brown and Thomas (1978) found that only 9% of applied N was removed by a year-round grass cover in a sand over a septic field in a humid sub-tropical climate.

Nitrogen removal by plant uptake also is dependent upon crop removal. If grass or crops are not harvested and removed from the site, much of the nitrogen will eventually be returned to the soil in various organic forms. Once an approximate equilibrium is established between plant uptake of nitrogen, plant decomposition and nitrogen mineralization from plant residues, the net effect of grass or crop cover on the nitrogen pool would be small. Therefore, it is essential that plants be harvested from above septic leach fields if model calculations of nitrogen leaching which include a plant uptake factor are to be reliable.

4.1.1.2 Nitrification. The oxidation of ammonium to nitrate, termed nitrification, is primarily a biological process involving the conversion of ammonium to nitrites by the bacteria Nitrosomonas and Nitrosococcus followed rapidly by further oxidation to nitrate by Nitrobacter. The nitrifying organisms are autotrophs requiring no organic carbon source so that the reactions may be written as

$$NH_4^+ \; 2O_2 \xrightarrow[\text{Nitrosococcus}]{\text{Nitrosomonas}} NO_2^- + H_2O + 2H+$$

$$NO_2 + 1/2O_2 \xrightarrow{\text{Nitrobacter}} NO_3^-.$$

Broadbent et al. (1976) observed that nitrate oxiders are sensitive to NH_4^+ and free ammonia, but adsorption of NH_4^+ in soils usually provides suitably low solution concentrations for oxidation to proceed.

Oxygen is the most common environmental restraint to nitrification, which requires 4.56 mg of oxygen per 1 mg of nitrogen oxidized (Lance, 1975). While the moisture regime of the soil can indirectly control the process by restricting soil oxygen, extremely dry conditions may reduce bacterial populations and limit nitrification. The optimum temperature range for nitrification is between 30 and 35°C, and at temperatures below 5°C the process rapidly decreases until cessation at 0°C (Alexander, 1977). However, there is evidence that the bacteria can adapt to climate (Mahendrappa et al., 1966), and Frederick (1956) measured significant nitrification at as low as 2°C.

An abundant base status of the soil is important to nitrification according to Brady (1974), who noted that even when the soil is below pH 5 nitrification can continue in organic soils when sufficient bases are present. In strongly acidic inorganic soils, long-term application of wastewater which is usually near neutrality can increase soil pH into an acceptable range, particularly where the soil has a low buffering capacity (Sopper, 1973).

Several points may be noted regarding nitrification in the Pinelands. Although aeration below septic lines should be adequate for NO_3^- formation, the initially low pH and base status of the native soils may discourage the oxidation of ammonium. Additionally, the cold temperature regime during much of the year can inhibit activity of the nitrifying bacteria, and movement of NH_4^+ into groundwater may be expected.

4.1.1.3 Fixation and adsorption of ammonium. Fixation of ammonium can occur by interactions with either 2:1 layer silicates or soil organic matter. As clay and organic matter content increases in soil, NH_4^+ fixation also increases. Burge and Broadbent (1961) studies organic soils and found a linear correlation of ammonium fixation and organic carbon associated with phenolic hydroxyl sites. Once fixed, nitrogen was relatively unavailable. Strongly acid soils inhibited fixation by organic matter. Table 4.2 summarizes a review by Nommik (1965) of the fixation by the inorganic soil fraction. The interaction involves NH_4^+ occupation of the interlayer sorption sites of 2:1 layer silicate minerals and the subsequest fixation by contraction of the mineral interlayer spaces.

TABLE 4.2 Ammonium fixation tendencies related to varying properties of mineral soils.

Parameter	Fixation Response	Comment
NH_4 concentration ↑*	↑	Equilibrium between fixed and exchangeable NH_4^+; percent fixation decreases
Temperature ↑	↑	Affects rate but not necessarily absolute fixation
Soil Moisture ↑	↓	Related to lattice contraction
Clay Content ↑	↑	Proportional to micaceous (2:1 layer) mineral content
K^+ concentration ↑	↓	Competition for interlayer sites
pH ↑	↑	Weak correlation; at low pH, hydroxyl-Al groups reduce the lattice collapsibility
PO_4^{3-} ↑	↑	Low solubility of PO_4^{3-} encourages fixation of the companion NH_4^+

* ↑ = Increasing; ↓ = Decreasing

Adsorption-desorption of NH_4^+ in the absence of nitrifying conditions is similar to cation exchange of other ions. The order of affinities for cations in soil solution is $Al^{3+} > Ca^{2+} > Mg^{2+} > K^+ > Na^+$ (Brady, 1974). The relative strength of adsorption is governed by the ionic strength and hydration radius of a cation species. As in the case of fixation, NH_4^+ competes about equally with K^+ for adsorption sites since both have equivalent charges and hydration radii. Preul and Schroepfer (1968) found that equilibrium between adsorbed and solution concentrations was approached rapidly, and Preul (1966) noted that desorption occurs easily. The low cation exchange capacity of Pinelands soils and their strong acidity indicative of codominance of Al^{3+} and H^+ on exchange sites suggest that the region's soils will adsorb little NH_4^+.

4.1.1.4 Immobilization of ammonium and nitrates. The immobilization of an inorganic nutrient is the result of microbial assimilation into organic compounds. Bacteria, actinomycetes, and fungi most readily assimilate ammonium salts above other nitrogen forms. However, many strains utilize nitrate as well (Alexander, 1977) and large quantities of nitrate can be immobilized in the absence of sufficient ammonium (Overrein and Broadbent, 1967). Immobilization and mineralization are simultaneous processes and the net accumulation of organic or inorganic nitrogen is a function of organic matter availability and of the C:N ratio of organic substrates (Alexander, 1977). The low carbon to nitrogen ratios of the suspended and dissolved sewage organic matter should be lower than the generally recognized critical ratio of 20:1 to 30:1, and a net mineralization of N rather than immobilization will minimize the impact of immobilization below absorption trenches.

4.1.1.5 Volatilization. Gaseous losses of nitrogen can occur from either the ammonium or the nitrate forms. In an alkaline soil, ammonium salts react to form free ammonia gas as follows:

$$NH_4^+ + H_2O + OH^- \rightarrow NH_3 + 2H_2O$$ (Tisdale & Nelson, 1975)

With increasing NH_4^+ concentrations and increasing gas exchange, the percent loss to volatilization increases (Overrein and Moe, 1967). Acidic conditions and limited gas exchange below Pinelands septic leach fields would allow negligible ammonia volatilization.

Denitrification is the primary mechanism for gaseous loss of nitrogen from soils, and the keys to such losses are the soil redox potential, organic matter content, and presence of denitrifying organisms. Broadbent and Clark (1965) identified the responsible organisms as facultatively anaerobic bacteria which can use nitrate as a hydrogen acceptor in the absence of oxygen. Although there is disagreement as to the role of nitrous oxide, N_2O, in the denitrification sequence, Cooper and Smith (1963) experimentally found the reaction sequence to be as follows:

$$NO_3^- \rightarrow NO_2^- \rightarrow N_2O \rightarrow N_2.$$

Anaerobic conditions are necessary for volatilization of nitrate, and the critical value of redox potential has been measured to be between 330 and 350 mV (Patrick and Mahatrapa, 1968). However, microsites in an aerated soil, such as within soil aggregates, can allow denitrification when oxygen diffusion is limited (Broadbent and Clark, 1965). Focht (1974) reviewed environmental effects and noted that denitrification ceases at pH 4 or at extremely low temperatures. Cho et al. (1979) have asserted that the critical temperature, where denitrification ceases is 2.7° C in any soil. Acidity, temperature, and adequate oxygen below a septic leach field in Pinelands soils will limit denitrification losses. Lance (1975) further noted that the organic matter required as an oxidizable substrate is low in sewage effluent and may be inadequate for extensive denitrification.

4.1.1.6 Leaching of nitrogen. The two predominant leachable forms of nitrogen in septic fields are NH_4^+ and NO_3^-. Excess nitrogen in these forms which is not attenuated by the previously discussed mechanisms will leach with soil water moving toward the groundwater. Which nitrogen form or forms are present and in what ratios is dependent on the environmental conditions as modified by the influence of the septic system. Walker et al. (1973) found that, due to unsaturated flow below absorption fields in loamy sands, nitrification commenced within 2 cm beneath the crusted trench bottom and was essentially complete within 6 cm. Numerous studies have noted that NO_3^- is indeed the predominant form of nitrogen leaching below properly functioning soil absorption systems (Brown et al., 1977; Hook and Kardos, 1978; Walker et al., 1973; Dudley and Stephenson, 1973; Bouma et al., 1972; Miller, 1973). Up to 93% of applied nitrogen has been found to leach as NO_3^- from a deep sand (Hook and Kardos, 1978). Peaks in nitrate leaching occurred in early spring to summer, probably related to nitrification of NH_4^+ accumulated over the winter.

Leaching of sewage effluent ammonium is usually associated with a shallow water table which does not allow sufficient aerated soil depth for complete NH_4^+ oxidation. Dudley and Stephenson (1973), observing numerous functional septic systems surrounding a recreational lake, found that NH_4^+ contamination of groundwater usually occurred where water table depth was less than 1.5 m. A Long Island, New York survey came to a similar conclusion when high ammonium concentrations were found in groundwater below field lines in sandy soils (New York Department of Health, 1969). Ammonium concentrations were 26 to 37 and 57 to 75 mg/1 where water tables were 1.5 and 0.6 m below absorption trenches, respectively (Table 4.3). Brown et al. (1977) and Reneau (1977) also reported significant amounts of NH_4^+ in shallow groundwater; however, both studies noted occasional waterlogging of the systems. For a continuously operated septic system, available adsorption sites are rapidly saturated so that any unattenuated NH_4^+ will freely move to the groundwater.

4.1.1.7 Nitrate contamination of groundwater. Nitrate reaching the water table subsequently follows the general groundwater flow pattern, except where excessive hydraulic loading causes "mounding" of the water table which can locally induce up-gradient nutrient movement (Dudley and Stephenson, 1973). Figure 4.2 diagrams a profile of effluent NO_3^- movement in the vicinity of an operative absorption trench. Hansel (1968) noted that horizontal flow is predominant with little vertical dispersion. Table 4.4 summarizes a number of investigations of contamination from domestic septic systems. Nitrate and ammonium are conserved once they have entered the groundwater, and dilution is the only mechanism for reducing contaminant concentrations.

TABLE 4.3 Field observations of ammonium leaching.

Soil Texture	Depth to Water Table (m)	Effluent N (mg/l)	Groundwater NH_4^+ (mg/l)	Horizontal Distance Moved (m)	Reference
Sandy loam	1.0	23	4.1	12.0	Reneau, 1977
Fine to medium sand	1.5	-	26-37	0	New York Dept. of Health, 1969
Medium to coarse sand	0.6	-	57-75	0	
Sand	5 to 6*	-	65 <5	0 5	Walker et al., 1973
Sand	1.8**	29.8	<1 to 8.0	0	Brown et al.,
Clay	1.8**	29.8	<1 to 2.1	0	1977

* Saturated profile.
** Profile periodically saturated.

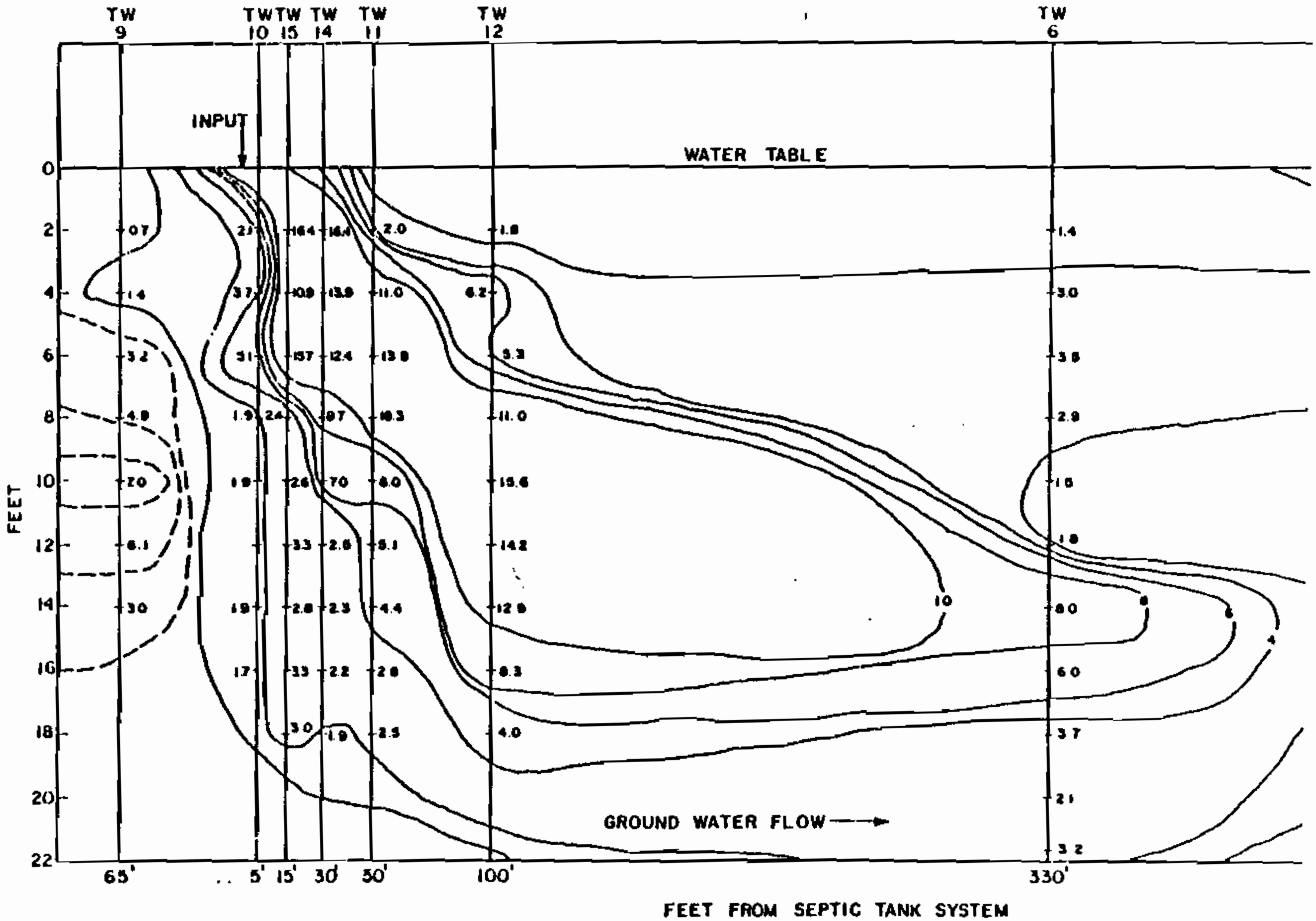

FIGURE 4.2 Nitrogen movement in the groundwater at a septic tank test site below a sandy soil, giving test wells (TW) sampled and the concentration (mg/l) of NO_3-N at two foot intervals below the water table (Ellis and Childs, 1973).

