Impact of On-site Sewage Disposal Systems on Surface and Ground Water Quality

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EXECUTIVE SUMMARY

On-site disposal of septic tank effluent is the most common means of domestic waste treatment in rural and unincorporated areas without sewer systems. Over 1.3 million families in Florida are served by on-site sewage disposal systems (OSDSs). These families introduce nearly 170 million gallons (643 million liters) per day into the subsurface environment, making it potentially one of the largest sources of artificial ground water recharge in the state.

Properly sited, designed, constructed, and operated OSDSs offer an efficient and economical alternative to public sewer systems, particularly in rural and sparsely developed suburban areas. Increasing public awareness and concern for environmental quality and public health require effective treatment and disposal of domestic wastes for all homes in unsewered areas.

Septic tank effluent contains varied concentrations of nitrogen, phosphorus, chloride, sulfate, sodium, toxic organics, detergent surfactants, and pathogenic bacteria and viruses. Widespread use of conventional[1] OSDSs can result in contamination of ground and surface water if the soil does not treat or purify the effluent effectively before it encounters ground water.

Transformation, retention, loss, or movement of nitrogen in natural soil systems is governed by the mechanisms of mineralization, nitrification, denitrification, adsorption, biological uptake, and

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[1]A conventional OSDS is defined as a septic tank and soil absorption trench or bed installed below the natural soil surface.

volatilization. Ground water monitoring studies and laboratory column studies indicate that approximately 20 to 40% of the nitrogen in effluent may be adsorbed or otherwise removed before the effluent reaches ground water.

In a properly sited, designed, constructed, and operated conventional OSDS, nitrification is the predominant mechanism in aerobic, water-unsaturated soil beneath the biological clogging mat or crust, and results in conversion of ammonium and organic nitrogen to nitrate-nitrogen. The soil cation exchange capacity is ineffectual in sorbing nitrate, a soluble anion, and the nitrate moves nearly uninhibited to ground water.

Numerous ground water monitoring studies have detected nitrate-nitrogen concentrations exceeding 10 mg/liter at considerable distance from absorption systems. Attenuation of nitrate by dilution is the only mechanism which significantly lowers nitrate-nitrogen concentration in ground water below conventional OSDSs in aerobic, water-unsaturated soils. Denitrification within a properly sited, designed, and operated conventional OSDS is unlikely, but under conditions of high water tables or slowly permeable soils, nitrate may be denitrified if a biologically useful source of organic carbon is readily available.

Water quality surveys throughout the United States have identified local and regional contamination of ground water and surface water by nitrate derived from OSDSs. Restricting OSDS density (the number of OSDSs per unit land area) lowers the nitrate input from OSDSs to ground water per unit land area and may effectively control levels of nitrate in ground water.

Phosphorus is retained or immobilized in natural soil systems by the mechanisms of adsorption, chemisorption, precipitation, and biological uptake. Ground water monitoring studies and laboratory column studies indicate that very limited phosphorus transport to ground water occurs in aerobic, water-unsaturated soils, and reductions in total phosphorus content of effluent in soil range from 85 to 95% or more. Phosphorus transport to ground water is likely to occur, however, in coarse-textured, non-calcareous, sandy soils that are low in organic matter or in shallow soils over fractured or solution-riddled bedrock.

Phosphorus derived from OSDSs has been detected above background levels in ground water adjacent to OSDSs under conditions of saturated flow due to high water tables or high hydraulic loading rates in numerous studies. However, phosphorus concentration in ground water is found to decrease with distance from OSDSs because phosphorus is capable of undergoing sorption and precipitation within ground water. Very low concentrations of phosphorus in ground water may be sufficient to cause contamination of surface water. Documented cases of contamination of surface water by OSDS-derived phosphorus have been reported where OSDSs are located within close proximity [i.e., less than 100 to 150 ft (31 to 46 m)] to surface waters, or where drainage tile or drainage ditches intercept ground water before phosphorus sorption, precipitation or uptake is complete.

Natural soil systems provide relatively ineffective retention of chloride and sulfate anions, and limited retention of anionic detergent substances and sodium cations. Septic tank effluent can contain appreciable quantities of these chemical constituents. Chloride anions are highly mobile and are not adsorbed or exchanged in soil, so that reduction of chloride concentration in ground water is by dilution. Soils possess more or less finite cation and anion exchange capacities and therefore have fixed capacities to remove sodium and sulfate ions from effluent. Reduction of sodium and sulfate ions from effluent by soil decreases with time as sorption sites become saturated, resulting in increased levels of these ions in ground water. Aerobic, water-unsaturated soil conditions promote biodegradation and adsorption of anionic detergent substances such as linear alkylate sulfonate (LAS).

Several ground water monitoring studies have reported movement of chlorides, sodium, and sulfate from OSDSs into ground water under aerobic, water-unsaturated flow conditions, and movement of LAS under water-saturated flow conditions. Several water quality surveys have detected elevated chloride, sodium, sulfate, and LAS concentrations in ground water associated with OSDSs. Because dilution is the primary mechanism for reduction of these constituents, restricting OSDS density may effectively control levels of these constituents in ground water.

Contamination of ground water has also resulted from disposal into OSDSs of household products containing toxic organic substances or heavy metals and from treatment of OSDSs with "septic tank cleaners" that contain toxic compounds.

Indicator organisms are commonly enumerated in effluent and ground water because the task of detecting all types of bacteria is complex and costly. Counts of total coliforms, fecal coliforms, and fecal streptococci are thought to reflect the presence of human pathogens in effluent and ground water.

Fecal bacteria are removed or otherwise inactivated from effluent in soil by the mechanisms of filtration, adsorption, and natural die-off. The biological clogging mat or crust that commonly forms within the first few inches of the soil below an absorption trench or bed has been found to be an effective barrier to bacterial transport. The removal of indicator organisms from effluent is also a function of the soil water/effluent flow regime. Transport of indicator organisms under water-unsaturated flow conditions is generally restricted to about 3.3 ft (1 m). Movement of indicator organisms over much longer distances has been reported under water-saturated flow conditions. Several ground water monitoring studies and water quality surveys have reported contamination of ground water and surface water originating from OSDSs under conditions of saturated flow, high effluent loading rates, and shallow depth to seasonal high water tables or fractured, jointed or solution-riddled bedrock. Bacterial contamination of wells by OSDSs is the second most common reason for well replacement in the southeastern United States.

Viruses occur in effluent in varied concentrations that reflect the combined infection and carrier status of the residents utilizing the OSDS. Viruses are removed or otherwise inactivated in natural soil systems by the mechanisms of adsorption, filtration, precipitation, biological enzyme attack, and natural die-off. Laboratory column studies indicate that virus adsorption in soil generally increases with increasing cation exchange capacity, clay content, specific surface area, and ionic composition of the soil solution. Low soil pH, low soil moisture content, and low effluent loading rates also increase virus adsorption in soil.

Several ground water monitoring studies have reported transport of viruses to ground water from OSDSs under conditions of saturated or near-saturated flow, high water tables, or high effluent loading rates.

Movement of effluent through 36 to 48 inches (90 to 120 cm) of soil under unsaturated flow conditions is commonly deemed essential to prevent contamination of ground water and surface water. Flow of effluent through water-unsaturated soil results in increased travel time for contaminants, better effluent/soil contact for promotion of physical, chemical, and biological processes, and improved effluent treatment by the soil.

A 24 inch (60 cm) separation distance between the bottom of the adsorption system and the seasonal high ground water, as required currently by Chapter 10D-6 of the Florida Administrative Code, may be inadequate to insure proper treatment of septic tank effluent and prevent contamination of ground water. In-situ ground water monitoring studies and laboratory column studies support a minimum depth of 36 inches (90 cm) of water-unsaturated soil between the bottom of the absorption system and the seasonal high water table.

Conditions of water-saturated flow in soil, high hydraulic loading rates, and shallow depth to seasonal high water tables or highly permeable bedrock have provided documented cases where nitrogen, phosphorus, bacteria, and viruses from septic tank effluent have resulted in contamination of ground and surface water supplies. Proper functioning of an OSDS is achieved only if a sufficient volume of aerobic, water-unsaturated soil is available to absorb the volume of effluent and purify it before it reaches ground water.

More than one half of the soils in the United States are unsuited for conventional OSDSs. Florida has a particularly high percentage of soils unsuited to conventional OSDSs due to conditions of periodically high water tables, low relief, and/or shallow depth to bedrock.

Conventional OSDSs can be modified, however, to improve effluent treatment and reduce the potential for ground water contamination. Proper siting, design, construction, and operation of OSDSs is the key to controlling ground water contamination by the various effluent constituents.

Research Needs

Research is needed to evaluate the fate and transport of chemical and biological contaminants in OSDS effluent. Research strategies and activities should achieve the following:

(1) Monitor the fate and transport of nitrates, phosphates, heavy metals, toxic organics, pathogenic bacteria, and viruses under water-unsaturated flow conditions in and around properly sited, installed, and operated conventional and alternative systems in an array of Florida soil, landscape, and climatic conditions.

A limited number of ground-water monitoring studies and water quality surveys have been conducted in Florida to assess the impact of OSDSs on water quality. One of the fundamental questions that needs to be addressed is "What is the current environmental impact of OSDSs in Florida?"

(2) Evaluate the adequacy of 24 inches (60 cm) of water-unsaturated soil above a seasonal high water table for effective removal of chemical and biological contaminants from effluent under Florida's soil and climatic conditions.

In-situ ground water monitoring studies and laboratory column studies support the maintenance of a minimum soil depth of 36 inches (90 cm) between the bottom of the adsorption system and the seasonal high water table. The 24 inch (60 cm) separation distance between the bottom of the adsorption system and seasonal high ground water table may be inadequate to insure proper treatment of septic tank effluent and prevent contamination of ground water.

This research should include intensive monitoring studies of conventional OSDS installations that have been installed under conditions such that the seasonal high water table is 24 inches (60 cm), or nearly so, below the absorption trench or bed bottom. This phase of the proposed research should comprise the most intensive part of the overall monitoring program recommended in item (1).

(3) Determine densities and setback distances for OSDSs which are adequate to maintain a high degree of water quality and insure protection for public health in future years.

On-site sewage disposal system density is the primary mechanism for controlling concentrations of nitrate, chloride, sulfate, and sodium in ground water. Phosphorus, bacteria, and viruses derived from OSDSs have been found to travel considerable distances in soil under water-saturated soil conditions. Subsurface tile drainage systems, and drainage ditches and canals located adjacent to OSDSs intercept effluent, reduce effluent-soil contact, and may result in contamination of surface water.

On-site sewage disposal systems may impact more severely on water quality in some parts of the state than in others due to differences in land-use, soil-water-landscape relationships, OSDS density, and recharge capabilities of aquifers.

Research in these areas should include both water quality surveys and mathematical analysis, including computer modelling, of the local and regional fate and transport of effluent constituents under a variety of soil, landscape, and land use conditions.

(4) Improve on the current understanding of seasonal high ground water elevations and fluctuations for an array of Florida soils and landscapes.

Water table monitoring studies should be carried out with state-wide centralized coordination. Monitoring sites should be selected to represent the most common soil and landscape types. Scientific analysis of water table and rainfall data should be carried out to predict water table response to variations in rainfall pattern, intensity, and amount at different times of the year. Knowledge of the relationships between seasonal high water tables and soil/vegetative indicators should be improved as feasible.

(5) Determine adequate design criteria for alternative OSDSs, with particular emphasis on proper selection and handling of fill materials for mounds and built-up lots.

Uniformity coefficient, grain size, texture, clay content, and lithology of fill material have been found to affect effluent treatment in mound systems and sand filters.

Water-unsaturated soil conditions and improved effluent treatment have been observed for elevated mound systems which distribute effluent by pressure-dosed distribution systems rather than by gravity systems. These phenomena need investigation in Florida. Also, development or refinement of Soil Conservation Service soil potential ratings for alternative systems such as mounds and built-up or filled lots would aid in proper siting and design of OSDSs.

(6) <u>Determine proper levels of use, maintenance, and management of OSDSs in order to maintain optimal performance with respect to sewage treatment</u>.

Lack of periodic maintenance can result in premature failure of OSDSs. Periodic removal of septage from septic tanks is necessary to maintain sufficient detention time of raw wastewater in the septic tank and hence provide effective pretreatment of effluent before discharge to the soil absorption system. Carry-over of wastewater solids to the absorption system can lead to clogging of the distribution system and infiltrative soil surface, and ultimately to system failure. Maintenance of OSDSs among homeowners is quite varied and in many cases is performed only after a "problem" with system functioning arises. Therefore, OSDS use, maintenance, and management and their effects on surface and ground water quality need investigation.

1. ON-SITE SEWAGE DISPOSAL SYSTEMS

An extensive freshwater system of streams, lakes, and ground water provides what has been referred to as "Florida's most important natural resource" (Florida House of Representatives, 1983). Ground water is a particularly important component of the freshwater system in Florida since it is the source of 92% of the state's drinking water supply (Hand and Jackson, 1982).

On-site sewage disposal of septic tank effluent is the most common means of domestic waste treatment in unsewered areas. Statistics indicate that over 1.3 million families in Florida are served by on-site sewage disposal systems (OSDSs). Currently, 27% of Florida's housing units utilize OSDSs (U.S. Dept. of Commerce, 1980). These families daily introduce a total of over 170 million gallons[1] (643 million liters) per day into the subsurface soil environment, making it potentially one of the largest sources of artificial ground water recharge in the state.

Projected increases in Florida's population from about 10 million people currently to nearly 15 million by the end of this century will very likely be accompanied by an increase in the number of on-site sewage disposal systems (Poole, 1984). Increased emphasis on environmental quality will require satisfactory treatment of domestic wastewater effluent for all homes in unsewered areas. Since Florida's ground water

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[1]This calculation based upon a family of 3 generating 44 gallons (166 liters)/capita/day (Clements and Otis, 1980).

is the primary supply of public drinking water, any contamination[1] of this water resource poses a potential threat to public health.

A recent report from the Florida House of Representatives (1983) states that an urgent and critical need exists to study the short-term and long-term effects of on-site wastewater disposal on Florida's ground water. The objectives for this literature review are: (1) to evaluate critically the capacity of on-site sewage disposal systems (OSDSs) for wastewater treatment based on previous research, and (2) to make recommendations for additional research needed to maintain a high degree of water quality and insure adequate protection of public health in future years.

An exhaustive literature search using the University of Florida library facilities, state and national library systems, and information retrieval (computer) systems as well as personal contacts with scientists involved in OSDS research, officials in county, state, and federal environmental regulatory agencies, engineering and consulting firms, and knowledgeable persons in OSDS manufacture and installation have been utilized to acquire the most current information pertaining to on-site treatment of wastewater and its potential to contaminate ground water.

Almost one-third of all homes in the United States dispose of domestic wastes through individual on-site sewage disposal systems. Septic tank-soil absorption systems represent about eighty-five percent of all individual disposal units (Scalf et al., 1977).

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[1]Contamination is generally defined as the degradation of the natural quality of ground water as a result of man's activities to the extent that its usefulness is impaired (Miller & Scalf, 1977).

The conventional system for on-site treatment and disposal of domestic wastes consists of a buried septic tank and a subsurface infiltration trench or bed. Modifications of the conventional septic tank include multi-chambered septic tanks, mechanically aerated septic tanks, disinfection units, nutrient removal systems, and wastewater segregation and recycle systems. Alterations of the conventional absorption bed or trench include: serial distribution units, alternating bed or trench units, dosed or pressurized intermittent application systems, evapotranspiration systems, electro-osmosis systems, mounds, fills, artificially drained systems, or combinations of these systems (Clements and Otis, 1980; Hansel and Machmeier, 1980).

The primary function of the septic tank is to separate solids from the wastewater, provide limited digestion of organic matter under anaerobic conditions, store solids, and allow the clarified effluent to discharge for further treatment and disposal (Scalf et al., 1977).

The partially treated effluent is discharged from the septic tank into the subsurface soil absorption system. The soil, an integral part of the OSDS, does more than absorb and dispose of the effluent. The effluent actually undergoes final treatment as it moves into and through the soil surrounding the disposal system. Successful functioning of the OSDS is achieved only if the surrounding soil absorbs the volume of effluent produced and if additional treatment in the soil then purifies the effluent before it reaches the ground water (Hansel and Machmeier, 1980).

Although the concept and design are relatively simple, the conventional OSDS is a complex physical, chemical, and biological system. Satisfactory treatment of effluent is a function of the characteristics of the wastewater, the design of the system components, construction techniques employed, rate of hydraulic loading, age of the system, periodic maintenance, climate, geology and topography, and morphological, chemical, and physical properties of the soil (Scalf et al., 1977).

The transport of various effluent constituents to ground water is related to the reactions and processes that wastewater constituents undergo in the septic tank and in the soil environment.

2. WASTEWATER CHARACTERIZATION

2.1 Wastewater Flow

The biological and chemical characteristics of wastewater from individual households can have a profound impact on the performance of OSDSs. Water use within homes typically results in intermittent flow patterns of wastewater varying in volume and quality (Siegrist et al., 1976). Effective management of any wastewater flow requires a reasonably accurate knowledge of its characteristics.

Clements and Otis (1980) have summarized data on wastewater flow from several studies and estimate the average daily wastewater flow from a typical residential dwelling to be 44 gallons (166 liters) per capita per day (pcd). The average daily quantities of wastewater can vary considerably from residence to residence, but are typically no greater than 60 gallons (227 liters) pcd and seldom exceed 75 gallons (284 liters) pcd (Table 1).

The intermittent pattern of individual wastewater-generating activities creates large variations in the wastewater flow rate from a residence. The typical daily flow pattern from various wastewater sources is illustrated in Figure 1.