TABLE 4.4 Field studies of NO_3^- movement below septic tank-absorption trench systems.

Soil texture	System age (yr)	Depth to water table (m)	Septic tank effluent N (mg/l)	Groundwater NO_3-N (mg/l)	Horizontal distance moved (m)	Reference
Sand	5	3-4	27.1-33.8	15.5	6.1	Dudley & Stevenson, (1973)
Sand	8	4	27.1-33.8	2.4-20.3	9.1	"
Sand	1	3.4	27.1-33.8	2.4-11.4	0.9	"
Sand	9	17.1	27.1-33.8	13.79	0	"
Sandy loam	-	-	9.2	15.0	0	Hook & Kardos, (1978)
Sand	15	1.5-1.8	-	8.0	100	Ellis & Childs, (1973)
Sand	-	5-6	-	40 10	0 70	Walker et al. (1973)
Loam & sandy loam	2-15	4.6-7.6	-	22-136	0	Miller (1973)
Loamy sand	-	-	80	30 15	3.7 9.1	Bouma, et al. (1972)
Gravelly sand	-	-	-	10	30	Hansel (1968)

TABLE 4.5 Summary of the expected fate of nitrogen in Pinelands soils receiving septic tank effluent.

Process	Relative importance	Limiting parameters
Plant uptake of NH_4^+ and NO_3^-	<10% of total N	Climate (short growing season) Trench area (high N application rate)
Nitrification	varies; usually large	Temperature (restrictions in winter) pH (acidic; low exchangeable bases)
Fixation of NH_4^+	slight	2:1 clay content (low)
Adsorption of NH_4^+	<1% of total N	Cation exchange capacity (low)
Immobilization of NH_4^+ and NO_3^-	slight	C:N ratio (low; net mineralization)
Volatilization of NH_4^+	negligible	pH (acidic) Gas exchange (depth in soil)
Denitrification of NO_3^-	low	Aeration (abundant O_2) Organic matter content (low) Temperature (restrictive in winter) pH (acidic)
Leaching of NH_4^+ and NO_3^-	>90% of total N	Attenuation by the above processes
Groundwater movement	free movement	Adsorption (low) Volatilization (low)

4.1.1.8 Summary of the fate of effluent nitrogen. The probable fate of nitrogen in sewage effluent applied to Pinelands soils is largely a function of the climatic regime, regional soils, and the characteristics of properly functioning septic tank-soil absorption systems. Table 4.5 describes what might be expected. In general, the potential appears to be great for significant nitrate and, at times, ammonium pollution in Pinelands Groundwater.

4.1.2 Leaching of Nitrogen Under Lawns

Losses of nitrogen from lawns are primarily dependent on the type and amount of fertilizer applied and the amount of water leaching through the soil. Application technique and frequency, vegetative cover, plant harvest, and soil properties such as pH are of secondary importance. In general, the majority of investigators have centered on agricultural losses of N and relatively little has been done regarding losses from home lawn areas. Many of the reactions are similar so that extrapolations can be made. Some studies have been done concerning N and other chemical losses from golf greens and athletic fields of sandy composition.

4.1.2.1 Source of N. The potential leachability of several fertilizer N sources was studied by Bredakis and Steckel (1963) using a fine sandy loam soil. Their results (Table 4.6) show potential losses of N from inorganic sources to be as high as 85-90% while that from organic sources may be as much as 55-70%. It is also apparent that the inorganic sources are much more quickly leached from soils (Figure 4.3). The research of Rieke and Ellis (1974) supports the conclusions

TABLE 4.6 Reports in the literature on nitrate nitrogen losses from various nitrogen sources.

N Source	Bredakis & Steckel (1963)		Snyder et al. (1980)		Brown et al. (1975)*	
	N Rate (kg/ha)	% lost	N Rate (kg/ha)	% lost	N Rate (kg/ha)	% lost
$(NH_4)_2SO_4$	97	88.5-90.8			146	8.94
NH_4NO_3			50	35.55	163	20.6
Urea	97	84.6-89.2			244	0.29
Milorganite	97	63.1-76.9			146	5.02
Nitroform	97	54.6-69.2				
IBDU**					146	0.74

* Brown's data is lower because a perched water table and the presence of free organic matter at a depth of 12 inches may have resulted in significant denitrification.

** Isobutylidene diurea (31% N).

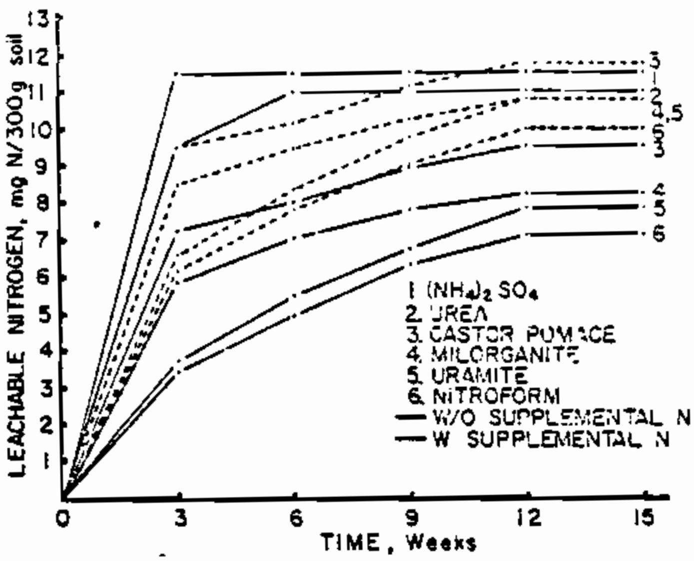

FIGURE 4.3 Rates of formation of leachable nitrogen from turfgrass fertilizers in soil at pH 5.5. (Bredakis and Steckel, 1963.)

of Bredakis and Steckel (1963). Rieke and Ellis suggested that applications of inorganic N be limited to less than 50 kg N/ha and that smaller applications be made more frequently throughout the season. Further studies by Brown et al. (1975) on NO_3^- losses from golf greens, showed that up to 20% of the N applied as inorganic NH_4NO_3 can be lost as NO_3^- in the leachate while only 0.3% of the N applied as urea was leached as NO_3^-. Concentrations as high as 314 mg/l NO_3^- were measured in the leachate from sandy golf greens treated with NH_4NO_3 while NO_3^- concentrations from greens treated with organic N sources remained under 20 mg/l. Rieke and and Ellis observed similar high concentrations (up to 111.4 mg/l NO_3^-) in plots treated with 390 kg/ha NH_4NO_3. Studies by Harding et al. (1963) of an orange orchard growing in Ramona stony sandy loam soil found greater than 200 mg/l NO_3^- in air dry soil at the 0-12 and 12-24" depths for three months following an application of 605 kg N/ha as $Ca(NO_3)_2$ in spring. This information is supplemented by Sunkel (1979) who measured 30-55 kg N/ha/yr leached from pastures on sandy soils. These losses amount to about 1 lb per 1000 ft per year which is approximately half of a typical N application. All studies indicated a rapid loss of N as NO_3^- following application of inorganic N fertilizers to sandy soils. The available literature also concluded that use of frequent small applications of inorganic sources will help minimize NO_3-N losses.

4.1.2.2 <u>Application rate and technique</u>. Available data in the literature on the effects of application rate on nitrate losses in sandy soils is very limited. From the study of Harding et al. (1963), it can be seen that NO_3^- concentrations in the soil at various depths

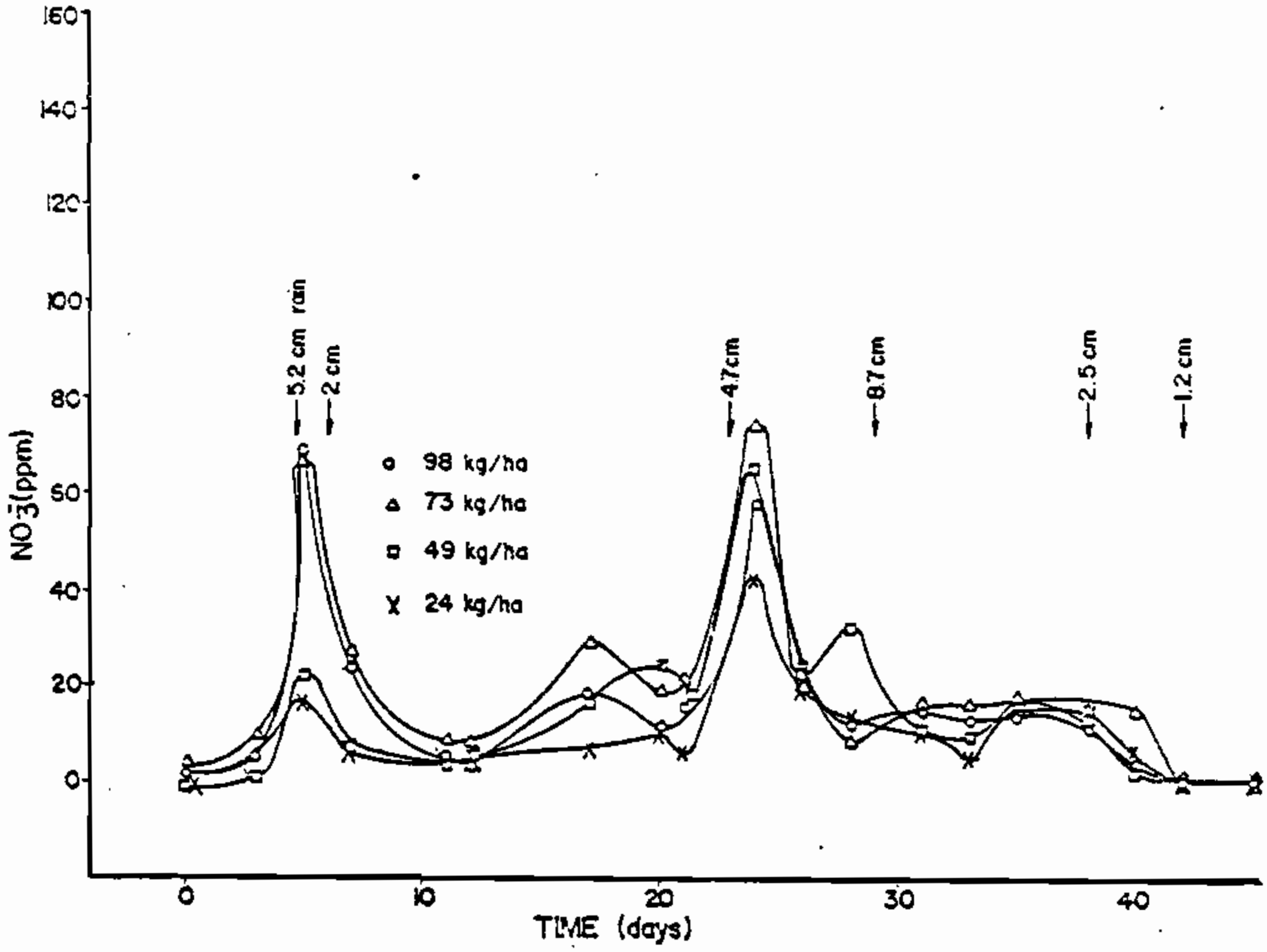

FIGURE 4.4 Concentrations of NO_3^- in the leachate from plots receiving applications of different amounts of N as $(NH_4)_2SO_4$ on August 2, 1974.

below the surface are strongly influenced by application rate. A total nitrogen application of 605 kg N/ha in three equal doses of $Ca(NO_3)_2$ applied in mid-February, mid-May, and mid-August resulted in nitrate concentrations in the 24-36 inch horizon of 35-85 mg/l compared to 35-200 mg/l for the same horizon in plots given the entire application in mid-February. Results from organic sources were similar but much lower in magnitude. Data from the studies of Brown et al. (1975) showed similar results (Figure 4.4). The effects of application rates were most pronounced immediately following the significant rainfall events, and in general the nitrate concentrations in the leachate were proportional to the application rates. Nitrogen losses are summarized in Table 4.7. From the table, it is evident that as the application rate increases, the loss of NO_3-N similarly increases on a kg/ha basis. Thus, to keep actual losses of N to a minimum, smaller and more frequent applications of N are required.

Recent research by Snyder et al. (1980) showed that daily fertilization via irrigation water is more efficient and results in lower overall NO_3^- concentrations in the leachate of sands.

TABLE 4.7 Leachate losses of NO_3-N from golf greens.

N application rate (kg/ha)	NO_3-N lost %	NO_3-N lost kg/ha
98	15.5	15.2
73	22.0	16.1
49	24.9	12.7
24	37.8	9.1

Caution must be exercised, however, since concentrations of NO_3^- are only a portion of the overall loss. In order to properly account for lost N, one must also know the volume of water leaching through the soil. Leachate volume is a function of the water applied to the surface as rainfall and irrigation. Therefore one must carefully regulate irrigation to prevent high N losses.

Mitchell et al. (1978) studied subsurface application of N to sandy soils in a golf green and found that subsurface fertilized plots lost 57% more N than plots receiving an equivalent surface application. The irrigation lines were at a depth of 25 cm below the soil surface which was probably a major cause for the increased NO_3-N leaching. Plant roots are generally concentrated heaviest in the top 15 cm. By applying N at a depth of 25 cm, there is less opportunity for plant uptake, soil adsorption, or transformation.

4.1.2.3 Effect of irrigation on nitrate losses. It appears from the available literature that the irrigation rate has a profound impact upon NO_3^- losses in sandy soils (Brown et al., 1977; Duble et al., 1978; Snyder et al., 1980). Brown et al. (1977) demonstrated that by lowering the irrigation rates it is possible to lower the NO_3^- concentrations in the leachate by 40-50% (Figure 4.5). In addition, the peak from the low irrigation rate treatment shown in Figure 4.4 is wider than the others, indicating slower movement and longer contact time for root uptake and other soil N transformations. If organic N sources are used as discussed earlier, the influence of irrigation would be less although the data of Rieke and Ellis (1974) indicated that it would still be significant.