TABLE 1

SUMMARY OF AVERAGE DAILY RESIDENTIAL WASTEWATER FLOWS

Study	No. of residences	Duration of study months	Wastewater Flow: Study average gpcd*	Wastewater Flow: Range of individual residence averages gpcd
Linaweaver et al., 1967	22	-	49	36 - 66
Anderson and Watson, 1967	18	4	44	18 - 69
Watson et al., 1967	3	2 - 12	53	25 - 65
Cohen and Wallman, 1974	8	6	52	37.8 - 101.6
Laak, 1975	5	24	41.4	26.3 - 65.4
Bennett and Linstedt, 1975	5	0.5	44.5	31.8 - 82.5
Siegrist et al., 1976	11	1	42.6	25.4 - 56.9
Otis, 1978	21	12	36	8 - 71
Duffy et al., 1978	16	12	42.3	-
	Weighted Average		44	

*gpcd = gallons per capita per day
Source: Clements and Otis, 1980.

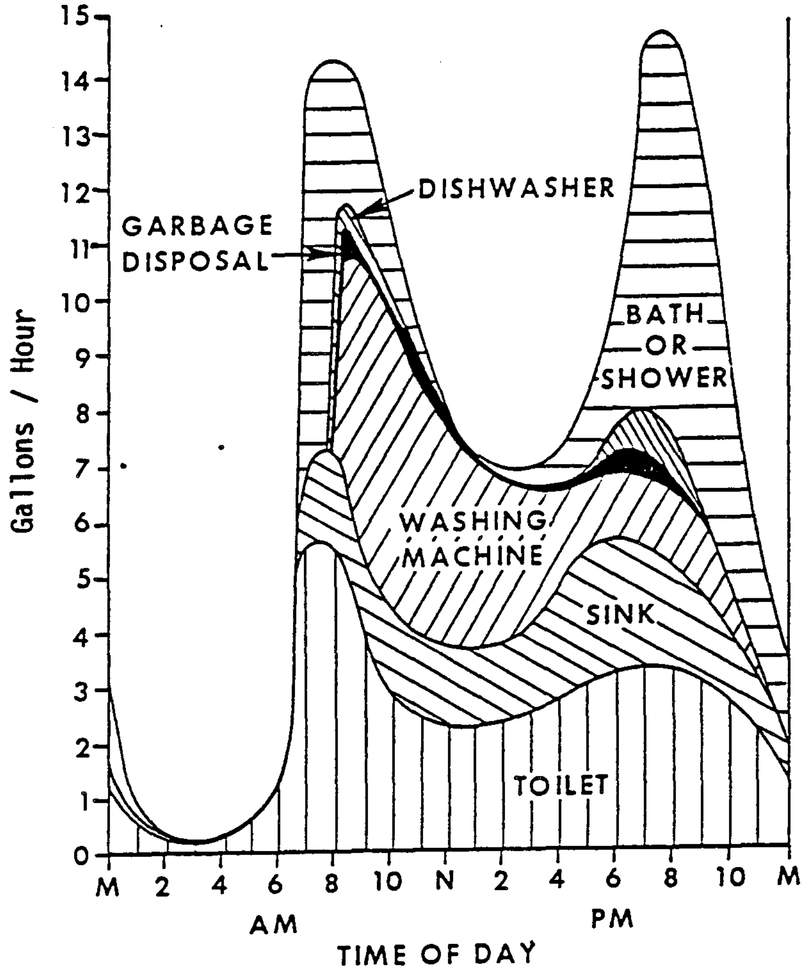

Figure 1. Typical daily household water use.
(Source: Kreissl, 1977)

2.2. Wastewater Quality.

Residential water-using activities contribute varying amounts of constituents to the total wastewater flow. Data from several studies that determined the daily mass loading rate and concentration of various constituents in residential wastewater have been summarized by Clements and Otis (1980) (Table 2).

Since individual water-using activities occur intermittently and contribute varying quantities of constituents, the strength of the wastewater generated from a residence fluctuates with time. Accurate quantification of these fluctuations is extremely difficult (Clements and Otis, 1980). Wastewater derived from these individual activities may be grouped into three major wastewater fractions: (1) garbage disposal wastes, (2) toilet wastes, and (3) sink, basin, and appliance waters. Gray water/black water systems isolate toilet and garbage grinder wastes (black water) from the remaining wastewater (gray water).

TABLE 2

CHARACTERISTICS OF TYPICAL RESIDENTIAL WASTEWATER[a]

Parameter	Mass loading gpcd[c]	Concentration mg/l[d]
Total solids	115 - 170	680 - 1000
Volatile solids	65 - 85	380 - 500
Suspended solids	35 - 50	200 - 290
Volatile suspended solids	25 - 40	150 - 240
BOD_5	35 - 50	200 - 290
Chemical oxygen demand	115 - 125	680 - 730
Total nitrogen	6 - 17	35 - 100
Ammonia	1 - 3	6 - 18
Nitrites and nitrates	<1	<1
Total phosphorus	3 - 5	18 - 29
Phosphate	1 - 4	6 - 24
Total coliforms[b]	-	10^{10} - 10^{12}
Fecal coliforms[b]	-	10^{8} - 10^{10}

[a]For typical residential dwellings equipped with standard water-using fixtures and appliances (excluding garbage disposals) generating approximately 45 gpcd (170 lpcd).

[b]Concentrations presented in organisms per liter.

[c]Grams per capita per day

[d]Milligrams per liter

Source: Clements & Otis, 1980.

3. EFFLUENT CONSTITUENTS

3.1 Wastewater Organics

Septic tank effluent contains appreciable quantities of organic substances. The variety of organic constituents includes proteins, lipids, nucleic acids, polysaccharides, organic acids, phenolic compounds, and detergents.

The concentrations of natural and synthetic organic compounds in the effluent may be collectively expressed in terms of biological oxygen demand (BOD), chemical oxygen demand (COD), and total suspended solids content (TSS). Mean concentrations of BOD, COD and TSS in typical raw wastewater are presented in Table 2.

A properly designed and maintained septic tank removes up to 60% of the BOD and up to 70% of the TSS from raw wastewater (Bouma, 1979). Hansel and Machmeier (1980) reported reductions in BOD concentrations from 270 to 400 mg/liter in raw wastewater and from 140 to 170 mg/liter in septic tank effluent. TSS concentration was reduced from 300 to 400 mg/liter in raw wastewater to 45 to 65 mg/liter in the effluent. Approximately 70 to 80% of grease and oil constituents in raw wastewater are retained in the septic tank (Laak and Crates, 1978). Removal of wastewater organics in the soil is achieved by filtration, decomposition, and incorporation into microbial biomass.

Miller (1973) summarized the metabolic pathways of degradation under aerobic and anaerobic conditions as:

Aerobic

$$(CHO)_nNS \rightarrow CO_2 + H_2O + \text{microbial cells and storage products} + \underset{\substack{\downarrow \\ NO_3^-}}{NH_4^+} + \underset{\substack{\downarrow \\ SO_4^=}}{H_2S} + \text{energy}$$

Anaerobic

$$(CHO)_nNS \rightarrow CO_2 + H_2O + \text{microbial cells and storage products} + \text{organic intermediates} + CH_4 + H_2 + NH_4^+ + H_2S + \text{energy}$$

Miller (1973) and Thomas and Bendixen (1969) estimated that 60 to 75% of the carbon metabolized by aerobic decomposition is evolved as CO_2. Under anaerobic conditions, 20% of the carbon is converted to CO_2, 70% is converted to organic intermediates, and 5% would be converted to CH_3, microbial cells, and storage products. The remainder of the wastewater organics not decomposed or incorporated into microbial biomass are removed from the effluent by filtration in the soil.

Wastewater organics play an important part in the formation of a biologically active clogging layer or crust layer which restricts movement of water and of inorganic and pathogenic constituents (Allison, 1947; Thomas et al., 1966; Bouma et al., 1972; Kropf et al., 1977). Bacteria growing under conditions of excess carbonaceous nutrients store polysaccharides as slime capsules. The slime capsules cover the soil particles at the interface of the disposal trench and the native soil (Tyler et al., 1978), causing a reduction in the effective pore diameter of the soil. This clogging effect increases hydraulic resistance to liquid flow and thus limits the infiltration capacity of the soil.

The formation of a crust layer has been found to be beneficial if it filters additional bacteria or suspended solids from the effluent or helps to maintain unsaturated flow conditions below the crust layer, even where there is saturation above the layer (Clements and Otis, 1980). Formation of the crust layer has also led, however, to premature failure of OSDSs in fine-textured soils due to clogging of soil pores (Bouma et al., 1972; Laak et al., 1974) and to ponding of the effluent in the absorption system.

3.2. Nitrogen

3.2.1. Fate and transport

Forms of nitrogen present in septic tank effluents include ammonia (NH_3), ammonium (NH_4^+), organic nitrogen ($R\text{-}NH_2$), nitrate (NO_3^-) and nitrite (NO_2^-). The mean total nitrogen concentration of septic tank effluent is 55 mg/liter (Otis et al., 1975) and ranges from 40 to 80 mg /liter (Preul, 1966; Walker et al., 1973; Siegrist et al., 1976; Sikora and Corey, 1976).

The types of nitrogen compounds and to some extent the total nitrogen concentration in the effluent are a function of the treatment in the septic tank. Anaerobic conditions prevail in conventional septic tanks, and the resulting effluent contains 75 to 85% soluble ammonium and 25 to 15% organic nitrogen (Bouma et al., 1972; Lance, 1972, 1975; Otis et al., 1975; Kristiansen, 1981a,b). Effluent from aerobic treatment tanks, however, contains primarily nitrate nitrogen (Otis et al., 1975).

Otis et al. (1975) found average concentrations of 38.7, 0.6 and 55.3 mg/liter for NH_4-N, NO_3-N, and total N, respectively, in conventional septic tank effluents and 0.02, 30.1, and 37.6 mg/liter for those respective nitrogen forms in an aerobic treatment unit. Approximately 10% of the total nitrogen in raw wastewater is removed via sludge which accumulates in the bottom of the septic tank (Hardisty, 1973; Laak and Crates, 1978; Laak, 1982).

Several mechanisms in the soil environment are available for transformation, retention, and movement of nitrogen. These mechanisms include: mineralization, nitrification, denitrification, adsorption, biological uptake, fixation, and volatilization.

Nitrification is an aerobic biological reaction performed primarily by obligate autotrophic organisms (Alexander, 1977). Under aerobic conditions, ammonia is transformed to nitrite primarily by the Nitrosomonas group of bacteria, as given by the equation:

$$NH_4^+ + 1\tfrac{1}{2}O_2 \xrightarrow{\text{bacteria}} NO_2^- + 2H^+ + H_2O + \text{energy}$$

Under continued aerobic conditions, nitrite is rapidly transformed by the Nitrobacter group of bacteria, as given by the equation:

$$NO_2^- + \tfrac{1}{2}O_2 \xrightarrow{\text{bacteria}} NO_3^- + \text{energy}$$

Nitrification occurs in the initial one foot (30 cm) of soil below the drainfield (Preul and Schroepfer, 1968; Dudley and Stephenson, 1973; and Magdoff et al., 1974) and proceeds to completion or nearly so when high ground water is absent (Bouma et al., 1972; Viraraghavan and Warnock, 1976a; Bouma, 1979) or when conditions of unsaturated flow exist (Bouma et al., 1972; Bouma, 1979).

Nitrate is very soluble and chemically inactive under aerobic conditions. It travels through the soil-water environment unimpeded. Unless conditions for anaerobic denitrification exist, nitrate will not undergo further transformations once in the ground water (Preul and Schroepfer, 1968; Bouma, 1975a; Hall, 1975). Most studies report attenuation of nitrate concentration by dilution only. The concentration of nitrate in ground water decreases as the nitrate diffuses and is dispersed into surrounding waters of lower nitrate content (Walker et al., 1973; Hook et al., 1978).

Denitrification is another important nitrogen transformation in OSDSs. It is the most desirable means of removing nitrogen (Peavy and

Brawner, 1979) and is the chief mechanism by which the NO_3^- concentration in the effluent can be decreased (Sikora and Corey, 1976; Bouma, 1979).

Denitrification is a biological process performed primarily by ubiquitous facultative heterotrophs (Alexander, 1977). Under anaerobic conditions and in the presence of an adequate energy source, reduction of nitrate to nitrogen gas occurs in two steps. In the first step nitrate is transformed to nitrite, as given by the equation:

$$NO_3^- + 1/3\ CH_3OH \xrightarrow{\text{bacteria}} NO_2^- + 1/3\ CO_2 + 2/3\ H_2O$$

In the second step nitrate is transformed to nitrogen gas, as given by the equation:

$$NO_2^- + 1/2\ CH_3OH \xrightarrow{\text{bacteria}} 1/2\ N\uparrow + \tfrac{1}{2}\ H_2O + OH + 1/2\ CO_2$$

This simplified equation does not include formation of nitrous or nitric oxide gas nor does it include synthesis of bacteria, both of which are also taking place.

In order for denitrification to occur in the soil beneath an OSDS, the nitrogen must be in the nitrate form and an energy or soluble carbon source must be available. Therefore, nitrification, an aerobic reaction, must occur before denitrification. These reactions are rarely present concurrently to any degree in conventional soil-absorption systems (Lance, 1972; Sikora and Corey, 1976; Bouma, 1979). It should be emphasized that the limited occurrence of denitrification reactions is such that the majority of nitrate generated by conventional OSDSs is transported by ground water and is not lost as nitrogen gas (Jones and Lee, 1977a,b).

Winneberger (1971) and Bouma (1975b) have indicated that denitrification may occur to a limited extent in "micro anaerobic zones" such as around decaying roots or in the interior of moist soil peds.

Significant denitrification may also occur when aerated septic tank effluents are applied to soils with restricted drainage or low hydraulic conductivity (Bouma et al., 1972; Bouma, 1975a,b; Otis and Boyle, 1976).

Harkin et al. (1979) indicated that dosed mound systems allow nitrification to occur within the mound fill, denitrification to occur at the natural soil surface, and either nitrification or continued denitrification to occur in the natural soil system, depending upon dosing-rate, moisture content, and texture of the soil. Dosed mound systems do not develop a mature biological clogging layer or crust, and nitrification occurs in the aerobic, sand-textured mound fill above the natural soil. In the natural soil, under anaerobic, water-saturated soil conditions, organic carbon in the surface horizon and in the effluent serves as an energy source for denitrification.

In cases where nitrification does not predominate, adsorption of ammonium (NH_4^+) can be significant (Sikora and Corey, 1976; Laak, 1982). Sikora and Corey (1976) have indicated that the primary factors determining the extent of ammonium adsorption are: the number of soil cation exchange sites exposed to the percolating effluent, the affinity of the sites for NH_4^+, the degree of exchange site saturation with NH_4^+, and the composition of the effluent. The concentration of competing divalent cations (Ca^{++} and Mg^{++}) in the effluent has a significant effect on NH_4^+ adsorption-desorption reactions (Magdoff et al., 1974). Preul and Schroepfer (1968) and Polta (1969) found that NH_4^+ adsorption ranged from 2 mg/100 grams of sandy soil to 100 mg/100 grams of fine-textured soil with a 30% clay content, in a study of Minnesota soils.

Under anaerobic conditions, NH_4^+ can be leached to the ground water. Cation exchange sites in the soil beneath an OSDS will eventually become equilibrated with the cations in the effluent. The effluent would then move to the ground water with its cation composition unchanged (Magdoff et al., 1974; Sikora and Corey, 1976; Brown et al., 1978a,b; Bouma, 1979). Adsorbed NH_4^+ can also undergo desorption, thus becoming subject to nitrification if reaeration occurs during seasonal variation in water table elevations (Laak, 1982).

Ammonia (NH_3) and ammonium (NH_4^+) can also become immobilized by incorporation into microbial or plant biomass. The low carbon/nitrogen ratio of effluent (Witt et al., 1975), however, limits the microbial immobilization of nitrogen in the septic tank and in the ponded effluents of the bed (Lance, 1972, 1975; Sikora and Corey, 1976). The small amount of nitrogen immobilized would become part of the sludge in the septic tank and part of the organic matter in the bed. The latter contributes to crust formation.

Upon the death and decomposition of cells, the nitrogen in microbial tissue is partially released. Research has indicated, however, that nitrogen incorporated into microbial tissue is held in a rather stable organic form that does not break down readily (Bartholomew, 1965; Frederick and Broadbent, 1967).

Immobilization of nitrogen by plants in the immediate vicinity of OSDSs can also occur, as indicated by the characteristic lush growth often observed near OSDSs. Brown and Thomas (1978) found that Bermudagrass (*Cynodon dactylon* L.) removed 8.9 to 45.6% of the nitrogen applied to soil by a septic tank distribution line in a study in Texas.

However, the amount of nitrogen generated by OSDSs greatly exceeds that which can normally be utilized by nearby plants (Sikora & Corey, 1976).

Mechanisms such as fixation and volatilization are thought to remove an insignificant amount of nitrogen from OSDS effluents (Lance, 1972; 1975). Fixation, in contrast to adsorption, produces a relatively stable form of nitrogen which resists nitrification and plant uptake. Fixation occurs in clay minerals by the entrapment of ammonium ions between intermicellular layers. It also occurs as a result of the formation of stable complexes with various organic fractions (Lance, 1972, 1975; Sikora and Corey, 1976).

The equilibrium concentrations of ammonium (NH_4^+) and volatile ammonia (NH_3) are a function of pH (Laak, 1982). Because the pH of OSDS effluent generally falls within the range of 7.5 to 8.0, less than 10% of the nitrogen would be in the gaseous phase. Furthermore, volatilization of significant quantities of ammonia requires considerable air-water contact, which is not provided in a subsurface disposal system (Sikora and Corey, 1976).

Studies by Sabey (1969), Sabey and Johnston (1971), Pilot and Patrick (1972), and Sikora and Corey (1976) have attempted to predict probable nitrogen forms present in the soil at various soil moisture tensions for an array of soil textural classes. The predicted nitrogen end products beneath OSDSs in soils of different textural classes are: nitrate in sands, sandy loams, loamy sands, and loams; a mixture of nitrate and ammonium in silt loams and some silty clay loams, with a possibility of decreasing total nitrogen via denitrification; and ammonium in some clay loams and clays.