Snyder et al. (1980) confirmed the above finding and provided insight as to the speed of NO_3^- movement through the sands. In their investigation, the authors reported that 12 cm of excess irrigation leached 34 to 54% of the applied N depending on the application technique. They found the downward moving N pulse to be at the 60 cm depth after six days due to an excess application of only

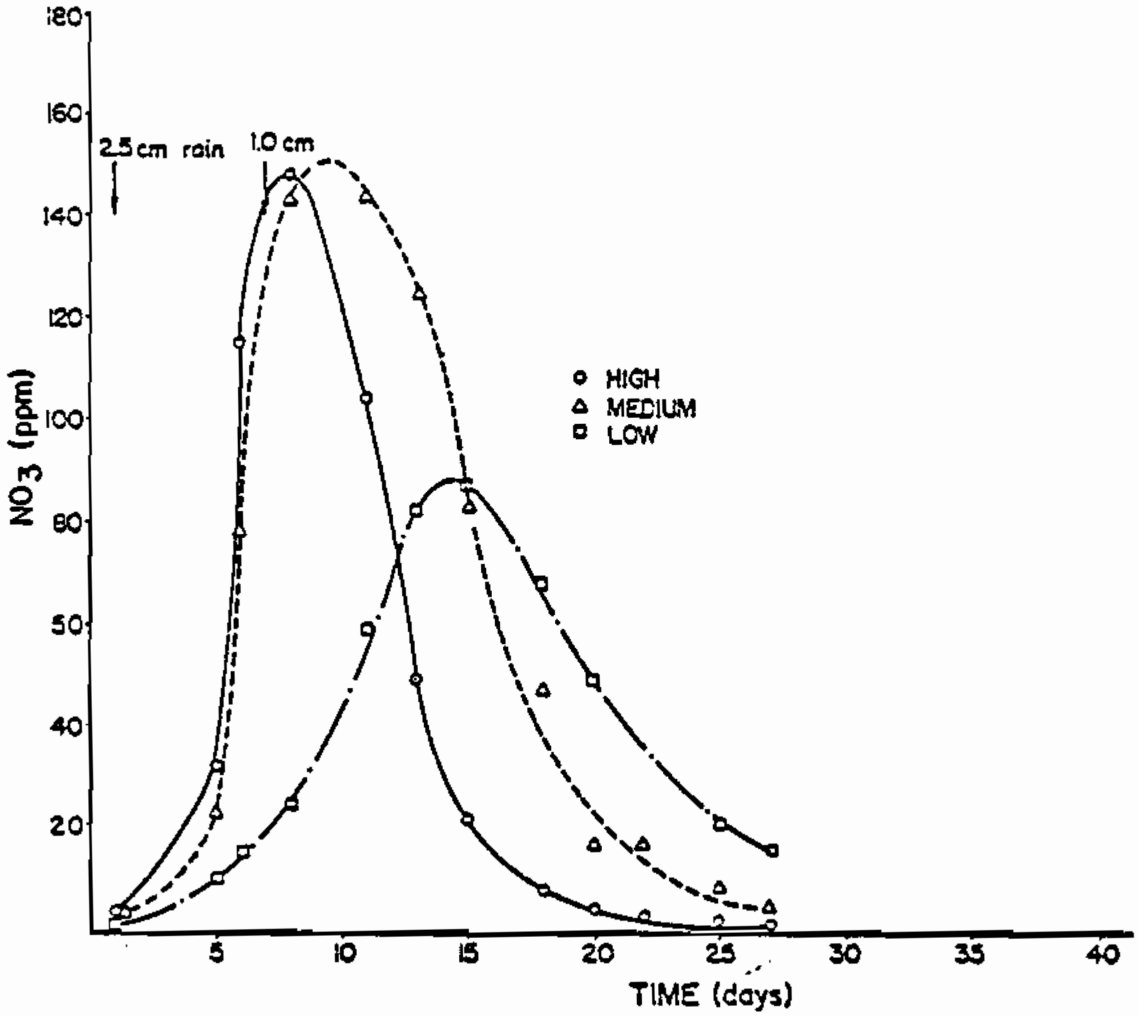

FIGURE 4.5 Effect of irrigation rate on leachate loss of NO_3-N from an application of 146 kg N/ha on July 26, 1973.

5.24 cm of water. Snyder et al. (1980) also calculated that the sand they studied only required 0.8 cm of water in excess of evapotranspiration to move the N pulse below the bulk of the root system. Thus one may conclude that to minimize NO_3^- losses, irrigation must be kept below that which can be lost through evapotranspiration. Losses from rainfall events cannot be prevented, but leaching can be minimized by decreasing irrigation before a rain and stopping irrigation after a rain until the soil has dried sufficiently to retain another water appplication.

4.1.2.4 Other parameters of interest. The literature presents a very limited insight into other parameters which may influence NO_3^- losses. In general, their impacts are rather small and therefore are only briefly considered.

From the data of Bredakis and Steckel (1963) one can see the effects of soil pH on NO_3^- losses. They found that as the soil pH decreased, the potentially leachable N increased, particularly in the first three weeks after application of organic sources. Increases ranged from 2.3% for $(NH_4)_2SO_4$ to 13.8% for Milorganite. Presumably the acid soil conditions (pH 5.5) aided the more rapid breakdown of the organic fertilizers compared to rates for neutral soil (pH 6.7). Urea was the only exception with an increase of only 1.5% due to acidity.

A glimpse of the influence of the plant root system can be gotten from the work of Snyder et al. (1980). They measured leaching losses in excess of 50% of the applied N when the root systems were severely damaged by nematodes and the applied irrigation exceeded evapotranspiration by 50%. It is difficult to quantitate this loss as compared to the loss a healthy root system might suffer, but in a healthy root system leaching losses may range from 5-10% of the applied N.

Allen et al. (1978), working with soil mixtures of 50% builder's sand and 50% Norfolk silt loam, demonstrated that losses of applied N from areas vegetated with tall fescue ranged from 0 to 11% while losses from fallow areas ranged from 29-72%. According to this information, it appears that vegetation may consume considerable quantitities of N. The idea of utilizing grasses as a means of N removal has been suggested in the past by Brown and Thomas (1978).

It is anticipated that although a ground cover and thatch buildup would be less effective in a sandy soil, dense sod cover, if harvested, would assist in removal of N and increase the evapotranspiration to help lower the NO_3^- leaching.

Thus from the available literature it appears that up to half of applied N may be leached as NO_3^- from home lawns on sandy slightly acid soils. Parameters which limit these losses are much lower irrigation, much smaller and more frequent applications and the use of slow release organic N sources.

4.1.3 Agricultural Fertilization with Nitrogen

Soluble agricultural fertilizers that are not strongly adsorbed by the soil may be carried below the root zone of plants by percolating water. Water reaching the groundwater below a fertilized field may contain unacceptable concentrations of NO_3-N as demonstrated by Stout and Burau (1967), Nightingale (1972) and Stewart and Viets (1968). Several factors including application rate and timing, crop uptake, soil permeability and temperature, and the form of fertilizer used will influence the amount of nitrate moving to the groundwater. Douglas (1976) reported that nitrate nitrogen concentrations below fertilized fields of Freehold loamy sand at the Adelephia field station were less than 10 mg/l, and that proper fertilization did not constitute a groundwater pollution problem. Furthermore, agricultural activities in the Pinelands are very limited. The major crops produced and fertilizer application rates are given in Table 4.8.

TABLE 4.8 Nitrogen fertilizer recommendations for agricultural crops given in lb/acre/year.

	Total	Broadcast and plowdown	Band place at planting	Sidedress-topdress
Corn, grain or silage	120	40-50	20-40	40-60
Small grains	0-90	0-30		0-60
Soybeans	0-20	-		-
Alfalfa	0			
Tall grass for hay	50-150			split application
Rye for grazing	100	50		5
Cranberries	40 as NH_4SO_4			20 spring 20 fall
Blueberries	80			

The 1978 agricultural statistics indicated that 19,700 acres of corn were grown in Burlington County, while only 1,200 and 300 were listed for Ocean and Atlantic counties. These are the three major counties within the Pinelands boundary, but it is likely that some of the acreage of corn reported for Burlington and Ocean counties may fall outside the boundary. Thus the acreage involved in corn production is very small.

With the exceptions of blueberries and cranberries, few crops which require extensive N fertilization are grown within the Pinelands. Since Blueberries and cranberries are grown on soils which have slowly permeable subsoil, downward migration of nitrogen should be minimal. Nitrogen losses in surface drainage water could, however, contribute to the nitrogen content of receiving streams. Data from Florida indicate that solutions from drained peat bogs contained nitrate nitrogen concentrations well in excess of 10 mg/l (Neller, 1944). No data is available, however, from the New Jersey Pinelands area concerning concentrations in the drainage water. If intensive truck crops are grown in the Pinelands, the accompanying high rates of N fertilization may lead to significant increases in the nitrogen concentrations in groundwater. Potential fertilizer losses may be comparable to those previously discussed for leaching of N under lawns.

4.1.4 Secondary Sewage Effluent Irrigation

One alternative to the individual on-site septic leachfield is the centralized collection and treatment of the raw sewage. Secondary treated sewage effluent often contains in excess of 20 mg/l nitrogen and cannot be released directly into streams without causing pollution. Tertiary treatment to remove nitrogen is very expensive and often does not reduce concentrations to less than 5 mg/l nitrogen. Recently, disposal by land irrigation has been used by many communities. When large volumes of water are applied, little or no nitrogen attenuation is achieved before the water enters the water table. When lower irrigation rates are used, substantial nitrogen removal can be achieved.

Hook and Kardos (1978) reported a hardwood forest in Pennsylvania on a well-drained sandy loam soil which was irrigated with approximately eight inches of secondary municipal sewage effluent per month. In the last seven of the nine years of application, the amount of nitrogen which leached was equivalent to the amount of nitrogen applied. Nitrate nitrogen concentrations in the leachate were usually greater than 15 mg/l. When the application rate was reduced to approximately 4 inches/month, the nitrate nitrogen concentration at 48 inches depth was still greater than 10 mg/l.

An old field with a sparse stand of white spruce on a clay loam was irrigated from April to November each year with approximately 8 inches of effluent/month. At 48 inches depth, the nitrate nitrogen concentrations rarely exceeded 10 mg/l (Hook and Kardos, 1978).

Mean annual nitrate nitrogen concentration in soil water in a red pine plantation remained below 5 mg/l when irrigated at 1 inch of effluent per week. When the application rate was increased to 2 inches/week, the nitrate nitrogen concentration of the soil water increased to 20 mg/l or more. In a mature forest on the same silt loam soil, nitrate nitrogen concentration in soil water remained below 10 mg/l at the application rates of 1 inch/week and 4 inches/week (Kardos and Sopper, 1973).

Thus it is evident that if application rates are kept low enough, irrigation may be an evironmentally acceptable means of sewage effluent disposal. At high rates however, excessive nitrate nitrogen pollution will result.

4.2 Phosphorus

4.2.1 Septic Fields

Contamination of ground water supplies by phosphorus (P) moving from septic drainage fields could be a severe problem in the soils of the Pinelands Region if the sandy soils have low P adsorption capacities. The review of the literature pertaining to reduction of phosphate concentrations by soils systems indicates that one of the primary factors in P removal is the tendency of phosphorus to sorb on soil particles. Soils vary greatly in their ability to adsorb soluble phosphate ions. The extent to which septic effluent is renovated by filtration through soil is influenced by the soil pH, clay content, the chemical and mineralogical composition of the clay, and cation exchange. The P removal process is further modified by the residence time in soil as determined by physical limitations such as restricted permeability and adsorption/desorption.

Phosphorus in septic tank effluent originates from two main sources, detergents and human waste. The relative contribution of detergent P will vary with the amount of detergent used and with its P content, and this should be reflected in forms of P present in soil solution. Anaerobic digestion occurring in the septic tank converts most of the phosphorus, both organic and condensed phosphate (pyro-phosphate, for instance), to soluble orthophosphate. Magdoff et al. (1974) and Otis et al. (1975) found more than 85% of the total phosphorus in most septic tank effluents studied to be in the orthophosphate form.

4.2.1.1 <u>Sorption mechanisms</u>. At low concentrations (5 mg/l) in the equilibrium solution, the phosphate ion becomes chemisorbed on the surfaces of Fe and Al minerals in strongly acid to neutral systems. The pH of septic tank effluent is nearly or just above neutral, whereas the pH of soils mapped in the Pinelands range between 3.6 and 7.8 though most fall below pH 5. Therefore Fe and Al would participate in P immobilization reactions in absorption field systems to an extent governed by the limits of Fe and Al activities in dilute solution. A rise in the P concentrations in soil solution above the solubility product of P complexes results in the formation of phosphate precipitates such as strengite ($FePO_4.2H_2O$) and variscite ($AlPO_4.2H_2O$) which form in acid soils. Hydroxyapatite ($Ca_{10}(PO_4)_6(OH)_2$) is the stable calcium phosphate precipitate in the pH range encountered in most absorption fields (Figure 4.6).

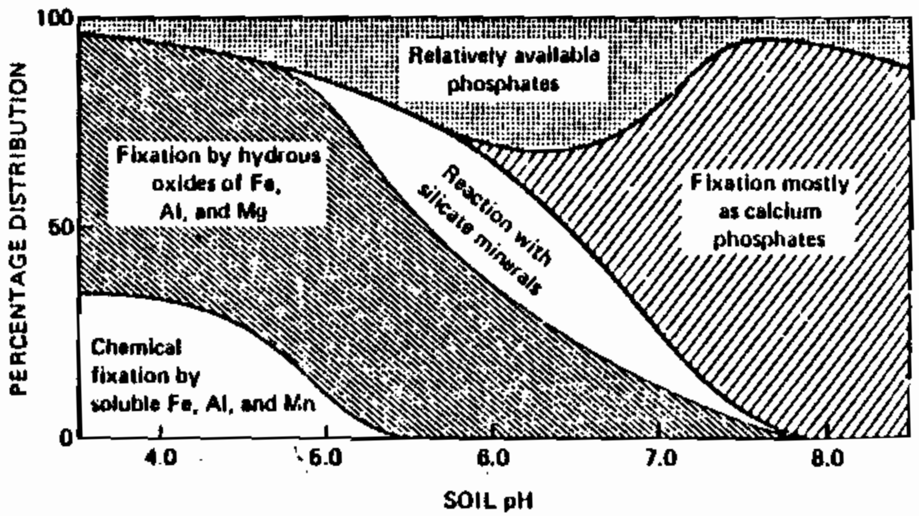

FIGURE 4.6 Forms of phosphorus typically present in soil as a function of pH (after Brady, 1974).

Anaerobic conditions which sometimes exist under septic drainage fields may result in an increase in inorganic P dissolved in soil solution at the same redox potential range over which ferric - P complexes are reduced. The released phosphorus then binds to Fe and Al hydrous oxides and a new solution equilibrium is reached.

The rate at which phosphorus is sorbed from solution onto the surfaces of soil has been shown to consist of a two-fold process. A rapid initial reaction is followed by a much slower reaction that may continue for several weeks (Chen et al., 1973). Soil material directly under the drainage field will be exposed to relatively high concentrations of P participating in a slow reaction which is weakly influenced by diffusive and adsorption/desorption mechanisms.

4.2.1.2 P adsorption capacity. Researchers attempting to describe the kinetics of P movement in soils have utilized Langmuir and Freundlich models to predict a relatively sharp boundary between a zone of maximum P adsorption near the drainage tiles and underlying soil. Field investigations of effluent movement show that in soil solutions at equal distances beside the absorption trench, phosphorus reached similar concentrations after the system reached equilibrium or near equilibrium. After a layer of soil surrounding the trench becomes saturated with phosphorus, the concentration of phosphorus in that zone (15 cm) reaches the concentration in waste water in the trench. Sawhney and Starr (1977) concluded that concentrations of only 0.5 mg/l at 60 cm depth in this system under investigation show that a soil with a deep water table below the drainfield should effectively renovate waste water effluent for a number of years, and should permit only minimum additions of phosphorus to the groundwater. Sawhney and Starr (1977) also noted a regeneration of the adsorption capacity of a soil after a dose rest cycle, but did not attempt to describe a mechanism.