3.2.2. Water quality surveys

One approach that has been utilized to assess the impact of OSDSs on water quality is to survey water quality of private wells in areas served by OSDSs. The results are then compared with background water quality standards from wells free of OSDS influence. The effect of OSDSs on surface water quality has also been examined by survey methods.

On-site sewage disposal systems have been identified as a primary source of increasing nitrate, chloride, sulfate, and detergent methylene blue active substance (MBAS) levels in ground water on Long Island, New York (Flynn et al., 1958; Perlmutter et al., 1964; Nassau-Suffolk Research Task Group, 1969; Smith and Baier, 1969; Perlmutter and Guerrera, 1970; Perlmutter and Koch, 1971, 1972; Kimmel, 1972; Smith and Myott, 1975; Garber and Sulam, 1976; Ku and Sulam, 1976; Ragone et al., 1976; Katz et al., 1977; Kreitler et al., 1978; Katz et al., 1980; Porter, 1980). Long-term records of ground water quality indicate that the concentrations of various effluent constituents in ground water have increased substantially since 1910. These trends appear to be associated with increased OSDS density.

In recent decades there has been a population shift from urban to suburban areas. This has resulted in encroachment of housing developments on previously cultivated lands. The impact of this population shift on changing sources of nitrate-nitrogen concentration in ground water has been analyzed using nitrogen δN^{15} isotopes. Three nitrogen-isotope (δN^{15}) ranges have been defined for nitrate from different sources. Kreitler (1975) and Kreitler and Jones (1975) found that δN^{15} values for nitrate from unfertilized cultivated fields (i.e., nitrogen from oxidation of organic matter and rainfall) ranged from +2°/00 to

+8°/00. δN^{15} values of nitrate from artificial fertilizers ranged from -8°/00 to +6.2°/00, and nitrate from animal-waste sources (i.e., wastes from humans and livestock) ranged from +10°/00 to +20°/00. Kreitler et al. (1978) reported that differences in δN^{15} with depth in the ground water of Long Island, New York were related to a change in land use. They suggested that lower δN^{15} values in the deeper aquifer, the Magothy aquifer, were associated with recharge from cultivated fields during the agricultural period, while δN^{15} values in the shallow aquifer reflected current inputs from OSDSs. McCulley (1978), however, did not report an isotopic shift in his investigation in the Arroyo Grande Basin in California. He concluded that there was insufficient recharge from OSDSs to cause an isotopic shift from the dominant agricultural source. Spalding et al. (1982) also concluded that recharge to ground water by OSDSs in the Burbank-Wallula, Washington area was insufficient to cause an isotopic shift in δN^{15} levels from the dominant agricultural source.

A water quality survey of Spinnaker Cove, Mashpee, Massachusetts by the Environmental Management Institute (1975) identified shoreline OSDSs as the source of contaminants. Peak ammonium (NH_4-N) levels in the surface water corresponded to ground water plumes originating from OSDS effluent. A water quality survey of 100 private wells in Falmouth, Massachusetts by the Barnstable County Public Health Department reported elevated ammonia levels in 33% of the wells and elevated nitrate levels in 18% of the wells tested. OSDSs were identified as the primary source of well contamination (K-V Associates Inc., 1983).

Nitrate and chloride contamination of the water table aquifer by OSDSs has been reported along the Coastal Plain of Delaware (Miller, 1972, 1975; Robertson, 1977, 1979; Ritter and Churnside, 1984). Miller

(1972, 1975) found that 25 to 40% of the soils in a study area in New Castle County, Delaware had severe limitations for conventional OSDSs due to seasonally high water tables or restrictive permeability. Approximately 69% of the areas investigated by Robertson (1979) and Ritter and Churnside (1984) were located in permeable, excessively drained soils. Miller (1975) noted that density of OSDSs had a major impact on elevated nitrate and chloride levels in localized areas throughout Delaware. On-site sewage disposal systems were identified as the major source of nitrates in thirteen of twenty-three study areas in Sussex and Kent Counties (Ritter and Churnside, 1984). Robertson (1979) reported that the total amounts of nitrates and chlorides from OSDSs were exceeded only by inputs from confined poultry operations in Sussex County.

A survey of sixty-two wells in twelve housing subdivisions served by OSDSs in Raleigh, North Carolina reported high nitrate and chloride concentrations in ten wells and low levels of detergent branched chain alkyl benzene sulfonate (ABS) in nine wells (Chemerys, 1967). On-site sewage disposal systems were implicated as the source of the contamination.

The occurrence of high nitrate levels in surface water and ground water samples in Dade County, Florida led to several studies which identified agricultural activities as the primary source of nitrate contamination and OSDSs as a secondary contributor to contamination (Pitt, 1974a,b,c; Orth, 1976; Pitt et al., 1975; Donahue, 1978; Department of Environmental Management, 1980; Church et al., 1980; Yoder et al., 1981). Pitt (1974a,b,c) and Pitt et al. (1975) collected 324 groundwater samples from forty-two wells at five sites in Dade County. The contribution of OSDS effluents to ground water contamination was indicated by the presence of elevated ammonia, nitrogen, phosphorus, sodium, total coliform, fecal coliform, and fecal streptococcus levels in several wells. Church et al. (1980) examined water quality at twenty-six sampling stations in undeveloped areas, agricultural areas, and residential areas served by OSDSs throughout Dade County. Degradation of water quality by agricultural activities resulted in inputs of nitrate and potassium, whereas residential areas served by OSDSs contributed nitrate and sodium.

An increase in nitrate concentration in the sand and gravel aquifer of Pensacola, Florida was noted by Trapp (1972). Nitrate levels as high as 44 mg/liter in isolated areas, and a trend toward increasing nitrate concentration in the downtown area of the city, were indicated. Possible sources of contamination were identified as OSDSs, industry, and broken sewer lines.

The National Eutrophication Survey Staff (1975a,b; 1976a,b; 1977a,b,c,d,e,f,g,h,i,j,k,l,m,n,o,p,q,r,s,t,u,v,w,x,y,z,aa,ab,ac,ad, ae,af,ag,ah,ai,aj,ak) surveyed water quality of 41 Florida lakes over a one-year period and estimated annual total nitrogen input from point and non-point pollution sources. An assessment of the trophic condition and identification of the nutrient controlling or limiting eutrophication of the lakes was given. Estimated nitrogen input from OSDSs to lakes was related to the number of OSDSs within 100 ft (30.5 m) of the lakes. Per capita total nitrogen input to lakes from OSDSs was estimated to be 9.38 lbs (4.26 kg), based on the assumption that 100% of the nitrogen in effluent would reach the lake. The assumption that 100% of the nitrogen in effluent would reach the lake is a worst-case situation, because research indicates that 20 to 40% of the nitrogen in effluent may be removed before the effluent reaches ground water (DeVries, 1972; Andreoli et al., 1979; Harkin et al., 1979; Peavy and Brawner, 1979; Laak, 1982).

Total nitrogen input attributed to OSDSs was expressed as a percentage of the total nitrogen input from all point and non-point sources. Estimated total nitrogen inputs to the lakes from OSDSs ranged from less than 0.1% to 13.6% of the total nitrogen input. Increasing the number of OSDSs surrounding lakes would result in greater inputs of nitrogen to the lakes from OSDSs. Algal assay results and water quality data indicated that nitrogen was the rate-limiting nutrient, controlling eutrophication in 27 of 41 lakes surveyed. Nitrogen was found to be the rate-limiting nutrient in an additional 13 of the 41 lakes during 2 of 3 sampling dates.

Russell and Axon Inc. (1980) surveyed ground water and surface water quality adjacent to the Loxahatchee River in Jupiter, Florida. A preliminary study (Russell and Axon Inc., 1979) identified the area as having a high potential for ground water contamination based on soil and hydrologic parameters, and on high OSDS density. Twenty five monitoring wells, 15 to 30 ft (4.5 to 9.1 m) in depth, and fourteen private wells, 25 to 100 ft (7.6 to 30.5 m) in depth, were sampled throughout the study area. The location of monitoring wells with respect to distance from OSDSs was not indicated. Seven surface water samples from ditches and/or ponds were also analyzed.

Mean total (Kjeldahl) nitrogen, ammonium, and nitrate concentrations were 4.30, 1.01, and 0.31 mg/liter, respectively, for the twenty five monitoring wells, and 0.71, 0.30, and 0.23 mg/liter, respectively, for the fourteen private wells. Total (Kjeldahl) nitrogen is the sum total of organic nitrogen and ammonium. The low levels of nitrate in ground water throughout the study area were attributed to saturated anaerobic soil conditions which inhibited nitrification. The elevated total (Kjeldahl) nitrogen and ammonium levels in the twenty-five monitoring wells represented the input of nitrogen by OSDSs to the shallow ground water. Mean sodium concentration in the shallow monitoring wells was 26 mg/liter and ranged from 5.3 to 77 mg/liter. Orthophosphate concentration ranged from 0.05 to 0.64 mg P/liter, with a mean of 0.13 mg P/liter. Mean fecal coliform and fecal streptococcus counts were 1600 and 1200 MPN/100 ml, respectively. Fecal coliform/fecal streptococcus ratios (Geldreich et al., 1968) identified contamination originating from human origin in 53% of the samples. Based on the nutrient and bacteriological status of the monitoring wells, Russell and Axon Inc. (1980) identified OSDSs as the principal source of contamination of the shallow aquifer.

Brooks and Cech (1979) reported contamination of rural water supplies in Houston County, Texas as a result of inadequate distance between OSDSs and private water wells. Wells with the highest observed concentration of nitrates were those located in close proximity to OSDSs. Nearly all dug wells with depths of 50 ft (15 m) or less were found to contain fecal coliforms or fecal streptococci.

Piskin (1973) reported that OSDSs and feedlot runoff were the major sources of elevated nitrate and chloride in ground water sampled in localized rural and urban areas of Hall County, Nebraska. The average nitrate concentration from 511 wells was 14 mg/liter. The highest concentration of nitrate occurred at or near the surface of the water table, and decreased with depth.

A survey by the Wisconsin Department of Natural Resources revealed that 80% of the OSDSs in Westboro, Wisconsin were malfunctioning due to improper siting, construction, operation, or maintenance. Many of the systems were found to be connected by drains which discharged directly into surface waters. Subsequent monitoring of thirty wells indicated bacterial contamination in seven wells and consistently high nitrate-nitrogen concentration (above 10 mg/liter) in one. Although the source of contamination had not been traced directly to improperly functioning OSDSs, they were identified as the probable source (Otis and Stewart, 1976).

Kerfoot (1978), Kerfoot and Brainard (1978), and Kerfoot and Skinner (1981) identified shoreline OSDSs as the source of nitrogen and phosphorus contamination of Crystal Lake, Benzie County, Michigan. Dye tracer studies indicated that more than ninety ground water plumes originating from OSDSs were entering the lake. On the basis of observed total nitrogen and total phosphorus concentrations in the ground water plumes, Kerfoot and Skinner (1981) estimated that effluent wastewater transmitted 16.0% and 0.7% of its total nitrogen and phosphorus loads, respectively, to the lake.

Schaff (1978), De Walle et al. (1980), and De Walle and Schaff (1980) reported that ground water and surface water quality has gradually deteriorated in central Pierce County, Washington, apparently due to increasing urbanization utilizing OSDSs. They indicated that 41% of the population was dependent on OSDSs for wastewater treatment. Analysis of 386 ground water samples revealed significant increases in nitrate, chloride, and sodium concentrations above background levels. Bacterial contamination was noted for several wells which obtained their water from the shallow water table aquifer.

Quan et al. (1974) reported that OSDSs were the primary cause of ground water degradation by nitrate and sulfate in East Portland, Oregon. Areas served by OSDSs had ground water nitrate-nitrogen levels of 3.7 to 10.9 mg/liter above background levels obtained from areas served by sewage treatment facilities.

Kaufman (1976, 1977) reported a decline in the quality of near-surface ground water in Las Vegas Valley, Nevada. OSDS effluent was identified as a principal source of ground water recharge and as a principal source of nitrates, chlorides, and sulfates.

The California Department of Water Resources (1962) and Stout et al. (1965) studied nitrates in ground water of the Grover City - Arroyo Grande area in San Luis Obispo County, California. Fertilizer, sewage effluent, and OSDSs were identified as primary sources of nitrate contamination. The highest nitrate concentrations appeared in ground water downgradient from the unsewered areas of Grover City. These areas were served by OSDSs.

Schmidt (1972, 1977) reported nitrate and chloride contamination of ground water by OSDSs in the east-central part of the San Joaquin Valley in California. Nitrate concentration was found to be stratified in the aquifer beneath unsewered metropolitan areas of Fresno and Clovis. Nitrate and chloride concentrations in the ground water were related to OSDS density, longevity of installation, and soil and hydrologic characteristics.

Wilson et al. (1979) surveyed surface and ground water quality at twenty monitoring sites in Stinson Beach, California. The surface failure rate for OSDSs was 10%. This resulted primarily from clogged systems and improper design and maintenance. Fluorescein dye studies indicated that OSDS effluents were entering the ground water. However, nitrate and ammonium concentrations in several ground water monitoring wells were only slightly above background levels. Bacteriological assessment of ground water was hampered by coliform contamination of background water quality wells. Monitoring wells installed by Wilson et al. (1979) were constructed to intercept only the top surface of the ground water. Childs et al. (1974) and Rea and Upchurch (1980) have indicated, however, that contaminants may move through the ground water in complex, bifurcated plumes, so a three-dimensional array of multi-leveled wells may be necessary to detect the plumes.

Kreitler and Browning (1983) conducted a water quality survey near Georgetown, Grand Caymen Island, West Indies. The dominant land use was small communities with isolated houses served by OSDSs. No extensive cultivation or feedlot systems existed. The water table was within 9.8 ft (3 m) of the land surface and the depth of wells was less than 33 ft (10 m). Nitrate concentrations in the ground water ranged from 11.9 to 29 mg/liter. Nitrogen isotope (δN^{15}) analyses (Kreitler et al., 1978) identified the nitrates as originating from animal wastes (i.e., septic tank and cesspool effluent).

3.2.3. Ground water monitoring studies

On-site sewage disposal systems are but one potential source of contamination to surface water and ground water, which makes it difficult to assess accurately the affects of OSDSs on ground water using a water quality survey. The ground water monitoring approach is potentially a more accurate method of evaluating the contribution of OSDS effluent to ground water contamination.

The ideal ground water monitoring approach involves a three dimensional array of ground water monitoring wells surrounding an OSDS. Childs et al. (1974) and Rea and Upchurch (1980) suggested that monitoring wells should be of variable depth to insure detection of both shallow and deep contaminant plumes. An additional approach involves the use of piezometers or suction lysimeters placed in the soil surrounding an OSDS, to detect the effluent's contribution to soil water.

Several studies have employed the ground water monitoring approach. Preul (1966), for example, monitored nitrate-nitrogen concentrations in ground water adjacent to six OSDSs. Average nitrate-nitrogen concentration exceeded 10 mg/liter at a distance of 40 ft (12.2 m) from the drainfield in three of the systems. A nitrate-nitrogen concentration of 10 mg/liter was also detected in ground water sampled 90 ft (27.4 m) from the drainfield at one of the three systems. The background nitrate-nitrogen concentration of the ground water was 0.1 mg/liter.

A significant concentration of nitrate was detected in shallow ground water at a distance of 140 ft (43 m) downgradient from a seepage pit in Minnesota (Polta, 1969). At a distance of 70 ft (21 m) downgradient from the seepage pit, nitrate-nitrogen concentration was approximately 10 mg/liter. Site conditions were not identified.

Polkowski and Boyle (1970) monitored nitrate-nitrogen levels of ground water receiving OSDS effluents from a school in Wisconsin. Nitrate-nitrogen concentrations as high as 21 mg/liter and 10 mg/liter were recorded at distances of 15 ft (4.6 m) and 100 ft (30 m), respectively, from the system. Nitrate-nitrogen levels in the ground water decreased to 2.4 mg/liter at a distance of 265 ft (81 m). They concluded that dilution was the primary mechanism responsible for decreasing nitrate-nitrogen levels.

Lee (1972, 1976) monitored movement of nutrient-enriched ground water into Lake Sallie, Minnesota using seepage meters installed in the lake bottom. The highest nitrate concentrations in seepage inflow were found along the shore where OSDSs were used. The average nitrate-nitrogen concentration in ground water from shallow monitoring wells ranged from 12.4 to 59.1 mg/liter. The seepage inflow 23 ft (7m) from shore contained nitrate-nitrogen concentrations that ranged from 8.8 to 16 mg/liter, while ground water discharged 39 ft (12 m) and 56 ft (17 m) from shore contained less than 0.005 mg/liter. Ammonia-nitrogen concentrations in the seepage inflow were initially high but decreased to 0.25 mg/liter over the two-month monitoring period. One seepage meter which appeared to be strongly affected by an OSDS consistently contained over 2 mg/liter of ammonia-nitrogen.

Lee (1972) estimated that 40% of the total nitrogen load leaving an OSDS near the lake eventually reached the lake via shallow ground water flow. Dudley and Stephenson (1973) found significant contamination of ground water at eleven OSDS sites in Wisconsin. Nitrification was incomplete in two systems located in dense glacial till but in all of the other cases, nitrification was rapid. Under conditions of seasonally high water tables and slow permeability, ammonium-nitrogen concentrations greater than 10 mg/liter were observed below absorption trenches. These high ammonium concentrations were attenuated downgradient by cation exchange. Under aerobic conditions eight of the eleven sites had nitrate-nitrogen levels greater than 5 mg/liter within 10 ft (3 m) of the system. At five of these sites, nitrate-nitrogen levels were greater than 10 mg/liter. Beyond 25 ft (7.6 m), average nitrate-nitrogen levels in ground water did not exceed 5 mg/liter for eight of the OSDS sites.