The depth of penetration of P saturated layer can be calculated if the loading rate and P immobilization capacity of the soil are known. In order to predict the concentration profiles of phosphorus as functions of depth and time, some experimental data are needed. Hydraulic conductivity and capillary potential data should be known over the whole range of saturation values. Also needed are the P equilibrium relationship between the solid phase and the liquid phase, and the value of the overall mass transfer coefficient.

When the liquid flow can be considered constant, however, the hydraulic conductivity and the capillary potential data as a function of saturation are not required. The equilibrium relationship of phosphorus in the liquid phase with phosphorus in the stationary phase is simply found from adsorption capacity data, once the adsorption rate constants and the adsorption maxima are determined from Langmuir isotherm data. The Langmuir adsorption maximum would have to be considered a minimum value for potential P immobilization since the amounts of phosphorus retained by the soils may be reinforced by contributions via long term precipitation mechanisms undetected by Langmuir isotherm data.

Some illustrative examples of measured P adsorption capacities are 100 μg P/g soil for a sandy soil below a septic field (Walker et al., 1973) and 121 μg P/g soil for a sandy soil in laboratory columns (Magdoff et al., 1974).

4.2.1.3 Phosphorus retention in sandy soils. Based on previous studies of the chemistry of phosphorus in soil solution, it is expected that the potential of a soil to remove phosphate from septic tank effluent is controlled by the mineralogy of the soils rather than by the soil particle size distribution. The two are, of course, very much related when Fe and Al hydrous oxides occur in the clay-sized fraction of soils as coatings on the surfaces of soil particles. Trela (1980) reviewed the literature pertaining to phosphorus renovation mechanisms in soil and concluded that the sands and loamy sands of the Pinelands should have a very low P fixation capacity and thereby are unsuitable for septic tank effluent disposal. The volume of soil available for waste renovation is probably insufficient when loaded at high rates to a depth of 70 and 90 cm, unless this zone is underlaid with medium to fine textured materials acting as a buffer at least 35 cm thick.

Horizontal displacement of effluent caused by a restrictive layer may be advantageous. Reneau (1979) observed that as effluent moved through a fine loamy soil, phosphorus concentrations decreased logarithmically with distance. As much as 87% of the observed reduction in phosphorus could be attributed to distance. Thus, horizontal flow increases the soil volume available for phosphorus attenuation. Additionally, the restrictive layer can effectively prevent the leaching of P into potable aquifers or into the regional groundwater flow pattern.

Data from samples collected from soils surrounding septic tank systems at Houghton Lake indicated that attenuation of phosphorus in sandy aquifers is primarily by dilution (Ellis and Childs, 1973). P-bearing septic effluent moves in a well defined plume below the water table as illustrated in Figure 4.7. Numerous investigations of phosphorus movement in both perched and unrestricted groundwater illustrated that phosphorus moves rather freely once it has entered the saturated zone (Table 4.9). In each case, dilution appeared to account for most of the reduction in groundwater P concentrations with distance.

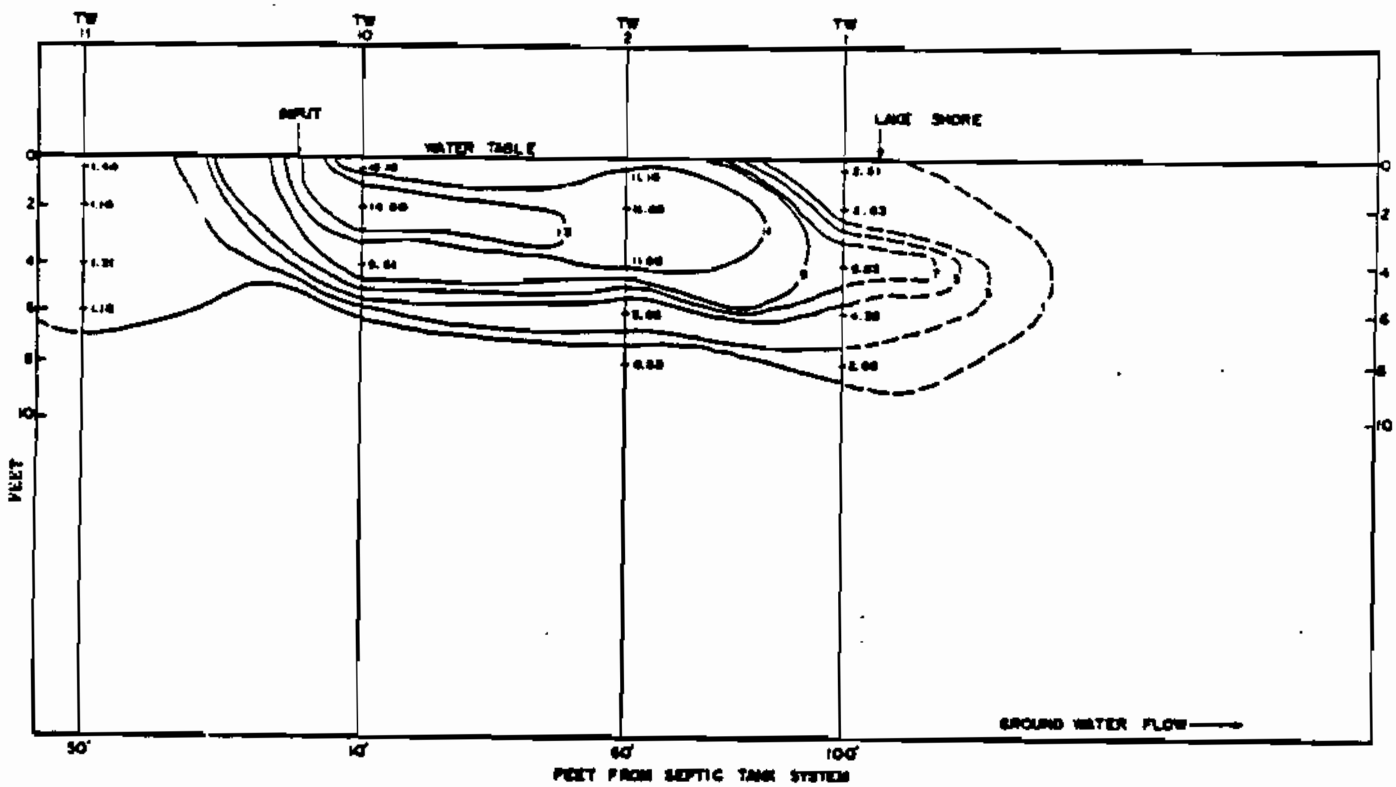

FIGURE 4.7 Phosphorus movement in the groundwater at a septic tank test site below Nester loam, giving the test wells (TW) sampled and the concentrations (mg/l) of phosphorus at two foot intervals below the water table. (Ellis and Childs, 1973).

In summary, an examination of data representative of values of phosphorus retained in selected soils reveals that vertical movement is greatest through sandy soil. Where vertical movement is impeded, or where the P adsorption capacity of the soil is exceeded with subsequent P movement into the groundwater, phosphorus can move horizontally for extended distances since saturation reduces the effectiveness of soil-effluent interactions. Furthermore, the data of Ellis and Childs (1973) support the generalization that P retention improves with P adsorption capacity and equilibration time in soil, which can be indicated by soil hydrologic groups (Dudley and Stephenson, 1973; Humphreys and Pritchett, 1971).

4.2.2 Sewage Effluent Irrigation

Secondary treated effluent contains significant amounts of phosphorus as phosphates, and applied effluent phosphorus will be taken up by plants or adsorbed by the soil. When adsorption sites near the surface are saturated, the excess phosphorus will leach to successively deeper layers although vegetation can remove significant amounts of phosphorus. Thus, when irrigation rates are sufficiently low, the combined uptake and adsorption should minimize the downward migration in soils with relatively low adsorption capacities. Throughout 11 years of effluent application in Pennsylvania, no sites showed leaching of more than 3% of total phosphorus applied, and where waste water was applied to existing crop systems at rates of 1 to 3 inches/week, phosphorus removal was usually complete (Kardos and Hook, 1976).

In soils with restricted drainage or in heavily fertilized soils under irrigation, concentrations of phosphorus in effluent from tile drains ranged from .05 to .23 mg/l (Sawhney and Starr, 1977). Concentrations as high as 1 mg/l were observed in effluent from the drains installed in shallow tilled sand under a heavily fertilized citrus crop (Calvert, 1975). Moderate rates of waste water application (5 cm/wk) for several years resulted in very little movement below the surface 15 cm of soil (Hill, 1972; Hook, et al., 1973; Kardos, 1976). However, high application rates of about 35 cm per week (Adriano et al., 1975) and 230 cm per week of secondary treated effluent to sandy soils for several years produced 1.8 and 5 mg/l phosphorus in subsurface water, respectively (Bouwer, 1973).

4.3 pH, Nutrients and Metals

Since it is known that the buffering capacities of the acid sandy Pinelands soils are low it is assumed that the subsoil and ground water pH can vary with the pH of the materials leaching from septic systems. Little is known about the strength of effluent components or their chemistry which may influence soil pH. Presently, field data are lacking to verify the above assumptions regarding the influence of pH on nutrient and metal solubilities in soil directly below septic drainage fields. This remains an area for future investigation.

TABLE 4.9 Phosphorus movement below septic tank-soil absorption trench systems.

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	-	.099	1.5-1.8	100	Ellis & Childs (1973)
Loamy sand	8	11.5	11.6	.9-1.2	9	"
Sand	5	27.1-33.8	.04-.05	3-4	6.1	Dudley & Stevenson (1973)
Sand	8	-	.65	4	-	"
Sand	9	-	up to 5.5	17.1	12.2	"
Sand	-	13.16	.05-.28	7.5	18	"
Sand	-	5.5	0.5	-	.7	Sawhney & Starr (1977)
Sandy loam	-	10.8	.01-.55	-	10.4	Reneau (1977)

5.0. BACTERIA VIRUS AND PATHOGENS

An important consideration for the soil disposal of septic tank effluent is the fate of microorganisms in the soil and groundwater environment. The principle biological contaminants found in domestic wastewater can be divided into four groups: bacteria, viruses, protozoa, and helminth parasites. The most common diseases associated with water-borne pathogens include typhoid fever, bacillary dysentary, infectious hepatitus, and amoebic dysentary. Survival, and the adsorption and movement of pathogens in soil are the major factors which influence the accumulation and passage of pathogens into groundwater. Table 5.1 shows the estimated number of pathogens added to each soil group from septic effluent.

TABLE 5.1 Estimated rates of pathogen application from septic tank effluent.

Soil hydro. group	Effluent ($1/m^2/yr \times 10^{-4}$)	Pathogens (org./m^2/day $\times 10^{-3}$)			
		Total coliform	Fecal coliform	Fecal streptococcus	Virus
A	3.7	16.9	4.3	0.6	7.7
B	1.5	6.9	1.7	0.2	3.1
C	0.92	4.2	1.1	0.15	1.9

5.1 Survival

5.1.1 Survival of Microorganisms

The survival of microorganisms in soil can range from a few days to several years (Miller and Wolf, 1975). Estimated survival times for bacteria and viruses are given in Table 5.2 and Figure 5.1. Persistence

TABLE 5.2 Estimated density of pathogens in raw sewage or septic tank effluent.

Organism (CFU/liter)	Foster & Englebrecht 1973	McCoy & Ziebell 1975	Viraraghaven & Warnock 1976	Brown et. al. 1977	Bouma et. al. 1972	Mean	Estimated Survival Ti (Miller and Wolf, 197
Total coliform (x 10^3)	-	-	230	266.1	4.8	167	100-150 days
Fecal coliform (x 10^3)	8	42	16	110.8	.6	42	-
Fecal streptococcus (x 10^3)	-	.38	11	-	-	6	-
Virus (x 10^3)	151.4	-	-	.08	-	76	-
Pseudomonas Aeruginosa	-	880	3	-	-	-	-
Salmonella sp (x 10^3)	75.7	-	-	-	-	-	1-85 days
Entamoeba histolytica	60	-	-	-	-	-	40-170 days
Helminth (x 10^2)	9.5	-	-	-	-	-	2.5-7 years

in soil is dependent on moisture, temperature, ionic strength, pH, dissolved gas concentrations and nutrient concentrations. Population changes occur in response to fluctuations in these factors and these fluctuations may occur due to climatic changes, the actions of microorganisms, or the actions of humans (Duboise et al., 1979). For all water borne pathogens the single most important factor influencing survival is soil moisture (McFeters et al., 1974). Fecal coliform survival in soil has been reported to range from 3.3 days (Van Donsel et al., 1967) to 150 days (Miller and Wolf 1975). Gerba et al. (1975) found survival of E. coli and Streptococcus faecalis to be less than one week in a peat soil with a pH of less than 4.5. In a study on the effects of seasonal variations on survival of indicator bacteria in soil, Van Donsel et al. (1967) found survival of bacteria in the summer in exposed areas to be twice that of shaded areas. This study also noted an aftergrowth of coliforms after temperature and rainfall variations. Taylor and Burroughs (1971) also observed increased levels of bacteria due to rainfall within 24 hours of sludge application.

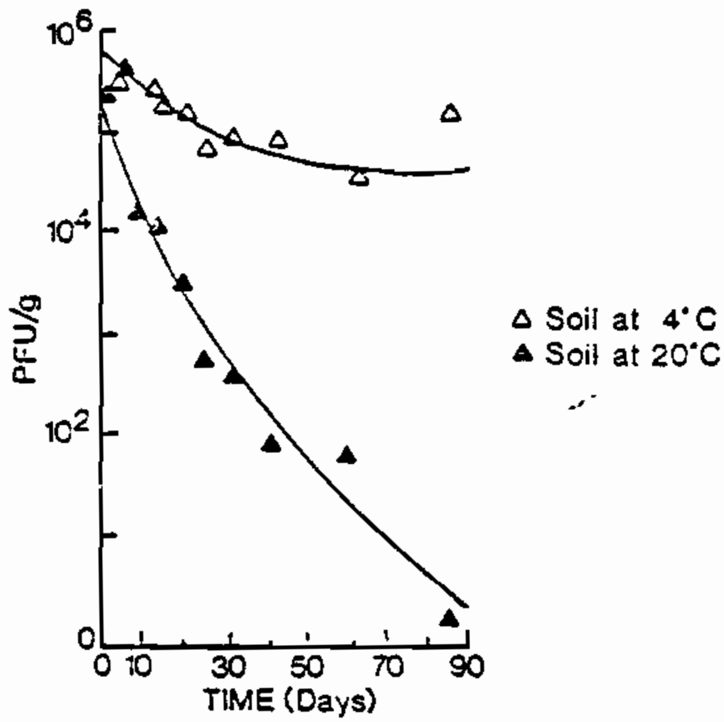

FIGURE 5.1 Survival of poliovirus in sandy forest soil (Duboise et al., 1976).