Nitrate-nitrogen levels as high as 30 mg/liter at a depth of 12 ft (3.7 m) below an OSDS were reported by Bouma et al. (1972). An additional OSDS site had ground water nitrate-nitrogen concentrations of 15 mg/liter at a depth of 30 ft (9.1 m). Monitoring wells adjacent to an OSDS installed in a poorly drained site yielded ammonium-nitrogen concentrations as high as 75 mg/liter. The ammonium concentration decreased rapidly with distance from the system.

Walker et al. (1973a,b) monitored ammonium and nitrate concentrations in shallow ground water wells adjacent to five OSDSs in Wisconsin. Ammonium was the predominant nitrogen species in ground water adjacent to an OSDS installed at a poorly drained site. Ammonium-nitrogen concentrations as high as 70 mg/liter in ground water adjacent to the drainfield were reduced to 5 mg/liter at a distance of 15 ft (4.6 m) from the drainfield. The remaining systems were located in well-drained and moderately well-drained soils where nitrification predominated. Nitrate-nitrogen levels in the ground water adjacent to the drainfield were as high as 40 mg/liter in the upper one foot (30 cm) of the water table, but decreased to approximately 10 mg/liter at a distance 230 ft (70 m) downgradient, due primarily to dilution. Nitrate removal by denitrification was considered highly unlikely, and significant local ground water contamination was anticipated. Walker et al. (1973b) estimated that an OSDS serving a family of four would contribute approximately 73 lbs (33 kg) of nitrogen per year to the ground water.

Ellis and Childs (1973) and Childs et al. (1974) identified three of nineteen monitored OSDSs as contributors to ground water contamination around Lake Houghton, Roscommon County, Michigan. Monitoring wells 15 to 30 ft (4.6 to 9.1 m) from the OSDSs detected nitrate-nitrogen levels as high as 16.5 mg/liter in the upper 6 ft (1.8 m) of the water table. As the nitrate-enriched plume moved along the hydraulic gradient, it moved deeper into the ground water. The maximum concentration of nitrates at 50, 100, and 330 ft (15, 30, and 100 m) downgradient occurred at depths of 8, 10 to 12, and 14 ft (2.5, 3 to 3.7, 4.3 m) below the water table surface, respectively. Nitrate-nitrogen levels at a distance of 100 ft (30 m) downgradient exceeded 10 mg/liter, while nitrate-nitrogen levels 330 ft (100 m) downgradient were in the 5 to 10 mg/liter range.

Childs et al. (1974) indicated that sampling plans designed to detect and quantify waste migration in ground water should be predicated on the concept that the waste plume may be complex and that the contaminant plume may not always follow regional ground water flow due to perched water tables or subsurface horizons which restrict vertical flow.

Missimer Associates Inc. (1976) monitored ground water quality at four sites on Sanibel Island, Florida. Two of the sites which they investigated were located in subdivisions served by OSDSs, while the remaining two sites were located in undeveloped areas. Ground water adjacent to an OSDS contained three times more total nitrogen than ground water from the most naturally nutrient-enriched area studied. An ammonium-nitrogen concentration of 37 mg/liter was detected in ground water immediately adjacent to the drainfield and decreased with distance downgradient. Levels were only 2.3 mg/liter approximately 10 ft (3 m) from the drainfield. Conversely, a low nitrate-nitrogen concentration of 0.06 mg/liter, detected adjacent to the drainfield, increased to 15.2 mg/liter approximately 3 ft (0.9 m) downgradient. The OSDS was estimated to be the source of 5 to 10% of the nitrogen entering West Rock Lake, approximately 15 ft (4.6 m) away. A second OSDS monitored with only two water table wells, which were upgradient from the OSDS, had total nitrogen, ammonium, and nitrate concentrations below background levels for the two undeveloped areas.

Viraraghavan and Warnock (1976a,b,c) reported on an OSDS ground water monitoring study near Ottawa, Ontario, Canada. A textural analysis of the soil indicated a 24 inch (60 cm) increment of sandy clay overlying clay with less sand. A vertical distance of 34 inches (86 cm) separated the base of the effluent tile line from the seasonally high water table. Ground water samples collected from the water table directly beneath the absorption system yielded mean ammonium-nitrogen concentrations of 20.8 mg/liter, ranging from 0.3 to 75 mg/liter. Mean nitrate-nitrogen concentration in the ground water beneath the system was 0.7 mg/liter, ranging from 0 to 0.32 mg/liter. Movement of ground water laterally through 10 ft (3 m) of soil reduced ammonium levels to 1.0 mg/liter and subsequently increased nitrate levels in the adjoining ground water. Beyond the 10 ft (3 m) distance, reduction in nitrate concentration was attributed to dilution.

A four year monitoring study of an OSDS in Wisconsin by Jones and Lee (1977a,b, 1979) detected very low levels of nitrate in ground water. Monitoring wells established 50 ft (15.2 m) downgradient from the drainfield had nitrate-nitrogen concentrations of less than 2 mg/liter in the ground water. The ammonium-nitrogen concentration in the ground water ranged from less than 0.05 to 0.36 mg/liter. Depth to the water table and soil texture were not indicated. Because monitoring wells were located 50 ft (15.2 m) downgradient from the OSDS, considerable dilution of nitrate may have occurred in the ground water.

Reneau (1977) monitored changes in various nitrogen fractions over a four year period at a site with a fluctuating water table on the Coastal Plain of Virginia. Groundwater above and in a very slowly permeable plinthic horizon (iron-rich hardpan) was analyzed for ammonium, nitrite, and nitrate concentrations. Ammonium-nitrogen in solution above the plinthic horizon decreased with increasing distance from the drainfield, in the direction of ground water flow. The decrease ranged from 23 to 4 mg/liter at distances of 6 inches (15 cm) and 40 ft (12 m), respectively, from the drainfield. Decreased ammonium concentrations were attributed to the processes of adsorption and nitrification. Nitrification above the plinthic horizon was apparently related to distance from the drainfield and fluctuations in the water table, which created alternating aerobic and anaerobic conditions. Nitrite and nitrate that moved into the plinthic horizon apparently did not undergo denitrification and were diluted by ground water. Nitrate-nitrogen concentration of ground water in the plinthic horizon ranged from 0.06 to 2.92 mg/liter.

Gibbs (1977a,b) monitored ground water quality adjacent to an OSDS near Lake Taupo, New Zealand. The soil was described as being formed in pumice of recent volcanic origin. Soil texture was not indicated. The system consisted of a septic tank and soakhole (i.e., seepage pit) located 75 ft (23 m) from the lake. The bottom of the soakhole was 3.3 ft (1 m) above the water table, with ground water samples being collected from the upper 4 inches (10 cm) of the water table. A thermal gradient between the effluent and ground water inhibited mixing of effluent with ground water. It confined the effluent to the upper 8 inches (20 cm) of the water table. Effluent migration patterns were determined by sampling ground water on a 3.3 ft (1 m) grid pattern over the area. Ammonium-nitrogen concentrations of 30 mg/liter in ground water adjacent to the soakhole resulted from direct discharge of effluent through a channel to ground water. A rapid decrease in ammonium-nitrogen concentrations with distance from the soakhole was attributed to nitrification, dilution, adsorption, and biological acitivity. Evidence of nitrification was found at the intersection of the channel and the water table. Conversion of ammonium to nitrate was apparently complete about 3 ft (1 m) from that point.

Nitrate-nitrogen concentrations of 50 mg/liter were detected in ground water approximately 16 ft (5 m) downgradient from the soakhole and decreased to 5 mg/liter approximately 49 ft (15 m) from the soakhole during this sampling program. Background nitrate-nitrogen concentration was 0.75 mg/liter. Using chloride ion concentration as a reference to estimate dilution, Gibbs estimated that 10.4 to 24.2% of the nitrogen entering the ground water was removed by the processes of adsorption, ion exchange, and biological activity. Variation in nitrogen removal (10.9% in summer and 24.2% in fall) was related to seasonal changes in lake level, evapotranspiration, temperature, and plant growth rates. These in turn affected the location of the water table surface in the soil profile. Seasonal variation in nitrogen reduction was thought to reflect the variation in renovating capacity of the soil horizons where the water table surface was located.

Peavy and Groves (1978) and Peavy and Brawner (1979) reported nitrate contamination in shallow monitoring wells surrounding an OSDS in Bozeman, Montana. The soil at the site ranged in texture from silty clay loam at a depth of 6 to 24 inches (15 to 60 cm) to gravelly loam at 24 to 48 inches (60 to 120 cm) and to sand, gravel, and cobblestones below 48 inches (120 cm). The water table was at approximately 4 ft (1.2 m) throughout the year. The depth to the base of the absorption system was not indicated. Mean nitrate concentration in two of the wells exceeded 45 mg/liter, and that of three wells exceeded 30 mg/liter. Based on nitrate input from the OSDS effluent, Peavy and Brawner (1979) calculated an overall increase of 2.4 mg/liter in the nitrate levels of ground water from each OSDS.