5.1.2 Survival of Virus

The length of survival of viruses in soil is greater than for bacteria. Yeager and O'Brien (1979) recovered fully infectious virions from soil at 4°C for up to 180 days. Leffler and Kott (1974) found that polio virus disappeared 63-91 days after application to sand saturated with distilled water. Viruses in soils are often associated to solids, and certain viruses associated with solids remain infective for extended periods of time (Moore et al., 1974). In addition, viruses are protected by septic tank liquor against inactivation under saturated soil conditions, although in dry soils this protection is lost (Yeager and O'Brien, 1979). These results indicate the capacity of viruses to survive in soil for extended periods of time in cold or even cool climates. The length of survival of viruses at warmer temperatures is substantially reduced from cooler temperatures, although at 20°C viruses have been shown to survive for up to 60 days (DuBoise et al., 1976).

5.1.3 Survival of Helminths and Protozoa

Helminths and protozoa possess an evolved adaptation which enhances their survival under field conditions. Cysts and eggs are extremely resistant, although loss of moisture eventually destroys most parasitic cysts and eggs (Hays, 1977). In a study by Cram (1943), viable *Ascaris* eggs were found after 81 days in a sludge which had been drying in a greenhouse at temperatures which frequently reached 46°C. Indeed, the resistance of *Ascaris* is emphasized by a WHO (1972) report which states that *Ascaris* are known to survive in moist soils for several years.

5.1.4 Summary of Survival

Several factors affecting the survival of pathogenic organisms in soil would be expected to limit persistence in the case of the Pinelands soils. The low moisture holding capacity, as well as its low pH (3.5-4.5), and organic matter content would be expected to limit survival of microorganisms in soil. However, the long term disposal of septic tank effluent may raise the pH and the organic matter of a soil; the impact of these, and other changes caused by sewage disposal on a soil's capacity to remove pathogens from sewage has not been ascertained. During winter months when soil moisture is high and soil temperature low, survival times will be longer. In fact, virus particles adsorbed on soils may well survive with only a slight reduction in population during the winter months.

5.2 Movement

5.2.1 Transmission of Microorganisms

It is evident that, particularly during the winter, pathogenic organisms can survive in the Pinelands soils for extended periods of time, enhancing the possiblity of transmission of pathogens into the groundwater supply. The movement and infectivity of pathogens is verified by a Center for Disease Control (1974) report which linked an outbreak of gastrointestinal illness with a septic tank located 150 feet from the contaminated well. The efficiency of the soil as a filter for pathogens increases with the size of the organism. The eggs and cysts of parasitic pathogens are considered to be removed near the soil surface because the size of the <u>Ascaris</u> eggs generally exceeds the mean pore dimaeter of soils (Miller and Wolf, 1975). Bacteria are primarily removed by the mechanical process of straining at the absorption trench-soil interface due to the accumulation of suspended particles and the sedimentation of bacterial clusters (Gerba et al., 1975). Viruses are the smallest pathogens which may contaminate groundwater, and the most important factor affecting virus retention in soils is adsorption to soil particles (DuBoise, 1979).

The sandy soils of the Pinelands could produce a high degree of bacterial purification during conditions of unsaturated flow. During periods of unsaturated flow, liquid moves only through the smaller pores of the soil (McCoy and Ziebeil, 1977). McCoy and Ziebeil (1977) also noted that, after the first 100 days of loading, a "clogging zone" develops at the soil-effluent surface resulting in a filtering effect which removes many bacteria. However, Hagedorn et al. (1978) found that rainfall washed large numbers of bacteria from a drain field which moved as a front in the direction of the water flow.

Reneau (1978) stated that the greatest potential for failure of a septic tank occurs during periods of high water tables. In the study by Brown et al. (1977) 120 cm of a sandy soil was sufficient for the removal of coliforms and coliphage. Hagedorn (1978) used antibiotic resistant bacteria to monitor bacterial movement in soil. The indicator bacteria were detected in a 300 cm well within 24 hours after application. Hagedorn also stated that the population of indicator bacteria reached a maximum during intervals associated with the rise of the water table following major periods of rainfall. The movement of bacteria in this case may have been due to conditions of saturated flow. In a study of sand bed filtration of septic effluent containing between 970 and 1260 coliforms per 100 ml., Aulenback et al. (1975) concluded that essentially all of the coliforms were removed in the first five feet of bed depth. The implications of this data with respect to the Pinelands soils are that a minimum of 120 cm is required between the discharge point and the groundwater tables, and septic tanks should be installed and utilized with caution in order to prevent conditions of saturated flow which might short circuit the system allowing effluent to reach the groundwater unfiltered.

5.2.2 Transmission of Virus

The primary factor influencing virus movements in soils is adsorption. Salt concentration, pH, organic matter content and soil composition affect the degree of retention (Gerba et al., 1975). The addition of 10^{-2} N $CaCl_2$ and $MgCl_2$ to sand columns drastically reduced the percentage of virus particles which passes through a sand column, while at low cation concentrations most viruses pass through the sand column (Leffler and Kott, 1974). DuBoise et al. (1976) detected viruses in a sandy forest soil for up to 91 cm from the point of innoculation. They also found that simulated cycles of rainfall and effluent application, resulting in ionic gradients, released adsorbed virus particles from the soil. Gerba et al. (1975) also found desorption and increased subsurface travel due to changes in the water quality. Wellings et al., (1975) reported the greatest distance of virus migration in soil. They detected a burst of virus in 3 and 6 m wells after 28 inches of rain fell in a three month period. Bacteriophage retention in soils is directly correlated to the clay content of the soil (Drewry and Eliassen, 1968). In a study of virus movement in a sand, a sandy loam and a garden soil, Laak and McClean (1967) found the largest virus breakthrough in sand columns, while the least breakthrough occurred in a sandy loam containing 7% clay.

Most of the studies evaluating virus adsorption to soils used bacteriophage or poliovirus type 1 as indicator. A study by Goyal and Gerba (1979) examined adsorption of several human enteroviruses and bacteriophages to nine soils, and concluded that virus adsorption to soils was highly strain specific. The most significant factor affecting virus adsorption was pH, with adsorption greatest in soils having a saturated pH of less than 5. In addition, soluble organics were shown to compete for adsorption sites as all of the viruses studied adsorbed less to soil in the presence of secondary treated sewage.

In a review of virus movement in soils, Sproul (1975) concluded that deep fine textured soils are most effective in the removal of viruses, although sandy soils may be effective provided they are deep enough. He recommends that the minimum acceptable depth between the point of application and groundwater be 5 feet, and that application rates of effluent should not exceed 0.4 to 0.7 gal/sq.ft. of bottom area/day. At greater flow rates virus adsorption is decreased. The design criteria of the U.S. Public Health Service Manual of Septic Tank Practices (1967) may be used with percolation test data for soils in the different hydrological groups. This data may be used to estimate application rates. Such calculations reveal that typical application rates for soils in hydrological groups A, B, and C are 2, 0.8, and 0.4 gal/ft^2/day, respectively. For septic disposal systems designed according to the Public Health Service criteria, too much water would be applied to soils in hydrological group A to allow for effective virus removal. Design application rates for the less permeable members of group B may be effective, but for the more permeable members of group B flow rates may be too great to retard virus movement. Design application rates for soils in group C would be adequate to meet the criteria suggested by Sproul (1975).

If virus movment into groundwater is demonstrated, the survival of pathogens in groundwater becomes a factor. McMichael and McKee (1965) reported a 98% loss of titer of poliovirus 3 in 11 days in water. The half-life of pathogenic bacteria in well water ranged from 2.4 to 26.8 hours in a study by McFeters et al., (1973), while a study by Mitchell and Starzyk (1975) found the 90% reduction times for *Salmonella typhimurium* and *Streptococcus faecium* to be seven, and more than 20 days, respectively.

5.2.3 Transmission Limitations

These results indicate that the transmission of pathogenic viruses into groundwater may be a problem in the Pinelands soils; the low cation exchange capacity and clay content of these soils will provide little resistance to virus movement, although the low pH would enhance adsorption. During winter months when survival is extended, substantial amounts of rainfall would desorb virus particles and permit their migration through the soil. It appears that a minimum of five feet between the tile drain and the water table will be required to provide adequate protection from viral contamination of groundwater. As a septic leach field ages, the organic "crust" which forms at the trench-soil interface acts as a medium for entrapment of bacteria and virus and for bacterial growth. In highly permeable soils, rapid infiltration following heavy rains can desorb and wash to the groundwater the bacteria and virus entrapped by the crust. Water will not move rapidly enough in less permeable soils to cause migration from crusts to a water table which is greater than 6 feet below an absorption trench. However, the demonstration of movement by pathogenic organisms into groundwater does not completely define the inherent risk since there is a lack of definitive data indicating the dose of pathogen required for infection by the oral route (DuBoise et al., 1979).

6.0. ORGANICS

An increasing number of organic substances are developed and put into use each year. Many of these are used in the household, while others are used as biocides, wetting agents and solvents which are applied to home lawns and agricultural fields. It has also recently become evident that the reaction of some household products including chlorine with the organics in sewage effluent may result in the formation of chemicals which are similar to manufactured biocides.

While many of the toxic organic compounds entering the soil may vaporize, be degraded by microorganisms or be adsorbed onto the soil surface, others are quite resistent to degradation and have properties which allow them to move quickly through the soil to the groundwater.

Many organic substances have extremely low solubility in water. This generally prevents large amounts of these compounds from entering the groundwater. However, because many of the substances are toxic at very low concentrations, solubility constraints are often not capable of totally preventing migration at levels which could be biologically significant. A comparison of the solubilities of selected pesticides with the maximum permissible concentrations in drinking water (Table 6.1) indicates that solubilization generally exceeds the permissible concentrations of these pesticides.

The organic substances which pose the greatest threat to groundwater resources are those which are relatively soluble, have low volatility and are resistant to degradation. For such chemicals, we must rely on the adsorption capacity of the soil to prevent their migration. Adsorption characteristics depend mainly on surface area, organic matter and pH. While the acid conditions in the Pinelands soil may retard the mobility of some organic chemicals, many of those commonly used will be more mobile under acid conditions. Both the very high sand content and the low native organic matter content of the soils will result in low adsorption capacities. Thus mobility of certain undesirable organic constituents through the unsaturated zone to the

TABLE 6.1 Comparison of maximum permissible concentration limits in drinking water and the solubilities of six pesticides (Freeze and Cherry, 1979).

Compound	Maximum permissible concentration (mg/l)	Solubility in water (mg/l)
Endrine	0.0002	0.2
Lindane	0.004	7
Methoxychlor	0.1	0.1
Toxaphene	0.005	3
2,4-D	0.1	620
2,4.5-TP Silvex	0.01	-

groundwater is expected to be rather rapid in the Pinelands soil.

Significant inputs of organics to groundwater in the Pinelands may come from septic effluents and from the biocides either used in agricultural practices or household pest control. These inputs become significant because of the low attenuating capacity of soils in the Pinelands and the pristine nature of its groundwater resources.

6.1 Septic Effluent Inputs to Groundwater

A partial list of the organic components of septic tank effluents is given in Table 6.2. Based on the analysis of leachate from sand lysimeters, 14% of the total organic carbon fraction passed through the soil with the first water front. Sandy soils of the Pinelands region have a low ability to attenuate septic effluents because of their low clay content and low native organic matter content. The effectiveness of a soil to renovate specific components in septic effluents such as biocides is a function of the soil's pH, sorption area and ionic exchange capacity. Properties that effect a biocide's attenuation in effluents are its volatility, molecular size, and ionic characteristics (Miller et al., 1977).

6.2 Biocide Inputs to Groundwater

As used here, biocides are organic chemicals used to combat pests or interfere with plant growth. In areas without appreciable

TABLE 6.2 Organic constituents of septic tank effluent (Miller et al. 1977).

COMMON	
Proteins	Organic acids
Lipids	Phenolic compounds
Nucleic acids	Surfactants
Polysaccharides	
OCCASIONAL	
Gasoline	Phenols
Biocides	

industrial development or waste disposal, the major biocide inputs to groundwater will be from agricultural and household use. While available literature indicates little input of biocides to groundwater, this evidence is largely based on the strongly adsorbed (to soil components) organochlorine pesticides (Giger et al., 1978).

6.2.1 Agricultural Inputs

Inputs of biocides to groundwater from agricultural practices in New Jersey are expected to come largely from the state's two main crops: corn and soybeans. Smaller contributions to groundwater contamination may come from the various small grains and forage crops grown.

In Production Recommendations for Field Crops for 1979, the New Jersey Cooperative Extension Service outlined a recommended biocide use plan for each major crop. While implying no endorsement, the report did refer to names of available herbicides and insecticides that might be used. Table 6.3 is a summary of these biocides and the crop for which they are used.

6.2.1.1 Corn. Often corn is preceded by a fall planted cover crop such as a rye. A knockdown contact herbicide is used to kill the rye prior to corn planting. Examples of such herbicides are Paraquat, 2,4-D and Roundup. Since most weed damage is done to corn in the first 3-4 weeks after emergence, early season weed control is widely practiced. Preplant emergence, preemergence, and post emergence herbicides include

TABLE 6.3 Herbicides and pesticides for use on crops in New Jersey (New Jersey Cooperative Extension Service, 1979).

Type Chemical	Corn	Soybeans	Small Grains	Forages
Herbicides	Atrazine	Treflan	Roundup	Eptam
	2,4-D	Vernam	Buctril	Tolban
	Paraquat	Tolban	Brominal	Balan
	Roundup	Paraquat	Bronate	Butyrac
	Bladex	Roundup	2,4-D	Kerb
	Sutan+	Premerge	Banvel	Premerge
	Eradicane	Lorox	-	Paraquat
	Lasso	Dintro	-	Furloe
	Dual	Alanap	-	Princep
	Prowl	-	-	Roundup
Pesticides	Diazinon	Dimethoate	Malathion	Azinphosmethyl
	Malathion	Malathion	Parathion	Furadan
	Parathion	Methomyl	Captan	Imidan
	Sevin	Parathion	Maneb	Malathion
	Furadan	Sevin	Thiram	Methoxychlor
	Methomyl	Trithion	Hexachloro-benzene	-
	Captan	Benlate	Vitavax	Diazinon
	Dichlone	Captan	-	Parathion
	Dyfonate	Maneb	-	Supracide
	Maneb	Thiram	-	-
	Thiram	Nemacur	-	-
	-	Mertect	-	-

Atrazine, 2,4-D and several others as listed for corn herbicides in Table 6.3. Pesticides used for corn include insecticides (Sevin, Parathion, Diazinon, Malathion) fungicides (Captan, Dichlone, Maneb, Thiram) and Nematicides (Furadan).

6.2.1.2 Soybeans. Since the early season weeds reduce soybean yields, preplant, preemergence and post emergence herbicides are widely used. Examples of these include Treflan, Vernam, Tolban and Premerge. In reduced tillage operations, Paraquat and Roundup are used as knockdown (contact) herbicides. Pesticides used on soybeans include insecticides

(Dimethoate, Parathion, Melathion, Methomyl, Sevin, Trithion) nematicides (Nemacur) and fungicides (Benlate, Mertect).

6.2.1.3 Small grains. Small grains grown in New Jersey include wheat, oats, barley and rye. Most of the grains are fall-seeded and serve as cover crops for the winter months. Since weeds are seldom a problem for these crops until early spring, post emergence herbicides are used. Examples of the chemicals are Buctril, Brominal, Bronate, 2,4-D and Banvel. Pesticides used on small grains include insecticides (Malathion, Parathion) and fungicides (Captan, Maneb, Thiram).

6.2.1.4 Forages. Forage crops grown in New Jersey include legumes (alfalfa, clovers, trefoil and lespedeza) and grasses (sudangrass, sorghum, bromegrass, and orchard grass). Herbicides used for these crops include preplant incorporated types (Eptan, Tolban, and Balan) and postemergence types (Butyrac, Furloe, Kerb, Premerge and Princep). Pesticides used for forage crops include Azinphosmethyl, Furadan, Parathion, Diazinon, Itmidan, Malathion, Methoxychlor and Supracide.

6.2.2 Household Inputs

Household use and disposal of pesticides is common in the United States. In areas such as the Pinelands which are not served by a community sewage treatment plant, the eventual depository of biocides used in the home is often the onsite septic system. Table 6.4 is a list of commonly used household pesticides.

6.2.3 Future Trends

While no large growth is expected for agricultural operations in the Pinelands region, it is possible that the number of households in the area will increase dramatically. Biocide inputs to groundwater would follow the trend of any increase in numbers of households with septic systems.

6.2.4 Soil Attenuation of Biocides

A soil's ability to attenuate biocides is directly related to its clay content and native organic matter content. (Duff et al., 1973; Fusi and Franci, 1971; Kawamori et al., 1971). Relative soil attenuation of several biocides is given in Table 6.5.

Various biocides are applied to soil with surfactants in agricultural operations. Studies have shown this to increase the soil mobility of herbicides and pesticides (Koren, 1973; Pitblado et al., 1972). In sands, even cationic biocides such as paraquat are leached downward by the saturated water flow (Watkin and Sager, 1971.) Since

TABLE 6.4 Commonly used household pesticides.

Pyrethrum	Carbaryl
Diazinon	Dysiston
Malathion	Allethrin
DDVP	Rotenone
Propoxur	Chlorpyrifos

Atrazine's mobility in sand is dependent on the soil organic matter content (Voitekhova, 1971), a sandy aquifer with low native organic content is an optimal medium for its movement (Schneider et al., 1970).

Thus, biocides from both agricultural fields and septic leach fields could enter the groundwater. Due to the complexity of the problem, however, there is no good means of predicting actual fluxes.

TABLE 6.5 General solubility, mobility and breakdown rate of some groups of biocides (Steenvoorden, 1976)

Biocide-Group	Solubility	Breakdown Rate	Mobility
organochlorine	very low	years	very low
organophosphate	low	weeks to months	low
carbamate	low-medium	days to weeks	low
ureum	low-medium	months to years	low to high
fenoxy-alkyl acid	medium	weeks to months	high
triazine	low	weeks	low

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6.3 Nitrosamines

The formation of nitrosamines in soil is a potential threat to public health because these compounds are the most broadly acting and among the most potent carcinogens known (Lijinsky, 1977). Ayanaba and Alexander (1974) and Pancholy (1978) have demonstrated the ability of nitrites to form nitrosamines in the presence of primary or secondary amines in water, sewage, or soil. The precursors of nitrosamines are quite common in the environment. Nitrites can be generated from fertilizers, sewage and food which contain nitrates (Lijinsky, 1977). Primary and secondary amines occur in urine (Brooks et al., 1972), feces (VanRheene, 1962), higher plants (Smith, 1971), and as a result of the degradation of pesticides (Ayanaba and Alexander, 1974). Thus, it is imperative that the fate of nitrosamines in soil be understood, and that the passage of nitrosamines into groundwater be prevented. Ayanaba and Alexander (1973) and Mills and Alexander (1976) demonstrated the formation of nitrosamines in soil and sewage supplemented with dimethylamine and nitrites. The highest rate of nitrosamine formation took place at a low pH, although the persistence was greater at a higher pH. The highest rate of formation in soils took place at a pH of 5.2, while nitrosamine formation in sewage was most rapid at a pH of 3.5. The potential for nitrosamine transmission exists in part because of the persistence of these compounds in soil and water. Tate and Alexander (1975) found a lag of nearly 30 days before their slow disappearance from soil, while in lake water no degradation was observed during a 100 day period. Overcash (1979) indicated that 75-150 days were required for complete reduction of nitrosamines applied to soils at a rate of 15-20 ppm.

The presence of nitrosamine in soil does not present a human health hazard unless it can be demonstrated that nitrosamines leach through soil or translocate into plants. When lettuce and spinach were grown in nitrosamine amended soil, dimethylnitrosamine was assimilated by the roots and translocated to the tops of the plants (Dean-Raymond and Alexander, 1976). In addition, nitrosamine moved through the soil as rapidly as chloride. Overcash (1979) disputed this finding and stated that the polarity of nitrosamines should result in adsorption to soils. However, competition for adsorption sites with other water soluble organic compounds in sewage may inhibit nitrosamine adsorption. These results indicate that the potential for environmental contamination with nitrosamines is great.

However, the formation of carcinogenic nitrosamines has not been demonstrated, except in samples which were amended with both methylamine and nitrite. Yet the widespread occurrence of these precursors in nature, and the potency of nitrosamines dictate the use of all available precautions for the prevention of nitrosamines entering the food chain. Such chemicals could form in the soil surrounding septic leach fields, and could possibly move to the groundwater, particularly in very sandy soils.

7.0 CONCEPTUAL MODEL AND ANALYTICAL PROCEDURE

7.1 Areal Model

Dilution models may be used to calculate the amount of acreage which must be set aside in order to dilute the pollutant received from a point source to an acceptable concentration. It must be recognized, however, that while this procedure provides a method of predicting the average concentration and the acreage required to achieve that average, water entering the groundwater from one area is not completely mixed with that entering the groundwater from an adjacent area. Therefore, areas of concentrations, in some cases well in excess of the average values calculated here, will be expected to occur.

7.1.1 Nitrogen

Four models are proposed which may be used to calculate the land areas required to limit the groundwater nitrate contributions from various nitrogen sources. Since the effects on groundwater of septic systems, secondary effluent irrigation, and lawn and agricultural fertilization are additive, each can be considered separately. Nitrogen added to soils via rainwater is not considered in the models. Rainwater nitrogen inputs are small, reported by Brady (1974) from 0.6 to 1.4 ppm in rainwater, compared to the quantities added by waste management and fertilization practices. Additionally, the background nitrate nitrogen concentration of 0.17 ppm reported for Pinelands groundwater already reflects rainfall inputs.

7.1.1.1 Septic. The models used by Brown et al. (1977) and Trela and Douglas (1978) were compared to ascertain a suitable method of calculating the acceptable spacing for septic fields on soil groups A and B.

Trela and Douglas (1978) used a very simple formula derived from chemical dilution principles. It is:

$$\frac{Ve(Ce-Cq)}{Vi\ Cq} = \frac{acres}{person}$$

where: Ve = volume of effluent in gal/person/year,
Ce = concentration of pollutant in the effluent in mg/l,
Cq = concentration of pollutant required by water quality standards in mg/l,
Vi = volume of infiltrating rainwater in gal/acre/year.

The formula used by Brown et al. (1977) to calculate the area needed to maintain a given water quality was:

$$A_t = A_f + \frac{\left(\frac{FL_f}{C} - D_f\right)A_f}{D_o}$$

where: A_t = the total area (ha)
A_f = the area of the septic field in ha (figured to be 120 cm wide effective areas for 30 cm wide trenches)
F = the unit conversion factor = 10
L_f = the flux of NO_3-N below the septic field area (kg/ha/yr)
C = the concentration of NO_3-N (ppm)
D_f = the equivalent depth of percolate below the open acres (cm/yr)
D_o = the equivalent depth of percolate below the open acres (cm/yr)

Both of the equations give the same result when the same input data are used. They both use rainwater (assumed to be free of nitrogen) to dilute the nitrate added by the septic leach fields.

For the volume of the infiltrating rainwater, Douglas and Trela (1978) used the average less the standard deviation, which equaled 12.3 inches of leachate per year. This method was selected to provide groundwater protection even during dry years, and results in requirements of larger set aside acreages than would be calculated from mean leachate volume. The use of the long term average groundwater infiltration seems to be a more logical approach since the entire aquifer is not replenished each year, and the impact on the groundwater concentration would be averaged over many years. Rhodehamel (1970) reported that 20 inches of water reached the groundwater table below the Pinelands each year. Therefore the 20 inches per year value is used here. Current literature indicates that the acid sandy soils in the concerned area will not support denitrification and therefore the denitrification loss of 30% in the group B soils assumed by Trela and Douglas (1978) is far too great. For the present calculations, denitrification losses are assumed to be negligible.

Additionally, data by Brown et al. (1978) shows that as much as 9% of the N applied to sandy soils may be taken up by vegetation during a growing season about twice as long as that existing in New Jersey. Therefore, vegetative uptake was calculated based on the length of growing season and of the size of the septic fields. The acceptable densities of septic fields were then calculated using D_0 = 20 inches, no denitrification and 4.5% grass uptake on soils in group A and 9% on the finer textured soils in group B. A family of 3.5 persons is assumed to generate a septic effluent nitrogen concentration of 44.6 mg/l (Table 2.4) at a rate of 11.2 mg/capita/day (Siegrist et al., 1977; Kuhner et al., 1977; Laak and Crates, 1977). This is calculated to be equivalent to 66 gallons of waste water per person per day. Results are presented in Table 7.1 and are considerably lower than the 7.9 acres/family reported by Trela and Douglas (1978) to keep levels to 2 ppm.

TABLE 7.1 Calculated area needed to dispose of septic water with an average NO_3-N content of 44.6 mg/l for a family of 3.5 persons.

Soil	Groundwater conc. of 10 ppm	Groundwater conc. of 2 ppm	Groundwater conc. of 0.17 ppm
	acres	acres	acres
Group A	0.51	3.2	39
Group B	0.47	3.0	37

For a family of 3.5 persons, an area of 0.47 acres would be required to limit groundwater additions to 10 ppm, 3 acres would be needed to keep it to 2 ppm and 37 acres would be needed to keep it to 0.17 ppm in a soil in hydrological group B. Uptake of nitrogen by grass has only a small impact on the area required, and vegetation harvest would be required to remove the nitrogen from the system.

7.1.1.2 Sewage effluent irrigation. A convenient model for calculating the impact of sewage effluent irrigation on the groundwater was suggested by Loehr et al. (1979). The depth of irrigation water which can be applied and still keep the nitrate concentrations in the leachate below a certain level can be calculated from the equation:

$$W = \frac{4.43C + a(P-ET)}{y - a - y(d+v)}$$

where W = wastewater addition (inches/month)
C = crop removal of nitrogen (lb/acre/month)
a = allowable nitrogen in percolating water (ppm)

P = precipitation (inches/month)
ET = potential evapotranspiration (inches/month)
y = total nitrogen concentration in waste water (ppm)
d = fraction which is denitrified
v = fraction which is volatilized as ammonia.
4.43 = units conversion factor.

As a fail-safe approximation, the ammonia volatilization and denitrification values have been set at zero. In the case of research findings relating to the specific site, these approximations could be relaxed.

The crop removal of nitrogen is estimated as 100 lb/acre/yr for a year round vegetative cover. The monthly break-down per acre was taken as:

J - 0 lb	M - 11 lb	S - 11 lb
F - 0 lb	J - 22 lb	O - 5.5 lb
M - 0 lb	J - 25 lb	N - 0 lb
A - 5.5 lb	A - 20 lb	D - 0 lb

The total nitrogen concentration in the waste water was taken to be 20 ppm. Typically 1/4 of this is as organic form and the remainder is nitrate and ammonia. The monthly average weather data summarized in Table 7.2 was utilized to calculate acceptable depths of application to prevent the groundwater from receiving more than 10 ppm; the calculations were repeated again to keep the leachate water concentration below 2 ppm. The resulting acceptable depths are presented in Table 7.3. January and February are the most limiting months, and if irrigation is to be done without storage, no more than 2.1 to 2.2 inches per month could be applied to prevent the leachate water from exceeding 10 ppm. If the protection level is set at 2 ppm, no more than 0.2 inches could be applied during these months, while the 0.17 ppm level allows only 0.02 inches of effluent.

For a sewage treatment plant serving 5000 individuals, each using 66 gallons per day, a typical level of use for sewered communities, an acreage equivalent to 250 acres would be required for the 10 ppm limit, while 2250 acres would be required for irrigation disposal of water if the groundwater were to be protected from receiving greater than 2 ppm. Disposal by irrigation is not feasible if the 0.17 ppm leachate N limit is adapted since greater than 25000 acres would be necessary for a community of 5000 persons. If an areal approach is taken, dilution credit could be given for untreated undeveloped adjacent areas.

Irrigation disposal thus could be effectively utilized for treated effluent without raising the elevation of the groundwater above specified limits. If the limits are low, however, very large acreages would be required.

TABLE 7.2 Precipitation and evapotranspiration values for Southern New Jersey.

Month	Precipitation in inches*	Evapotranspiration in inches**
January	3.18	1.0
February	3.08	1.0
March	4.05	1.2
April	3.34	2.0
May	3.36	3.0
June	3.24	3.2
July	4.67	4.0
August	4.50	4.0
September	3.21	3.2
October	2.86	2.3
November	3.86	1.3
December	3.73	1.0
Annual total	43.08	27.2

* Based on data collected by the U.S. Weather Bureau from 1941-1970.

**After Franklin (1979).

TABLE 7.3 Monthly and total application rates of secondary treated sewage effluent containing 20 ppm nitrogen which would not raise the leachate concentration above 10 ppm. The calculations are repeated for 2 and 0.17 ppm.

Month	Waste water application rate which would limit N concentration in percolating water to 10 ppm. (in/mo)	Waste water application rate which would limit N concentration in percolating water to 2 ppm. (in/mo)	Waste water application rate which would limit N concentration in percolating water to 0.17 ppm. (in/mo)
January	2.18	0.24	0.02
February	2.08	0.23	0.02
March	2.85	0.32	0.02
April	3.78	1.50	1.24
May	5.23	2.75	2.46
June	9.79	5.42	4.92
July	11.75	6.23	5.59
August	9.36	4.98	4.47
September	4.88	2.71	2.46
October	3.00	1.42	1.23
November	2.56	0.28	0.02
December	2.73	0.30	0.02
Total	60.19 in/yr	26.38 in/yr	22.47 in/yr
Total without storage*	24.96 in/yr	2.76 in/yr	0.24 in/yr

*Calculated using February as the limiting month.

7.1.1.3 Fertilizer

7.1.1.3.1 Home lawns. From the summary of available literature it appears that up to 52% of the N applied as inorganic N may be lost as NO_3^- which is leached to the groundwater. If slow release organic N sources are used the average nitrate loss drops to about 33% of the applied N.

For purposes of calculating N losses from lawns the following three possibilities will be considered.

1. A 1000 ft^2 house with a 1 car garage and 50 ft long driveway on a 0.25 acre lot. All land not occupied by the house and drive will be lawn.

2. A 1500 ft^2 house with a 2 car garage and 200 ft long drive on a 1.0 acre lot. Eighty percent of land not occupied by the house and drive will be lawn.

3. A 2000 ft^2 house with a 2 car garage and 500 ft^2 utility building with 1.5 acres of lawn on a 5 acre lot.

For purposes of this model it is assumed that the average homeowner will fertilize a lawn with 2 lb N/1000 ft^2 in April-May and 1 lb N/1000 ft^2 each June and August. This amounts to a total annual application of 36, 126 and 261 lbs N for cases 1, 2, and 3 respectively. If one assumes that the fertilizer is added from an inorganic source, the literature reviewed suggests that up to 18.9, 65.7 and 135.9 lbs N would be lost as NO_3-N from cases 1, 2, and 3 respectively. If organic N sources were used, losses would be 12.1, 42.0 and 87 lbs N for each of the cases. According to the data of Rhodehamel (1970) an average of 20 inches of water leaches to the groundwater annually. This would create NO_3-N concentrations of 16.9, 14.7 and 6.1 in cases 1, 2 and 3 with inorganic applications, and 10.7, 9.4 and 3.9 in cases 1, 2 and 3 with organic sources. Note the range of NO_3-N concentrations in the leachate: from a high of 16.9 in case 1 with inorganic N to a low of 3.9 in case 3 using organic sources.

The worst case calculations reveal that a buffer area of 0.2 acres/0.25 acre lot will be required to maintain 10 mg/l NO_3-N or less in water reaching the groundwater or 1.9 acres to maintain 2 mg/l NO_3-N or less in the leachate. To maintain a 0.17 ppm leachate NO_3-N limit, 24 acres of buffer land is necessary. This would be for small lot (1/4 acres) developments assuming that all other ground is lawn or covered by roads, houses, etc. Buffer area may take the form of parks etc., provided they are also heavily fertilized and maintained. As the lot sizes increase, and the actual amount of lawn area increases, more buffer area is needed. For five acre lots (case 3) using organic N sources, 0.4, 8.1, or 111 acres buffer is needed per lot to maintain the 10, 2, or 0.17 ppm NO_3-N level (Table 7.4).

TABLE 7.4 Extra land required as a buffer zone.

Case	10 ppm N in leachate		2 ppm N in leachate		0.17 ppm N in leachate	
	Inorganic N fertilizer	Organic N fertilizer	Inorganic N fertilizer	Organic N fertilizer	Inorganic N fertilizer	Organic N fertilizer
1 (acre)	0.2	0.06	1.9	1.1	24	15
2 (acre)	0.7	0.2	6.5	3.9	85	54
3 (acre)	1.5	0.4	13.5	8.1	175	111

If buffer areas are not set aside, then lawns should be prohibited, or fertilization would need to be discouraged. The use of low-nutrient requiring native vegetation and organic nitrogen sources on very small lawns may be a desirable alternative.

7.1.1.3.2 Agricultural fields. A nitrate leaching model, developed by Frère (1976) is described here to quantitatively assess the percentages of nitrogen applied as ammonium fertilizer which would move below the root zone and into the groundwater. The assumptions made in this model include: (1) any nitrogen present in soil before fertilization is ignored as are denitrification losses; (2) no nutrients are taken up in the winter by weeds or cover crops; (3) water flow through the soil during the dormant period is assumed to be by piston flow.

Piston flow describes the downward movement of soil water when the upper layers of soil become saturated. The depth increment of the downward moving water resulting from the i^{th} rainfall event which exceeds the soil storage capacity is given as:

$$Z_i = \frac{Y_{12}(i)}{\theta_F - E}$$

where: Y_{12} (i) is the volume of water moving from the first layer as a result of the i^{th} rainfall event.
θ_f is the volumetric field capacity;
E is an exclusion factor.

θ_f is the volumetric field capacity,
E is an exclusion factor.

The exclusion factor accounts for the soil water in smaller that do not contain nitrate. A number of studies (Kolenbrander, 1970; Levin, 1964; Yaalon, 1965) have shown that anions, like nitrate, move with the wetting front and thus will move faster than the total soil water does, especially under conditions of saturated flow.

When ammonium fertilizer is applied, it is assumed that ammonium is converted to nitrate. Five days after application the following temperature-dependent relationship is used for the nitrification process:

$$N(t+1)=N(t) + K(T) + A(t)$$

where: $N(t)$ is the nitrogen in the nitrate form on day (t)
$A(t)$ is the nitrogen in the ammonium form on day (t)
$K(T)$ is a temperature-dependent rate function given by the following equations:

$$K = .0032\ T - .012;\ 10°C < T < 35°C$$

$$K = .00105\ T + .000095T^2;\ 0°C < T \leq 10°C$$

$$K = 0 \quad T<0°$$

where: T is the soil temperature °C (Minshall, 1967).

Therefore the conversion of ammonium nitrogen to nitrate nitrogen is a temperature-dependent relationship and the lack of this conversion in cold winter months accounts for a buildup of ammonium nitrate.

As stated earlier, nitrate-nitrogen accumulates in the upper soil layers until available water storage capacity is exceeded by infiltration during a rainy day. As each precipitation event occurs in which the infiltration exceeds the water storage capacity of the soil, that amount of nitrogen which has been converted to nitrate, but which has not been taken up by the plant, is moved downward to a new depth.

On the date of full root extension, the percentage of nitrate loss can be calculated. All the nitrate below the root zone is summed and divided by the ammonium nitrogen which was added at fertilization to yield the percentage lost. Table 7.5 indicates the leaching depths for hydrological group A and B and the corresponding percentages of nitrogen lost for both spring and fall fertilizer applications as found for the New Jersey Pinelands.

Table 7.6 calculated by Frere (1976) lists the recorded rate of nitrogen fertilizer applied to corn during spring and fall onto soils in hydrological groups A, B and C. The table also illustrates the lb/acre and ppm nitrogen which would be lost to groundwater with the stated fertilizer application. The results are in essential agreement with

...ucs of Douglass (1976) that the concentrations of NO_3-N .. leachate from spring applied nitrogen in the the concentrations are in the range of 2-4 ppm NO_3-N.

TABLE 7.5 Leaching depths and percentage of Spring and Fall nitrogen fertilizer application lost to groundwater for the New Jersey Pinelands (after Frere, 1976).

	Hydrological Group		
	A	B	C
Annual leaching depth (inches)	17	12	11
% of fall fertilizer lost	65	40	18
% of spring fertilizer lost	20	5	0

TABLE 7.6 Nitrogen fertilizer applied to corn, resulting nitrate concentration in groundwater, and acreage needed (in addition to cropping fields) for dilution of groundwater to 2 or .17 ppm nitrate for the New Jersey Pinelands (after Frere, 1976).

	Spring Application			Fall Application		
	A	B	C	A	B	C
fertilizer applied (lb/acre)	75	75	75	75	75	75
N entering ground-water (lb/acre)	15	3.8	-	48.8	30	13.5
NO_3-N entering groundwater (ppm)	3.9	1.4	-	12.7	11.1	5.42
Area needed (in addition to acreage of cropped fields) for dilution to 2 ppm	1.0	-	-	5.4	4.5	1.71
Area needed (in addition to acreage of cropped fields) for dilution to 0.17 ppm	22	7.2		73.7	64.2	9.0

In order to meet the desired 2 or 0.17 ppm groundwater nitrate concentration, acreages necessary to dilution of the fertilizer nitrates are suggested. It is apparent the concentrations of nitrates in the groundwater could be increased by fertilizer applications. Proper management of agricultural fields or any area of nitrogen fertilizer application must include "buffer zones" to allow for dilution if the nitrate concentration is to be moved below 2 or 0.17 ppm for spring application. Fall fertilization should be discouraged due to low nitrogen uptake by plants.

7.1.2 Phosphorus

The literature abounds with reports of the accumulation and migration of phosphorus in the soils and groundwater below septic leach fields. While there is a background level of phosphorus in septic effluent originating from organic wastes, the use of phosphorus detergents can result in much higher phosphorus levels.

The mechanisms of phosphorus removal reviewed above indicate that the adsorption capacity of the soil must be relied upon as the primary mechanism for retarding the movement of phosphorus through the unsaturated zone to the groundwater. Another mechanism which could account for some removal would be plant uptake. Phosphorus loading rates in a septic field are expected to be such that plant uptake could account for only a very small amount of removal.

No measurements of P uptake by plants over a septic leach field are available, but our measurements of nitrogen uptake (Brown and Thomas, 1978) indicate that under conditions including a complete grass cover under a long growing season only 9% of the applied nitrogen was removed by vegetation, and it is suggested that removal of phosphorus would be of the same magnitude or less. Therefore, for the present purpose, phosphorus removal is expected to be dependent only on the adsorption mechanisms.

Phosphorus adsorption capacities have been reported for only one of the soil series of interest, but the values should approximate those of the other sandy soils in hydrological group A. An average phosphorus adsorption of 0.34 meq/100g soil was reported for the surface horizon of the Lakewood soil by Toth and Bear (1947). Thus adsorption capacity is equivalent to an adsorption capacity of 0.034 mg P/g soil. Laboratory tests may not fully evaluate the time dependent phosphorus adsorption mechanisms. However, Ellis and Childs (1973) reported that field observation of the maximum adsorption did not differ greatly from laboratory results. It is known that in the presence of iron phosphorus adsorption may be greater than the levels Toth and Bear (1947) reported for the surface soils. There is no data, however, which would allow evaluations of just how much greater the phosphorus adsorption would be for the acid subsoils under the Pinelands. Data reported by others, however, indicate that phosphorus adsorption may be as much as 10 times greater than reported for the Lakewood surface soil.

The phosphorus adsorption capacities may be utilized to calculate how much time will be required to saturate the adsorption sites below a septic leach field, and thus how long it will take for the phosphorus to break through to the groundwater.

The total adsorption capacity of the soil can be calculated as follows:

$$A = a\rho V$$

where: A is the phosphorus adsorption capacity of the soil between the the point of application and the water table (Kg)
a is the phosphorus adsorption capacity (mg/g)
ρ is the density of the soil g/cm^3
V is the volume of the soil (cm)

The volume is calculated as

$$V = bTD$$

where:

b is a factor to account for the horizontal spread of the phosphorus in the unsaturated zone.
T is the bottom trench area and
D is the distance between the bottom of the field and the water table.

For the present purpose, ρ is taken as 1.5 g/cm^3, a value typical for sandy soils. The parameter b is taken as 2, indicating a volume of soil twice that found directly below the bottom of the trenches is available for phosphorus adsorption.

Trench bottom area (T) is based on the design criterion developed for the infiltration rates. For soils in hydrological group A with permeabilities of 6 to 10 inch/hr, the design standard for a three and one half member household would call for bottom trench surface areas of 285 and 210 ft^2, respectively. For soils in hydrological group B, with permeabilities ranging from 6 to 1.5 in/hr, the design standards for a three and one half member household would call for bottom trench surface areas ranging from 360 to 720 ft^2, respectively.

These data were utilized to calculate the phosphorus adsorption capacities of a 6 ft. layer of soil between the bottom of the septic trench and the water table. The results are given in Table 7.7.

The adsorption capacities range from 4 kg from the smallest designed field in soil hydrological group A to 137 kg for the large field in soil hydrological group B with the greatest adsorption capacity. Low production of phosphorus is 0.75 kg/person/year, with typical values of 2.2 kg/person/year while the highest reported rates are 8.4 kg/person/year. These production rates are used to calculate the amount of time required to saturate the adsorption capacities of the soils.

The results of the calculations are given in Table 7.8 for the low, average and high production rates reported in the literature. For soils in hydrological group A receiving the average amount of phosphorus,

TABLE 7.7 Adsorption of phosphorus by 6 ft layers of soils below septic leach fields for soils of different hydrological groups and adsorption capacities. Septic systems are designed for an average household of 3.5 people.

Soil Hydrological Group	Trench Area (ft^2)	Adsorption (Kg) 0.034/mg/g	0.34 mg/g
A	210	4.0	40.0
	285	5.4	54.0
B	360	6.9	69.0
	720	13.7	137.0

breakthrough would be expected in less than a year for soils with P adsorption capacities of 0.034 mg/g. Breakthrough in soils in group B could take as long as 1.7 years. Available data on the P adsorption indicates that this would be the case for Pinelands soils. If P adsorption is 10 times greater, between 5 and 17 years would be required to fill the adsorption capacity in the unsaturated zone, thereafter P would be expected to move into the groundwater if the water table was at a depth of 8 ft.

Thus phosphorus breakthrough for any of the soils at any loading rate is just a matter of time. Although movement in the soil with the greater adsorption capacity is very slow, with sufficient time the concentrations entering the groundwater will be those leaving the septic leach field. These would range from 5.4 to 61 ppm with an average level of 16.0 ppm. Once in the groundwater phosphorus movement would be a function of the adsorption capacity of the aquifer, and the rate of flow. Adsorption and dilution are the two mechanisms which would decrease the areal concentration. Concentrations in the plumes originating from the points of applications would be much greater than areal concentration.

In summary, phosphorus may move rapidly through sandy soils with low adsorption capacities. The only data available indicates that the phosphorus adsorption capacities of Pinelands soils is very low, and as a result phosphorus in concentrations similar to that found in septic effluent (16.0 ppm) would soon reach the groundwater. The rate of movement in the aquifer would depend on the flow rate and P adsorption capacity, both of which are unknown at this point. It therefore

TABLE 7.8 Time for 6 ft layers of soils to reach phosphorus saturation.

Soil Group	Trench Area (ft^2)	Time until P saturation (yr) 0.034 mg/g	0.34 mg/g
		Low production of P	
A	210	1.5	15.0
	285	2.1	21.0
B	360	2.6	26.0
	720	5.3	53.0
		Average production of P	
A	210	0.50	5.0
	285	0.68	6.8
B	360	0.87	8.7
	720	1.70	17.0
		High production of P	
A	210	0.14	1.4
	285	0.18	1.8
B	360	0.24	2.4
	720	0.47	4.7

appears that the groundwater would eventually be contaminated with excess phosphorus below septic fields in Pinelands soils. Calculation of ultimate concentrations could be done in a similar manner as those for nitrogen presence herein.

7.2 Plume Model

While the models discussed above allow approximations of the amounts of contaminants and concentration of contaminants reaching the groundwater, it is assumed that the contaminants will be diluted by water infiltrating from the surrounding areas. The models assume ultimately that the contaminated water and diluted water will be completely mixed. This approach yields an estimate of average concentration of a given parameter. However it is well known that water entering an aquifer from a point source such as a septic leach field is not mixed completely with leachate water. Instead a plume of the contaminants spreads from the point or area of introduction. Thus in order to calculate set back distances for property boundaries, wells, streams or other places where impaired groundwater quality will have an undesirable impact, is it necessary to trace the movement of contaminant from the point of introduction.

The spreading of a conserved contaminant, one which does not react with the medium and is not transformed in transit, will take place as a result of mass water flow in the direction of the hydraulic gradient, as well as in directions perpendicular to the mass flow in the horizontal and vertical directions as a result of hydrodynamic or mixing dispersion. Additionally, molecular diffusion may result in the spreading of contaminants in any direction, but in most permeable aquifers diffusion is very small compared to mass flow and the associated hydrodynamic dispersion. Diffusion calculations indicate that concentrations of 10% of that introduced into a permeable but stagnant aquifer would be found at a distance of only 1 meter from the point of introduction after a period of 100 years.

Thus a plume of a nonreactive contaminant will spread mainly in the direction of water flow, and the concentration at any point and time within the plume will depend on the concentration and areal extent of the source of contamination and the velocity of flow. In general, finite element models have been utilized to simulate the spread of a contaminant introduced into an aquifer. Such models have been developed by Pickens and Lennen (1976) for both nonreactive and reactive constituents, and the necessary input data could be developed to allow their use to calculate the concentrations of nitrate, for example, which would be expected to reach a well or a stream at some distance from a septic leach field.

Such solutions are somewhat simplified in that they do not account for the presence of layers or lenses with greater or lower permeability than the rest of the aquifer. In general such layers would tend to restrict the movement of water to narrower zones, and would thus result in greater concentrations of contaminants reaching a particular location than would be calculated from such a model. Nonethelss, such calculations would provide reasonable estimates of the spread of contaminants with time.

Since nitrate does not likely undergo transformations within the groundwater, it is apparent that with time plumes will spread horizontally resulting in zones of contamination throughout the water body. If a well or stream is located at some distance down gradient from such a zone, elevated concentrations of a mobile contaminant will eventually reach that location.

8.0 SUMMARY AND RECOMMENDATIONS

The existing literature was utilized to select, justify and develop information and models on the probable movement of nutrients, organic contaminants and pathogenic organisms to the groundwater from "standard" septic tank-absorption trench systems, fields irrigated with secondary treated sewage effluent, and soils fertilized for lawn and crop production. Table 8.1 summarizes the results from model calculations.

A model used to calculate the concentrations of nitrate likely to reach the groundwater below septic leach field accounts for the impact of dilution by water from adjacent unfertilized areas. There is insufficient evidence of denitrification or other loss mechanisms, except for a small amount removed by vegetation for which corrections are made. The model indicates that for soils in hydrological group B, which are likely to be suitable for septic leach fields, an area of 0.47 acres per household of 3.5 persons would be required to prevent the mean nitrate nitrogen concentration in the leachate water from exceeding 10 ppm, an area of 3.0 acres would be required for the same size household to prevent the nitrate concentration from exceeding 2.0 ppm. An area of 10.5 acres would be required if the limit were set at the background water quality of 0.17 ppm nitrate nitrogen.

Phosphorus movement to the groundwater will depend mostly on the capacity of the soils between the bottom of the absorption trench and the water table to adsorb and retain the applied phosphorus. The available data indicate that for the permeable soils in hydrological groups A and B, breakthrough is expected to occur as soon as one year after the leach field is put into use. If the phosphorus retention capacity is greater than existing data suggest, breakthrough would be delayed for 5 to 17 years. Once breakthrough occurs, the added phosphorus will be transported unattenuated to the groundwater. It is therefore recommended that data be gathered on the phosphorus adsorption capacity of the subsurface soil horizons of soils likely to be used for septic leach field installations. It is also recommended that, if needed, equipment should be installed to add the chemicals necessary to perpetuate phosphorus.

TABLE 8.1 Undeveloped buffer areas required to limit nitrate concentrations in the leachate to selected levels.

Nitrogen source	Buffer area (acres) 10 ppm NO_3-N in groundwater	2 ppm NO_3-N in groundwater	0.17 ppm NO_3-N in groundwater
Home septic effluent			
-per person	0.13	0.86	3.0
-per 3.5 person household	0.47	3.0	10.5
Municipal secondary effluent			
-per person	0.05	0.45	5.0
-per 3.5 person household	0.18	1.6	17.5
Home lawns (inorganic N fertilizer)*			
-per 1/4 acre lot	0.2	1.9	24
-per 5 acre lot	1.5	13.5	175
Agriculture**			
-per acre of corn	0	0	7.2

* The use of organic N sources would reduce acreages by a factor of 0.63.

** Intensive vegetable crops could require larger set-asides.

Bacteria, viruses and selected pathogens are known to survive in acid soils particularly during periods of low temperature. When the water table is too close to the septic leach fields, these organisms can move into the groundwater. Five feet of soil has generally been shown to be effective in removing pathogenic organisms from contaminated effluent water. The bottom of an absorption trench is typically 2 ft below the surface, and it is therefore recommended that no septic leach field should be allowed in soils which have seasonal high water tables closer than 7 ft from the surface. Soil classification systems typically report only that the depth to water table is greater than 6 ft, and in most instances this would indicate acceptable conditions.

Viruses are known to move rapidly in soils with low cation exchange capacities and high permeabilities. Available data indicate that the subsurface horizons in soils in hydrological group A have permeabilities that would allow rapid movement of viruses. Even if water were slowed by the organic mat which is commonly found at the trench-soil interface, rapidly moving rainwater leachate would be expected to wash virus from the outside of the mat into the groundwater below the soils. All of the soils in hydrological groups C and D found in the Pinelands have water tables at depths of less than 6 ft. Therefore, the persistence and mobility of virus and microbes limits the usable soils only to those in hydrological group B with water tables greater than six feet from the surface. These include: Aura, Collington, Colts Neck, Downer, Freehold, Phalanx, Sassafras, Westphalia, and Woodmansie. Actual suitability should, however, be determined for each site to assure that the criterion considered above are met.

Irrigation with secondary treated sewage effluent could be utilized as a method of disposal. If properly designed, disposal could be achieved without elevating the pollutant concentrations entering the groundwater above predetermined limits. The concentration of nitrogen will limit the monthly application rates, and the maximum acreage would be required during winter months. The model utilized here indicates that treated effluent irrigation would require about 0.05 acre per capita to prevent the groundwater recharge from reaching 10 ppm, 0.45 acres per capita would be required to keep the recharge water at or below 2 ppm N, and 5.0 acres per capita would be required to prevent recharge from exceeding 0.17 ppm nitrate nitrogen. Where necessary, centralized sewage treatment and irrigation disposal may prove to be a usable alternative. Data on phosphorus adsorption will be needed however to fully assess the feasibility of irrigation disposal.

Significant amounts of inorganic nitrogen fertilizer sources applied to lawns may leach to the groundwater. The use of organic nitrogen sources decreases, but does not eliminate the potential for nutrient leaching. The impact of lawn fertilization could be offset by setting aside buffer zones to allow the infiltration of uncontaminated rain water to provide dilution. For 1/4 acre lots, it is estimated that an additional 0.2 acres would be needed to prevent the leach water from exceeding 10 ppm nitrate nitrogen if inorganic nitrogen fertilizer were used. To prevent leachate water from exceeding 2 ppm, 1.9 additional acres would be required, while 24 acres would be necessary to limit leachate NO_3-N to 0.17 ppm. For large lots with large lawns, greater acreages are required, such that a typical five acre lot with a 1 1/2 acre lawn would require an additional 13.5 acres to keep leachate concentrations below 2 ppm nitrogen, and 175 acres to keep it below 0.17 ppm. If buffer areas are not provided, lawn fertilization will have to be discouraged, and the use of native vegetation would be one feasible option.

While only limited acreages are utilized for upland agriculture, they can contribute to the nitrate reaching the aquifer. A model was utilized to calculate the expected nitrogen leaching from spring and fall nitrogen applications to corn. If recommended fertilizer additions are made in the spring, followed by split application during the growing season, only limited amounts of N will be leached. For soils in hydrological group A only 1.0 acres of land would be required to dilute the leachate to 2 ppm nitrogen for each acre of corn, while 22 additional acres would be required to keep the leachate below 0.17 ppm NO_3-N. Much larger losses are encountered from single Fall fertilizer applications, and groundwater protection (2 ppm N) would require 5.4 and 4.5 uncropped acres per acre of cropland for soils in hydrological groups A and B respectively. Fertilization of some crops, including soybeans, rye, hay, cranberries and blueberries, is expected to contribute very little nitrogen to the groundwater because of the small amounts of fertilizer used. Fall applications of nitrogen fertilizer should be discouraged.

8.1 Alternate Technologies and Management Strategies

Decisions on alternative technologies and management within the Pinelands should focus upon reducing the quantities of contaminants applied to the soil and upon providing adequate undeveloped buffer zones for dilution of substances reaching the groundwater. Technologies which reduce nitrogen concentrations in sewage, such as waterless toilets, could be utilized to decrease the lot sizes required for groundwater protection. However where conventional systems are permitted, lot size must be big enough or buffer areas designated to allow for dilution. Reduction of phosphorus loading could be accomplished by reducing or eliminating the use of phosphorus detergents. The resulting lower effluent phosphorus concentrations would lengthen the time before the adsorption capacity of the soil is exceeded. The use of alum could also effectively eliminate the phosphorus problem. Such methods would not, however, allow the relaxation of groundwater depth criterion unless methods were employed to eliminate virus from sewage effluent. However, no practical methods of achieving pathogen removal are known. Minimization of home lawn sizes and the use of nursery-grown native plant species would decrease the need for fertilizers and biocides, but buffer zones are required where lawns exist or are permitted.

8.2 Data Gaps

Studies should be initiated to determine the aquifer recharge rates under soils belonging to the different hydrological groups. First priority belongs to those soils in hydrological groups A and B. At present only the data of Rhodehamel (1970) is available and he cites only one number which must be used on soils in all hydrological groups.

Calculations by Frere (1976) indicate that for this area there are significant differences in recharge volumes between soil hydrological groups and these should be evaluated, since they greatly influence lot size and compensatory acreage calculations.

Since the use of soils in hydrological group A is excluded due to the rapid flux of water, which is likely to carry viruses, investigations should be undertaken to verify or disprove the validity of this criteria.

No data exists on the quality of runoff water from cranberry bogs and blueberry fields. The nutrient concentrations of these two streams should be determined.

Data on the phosphorus adsorption capacity of the soil material between 2 ft below the surface and the top of the aquifer is sorely needed. A sampling program involving coring of the representative soils in hydrological group B should be undertaken and the adsorption capacities of each strata should be evaluated in the laboratory. The potential of groundwater pollution depends greatly on the phosphorus adsorption capacity. Phosphorus adsorption data would also be useful for detailed design of secondary treated effluent irrigation disposal. Similarly, samples should be collected from the water bearing strata so that the mobility of P in the groundwater can be determined.

The complexity of the organic and inorganic reactions likely to occur in the soil surrounding a septic leach field make prediction of the impact of septic disposal on soil pH pure speculation. Field data on active systems should be collected to assess the possible pH effects.

9.0 A REVIEW OF THE "SOIL FACTORS WHICH LIMIT USE FOR SEPTIC TANK ADSORPTION FIELDS" - PREPARED BY THE SOUTH JERSEY RC&D

Soil factors which limit septic field use were broken down into six groups based on water table depth and permeability class. These are as follows:

Group 1. Flooding frequency (occasional to frequent)
2. Seasonal high water table (less than 4 feet) and soil drainage classes (somewhat poorly, poorly and very poorly drained)
3. Permeability (greater than 6 inches per hour) and soil drainage classes (excessively and somewhat excessively drained)
4. Permeability (greater than 2 but less than 6 inches per hour)
5. Variable and undeterminable (site investigation required)
6. Permeability (greater than 0.2 but less than 2 inches per hour).

Our review of the literature indicates that the mobility of pathogenic organisms would require at least 6 ft of soil between the trench bottom and the water table to prevent contamination. Septic leach fields are typically installed such that the bottom of the gravel trench is 2 to 2 1/2 ft below the surface, and thus a water table at 4 ft would provide only 1 1/2 or 2 ft of soil. Even if a water table is at 6 ft, only between 3 1/3 and 4 ft of soil would be available. Thus water tables should be at least 8 ft or even greater if the absorption trenches are placed at standard depths in the soil. Therefore, soils in groups 1 and 2 above are not acceptable for septic leach field disposal. For the New Jersey Pinelands, these soils include all members of hydrological group C and D and would also be excluded by both the U.S. Public Health Service and the N.J. Department of Environmental Protection as being "not acceptable."

Soils in group 3, with permeabilities greater than 6 inches per hour, are in hydrological group A, and present special concern because of the rapidity of internal drainage. As a result, the septic fields in such soils would, if present design criterion are followed, have very small trench bottom areas and may result in rapid downward movement of water and virus. While an organic mat will form to slow the movement of leachate, heavy rains will likely wash virus from the outside of the mat into the groundwater below these soils. Phosphorus would also move rapidly to the groundwater once the adsorption mechanisms in these soils are saturated. Thus the possibility of more rapid movement of pathogens and phosphorus to the aquifer would prohibit the use of soils with high infiltration rates.

Soils in group 4 with permeabilities between 2 and 6 inches per hour would also be expected to allow significant movement of virus and eventual breakthrough of phosphorus. These soils would be the well drained members of soil hydrological group B. Since trenches would be designed to have larger bottom areas than soils in group 3, it would take longer for the phosphorus to break through. The concentrations of both phosphorus and nitrate reaching the groundwater would be more dilute since they would be spread over a larger area than that of trenches places in more permeable soils. The anticipated low phosphorus adsorption capacities of group 4 soils could allow breakthrough in less than a year. These soils should be used only if the subsurface is less permeable than 2 inches per hour in order to minimize virus movement and if steps are taken to remove or precipitate the phosphorus.

The soils in group 5 will need to be investigated on an individual basis, since they have been previously disturbed and predictions of their properties can not be made.

Soils in group 6 with permeabilities between 0.2 and 2 inches per hour would be moderately drained members of hydrological group B. Water is not expected to move fast enough to flush virus to the groundwater, and the phosphorus adsorption capacity should be sufficient to prevent migration to the groundwater for at least several years. Group 6 soils are the best available for septic fields, but the soils may also require steps to precipitate phosphorus to provide long term groundwater protection.

Nitrogen removal will differ only slightly between soils in groups 4 and 6. If one is concerned only about the average concentration of nutrients in the groundwater after several years of use, differences between systems in soils which fall in groups 4 and 6 would be small. Where concentrations in plumes reaching wells and surface waters are of importance, the soils in group 6 would be more desirable than those in group 4.

In summary, soils in groups 1, 2, and 3 should not be used for standard septic leach fields under any circumstances, while the use of soils in group 4 is questionable. Soils in group 6 are the most suitable for septic systems, but special management may be necessary in some cases even in these soils.

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