Machmeier and Mattson (1978) reported nitrate-nitrogen and total (Kjeldahl) nitrogen determinations from suction lysimeters placed beneath an OSDS. The OSDS had two drainfields, and was located in an Ontonagon clay in Minnesota. One lysimeter was located in the middle of a 6 inch (15 cm) sand layer beneath the drainfield gravel, and the other lysimeter was placed at the bottom of the bed in contact with the soil.

Nitrate-nitrogen concentration in the effluent was less than 2 mg/liter. This was apparently due to anaerobic soil conditions as a result of ponding of effluent in the drainfields. Total (Kjeldahl) nitrogen concentration ranged from 15 to 70 mg/liter in the shallow lysimeters. The total nitrogen concentrations of the deeper lysimeters were about one-half of those found in shallow lysimeters.

Erickson and Bastian (1979) monitored ground water quality adjacent to a large OSDS installed at a rest area by the Michigan Department of Transportation. The system consisted of a large three-compartment septic tank that discharged effluent into four tile drainfields. The site was described as a loam-textured soil with a water table elevation 25 to 30 ft (7.6 to 9.1 m) below the soil surface. Monitoring wells adjacent to the system detected an average nitrate-nitrogen concentration of 25 mg/liter in the groundwater. The nitrate-nitrogen concentration in the wells varied from 3.1 to 66 mg/liter. Erickson and Bastian considered the system to be failing with regard to nitrate removal.

Reneau (1979) examined shallow ground water quality adjacent to an OSDS installed at a somewhat poorly to poorly drained site which was artificially drained by agricultural field tile. Ammonium-nitrogen concentration was highest adjacent to the drainfield (20 to 30 mg/liter) and decreased logarithmically with distance therefrom. Ammonium-nitrogen concentration in wells 15 ft (4.6 m) from the drainfield was less than 1.0 mg/liter. The reduction in ammonium concentration was attributed to adsorption and nitrification. The nitrate-nitrogen concentration increased to a maximum of 2.7 to 3.9 mg/liter at a distance of 15 ft (4.6 m) and then decreased with increasing distance from the drainfield. Nitrification during low water table periods resulted in elevated nitrate concentrations in ground water during periods of rising water tables. Decreases in nitrate concentration beyond 15 ft (4.6 m) were attributed to denitrification.

Andreoli et al. (1979) monitored nitrate-nitrogen concentration in soil water below an OSDS located at the Brookhaven National Laboratory, Long Island, New York. Approximately 36% of the total nitrogen applied to the soil in OSDS effluent was removed after passage through 24 inches (60 cm) of soil. Of the total nitrogen present in the unsaturated soil, 80 to 85% occurred as nitrate in samples collected at 24 inch (60 cm) and 48 inch (120 cm) depths. Nitrate-nitrogen concentrations at these two depths averaged 31.6 and 32.4 mg/liter, respectively.

Wolterink et al. (1979) sampled ground water adjacent to an OSDS studied by Rea and Upchurch (1980) in Florida. Nitrate-nitrogen concentrations in ground water approximately 10 ft (3 m) from a seepage pit ranged from 10.4 to 56 mg/liter. Twenty ft (6 m) from the seepage pit, nitrate-nitrogen concentrations ranged from 8 to 34.2 mg/liter. The depths from which the samples were collected were not indicated.

In an attempt to identify sources of subsurface nitrate pollution using stable nitrogen isotopes, Wolterink et al. (1979) collected a total of fifty-two ground water samples adjacent to OSDSs at eleven sites throughout the United States and the Caribbean (Table 3). Eighty-five percent of the samples exceeded a nitrate-nitrogen concentration of 10 mg/liter.

Rea and Upchurch (1980) studied contamination of ground water by an OSDS in Tampa, Florida. The system consisted of a septic tank and a seepage pit. The soils within the study area were described as Leon fine sand and Blanton fine sand. Leon fine sand has a severe limitation

Table 3. Summary of nitrate-nitrogen concentration in ground water adjacent to OSDSs examined by Wolterink et al. (1979).

Site location	No. of sites	No. of wells	No. of samples	NO_3-N mg/liter	No. of samples exceeding 10 mg/l NO_3-N
Houghton Lake, Michigan	2	4	10	8.1- 71.5	8
Pueblo, Colorado	1	1	6	18.9-106	6
Indian Hills, Colorado	1	1	5	8.3-108	4
Rockdale, Texas	1	2	4	1.8- 20.1	1
Texas A&M (College Station), Texas	1	3	4	5.7- 15	3
Tampa, Florida	1	2	12	8 -102.6	11
Runnels County, Texas	1	2	4	33 -711	4
Macon County, Georgia	1	1	1	260	1
Queens, New York	1	1	2	20 - 30	2
Grande Caymen Island (Caribbean)	1	1	4	11.9-29.0	4
Total	11	18	52		44

for OSDSs due to high water table conditions and periodic flooding. Blanton fine sand has a moderate limitation due to high water table conditions (Soil Survey Staff, 1981). The seepage pit extended 3 ft (1 m) below the watertable. Nitrate-nitrogen concentrations in ground water 13 ft (4 m) below the seepage pit exceeded 50 mg/liter. Nitrate-nitrogen concentrations greater than 30 mg/liter were detected 50 ft (15 m) downgradient and nitrate-nitrogen concentrations greater than 10 mg/liter extended beyond a distance of 80 ft (24.3 m). Nitrate-nitrogen concentration gradients indicated that nitrate from the OSDS was migrating into White Trout Lake, about 111 ft (34 m) away.

Starr and Sawhney (1980) monitored ammonium and nitrate concentrations beneath an OSDS in Connecticut. A septic tank and alternating open-bottomed concrete leaching chambers were installed in coarse sand and monitored over a two-year period. Nitrogen species present in the surrounding soil were strongly influenced by varying moisture conditions over the two-year period. In 1975, a year of greater rainfall, little of the ammonium from septic tank effluent was nitrified. Ammonium-nitrogen concentrations of 25 to 30 mg/liter were detected at depths greater than 3 ft (0.9 m) below the leaching chambers. In contrast, the ammonium was nitrified under lower rainfall conditions in 1976. Nitrate-nitrogen was detected in concentrations of 25 to 30 mg/liter at depths greater than 3 ft (90 cm) below the leaching chambers. Ammonium-nitrogen concentration dropped to 1 to 3 mg/liter. Starr and Sawhney estimated that 20 to 25% of the total nitrogen in septic tank effluent would reach the ground water. On the basis of an OSDS density of 2 systems/acre (5 systems/hectare) and a soil water flux of 24 inches/yr (60 cm/yr), they calculated that nitrate-nitrogen contamination of ground water would amount to 2.5 mg/liter/yr/OSDS. Similar results were reported by Peavy and Brawner (1979).

The movement and treatment of OSDS effluents on the lower Coastal Plain of North Carolina was examined by Carlile et al. (1981) and Cogger and Carlile (1984). The soils at the sites were derived from unconsolidated marine sediments ranging in texture from sand to clay. Fifteen of the seventeen systems examined were at least seasonally saturated by high water tables. The remaining two sites had water tables at least 12 inches (30 cm) below the base of the absorption system. Those systems which were nearly continually saturated had the highest concentration of contaminants in the ground water, and the contaminants moved the farthest in these systems. Contamination was generally confined to within 25 ft (7.6 m) of the systems, but more widespread movement was noted in several continuously-saturated systems. Nitrogen was found primarily as ammonium around the more saturated systems and as nitrate in several of the better aerated systems. Mean nitrogen concentrations of all wells ranged from less than 0.5 to 22 mg/liter for nitrate and less than 0.5 to 27 mg/liter for ammonium adjacent to the systems. The majority of wells had nitrate-nitrogen and ammonium-nitrogen concentrations of less than 4 mg/liter.

Water and Air Research (1981) evaluated surface and ground water quality adjacent to OSDSs near Lakes Cannon, Gordon, and Ned in Polk County, Florida. Various OSDS effluent constituents were detected in ground water monitoring wells, but it was concluded that OSDSs contributed a relatively small portion, 2 to 11%, of the total nitrogen load to the three lakes. Annual nitrate nitrogen loads from OSDSs to Lakes Gordon and Ned were estimated at 40 and 210 lbs/yr (18 and 95.5 kg/yr), respectively. Elevated total nitrogen concentrations were detected in monitoring wells adjacent to OSDSs and generally decreased with increasing distance downgradient. Total nitrogen concentration included the combined concentrations of organic nitrogen, ammonia, nitrite, and nitrate. Nitrate-nitrogen predominated in monitoring wells at Lakes Gordon and Ned, while ammonia and organic nitrogen predominated in monitoring wells at Lake Cannon. Mean nitrate-nitrogen concentrations in ground water adjacent to OSDSs at Lakes Gordon and Ned ranged from 7.7

to 9.1 mg/liter. Mean organic nitrogen plus ammonia-nitrogen concentration adjacent to the OSDS at Lake Cannon was 2.5 mg/liter. The possible influence of OSDSs on background water quality wells, and of previous fertilization during citrus production prior to residential development, were cited as factors which hampered evaluation of OSDS impact on ground water and surface water quality.

A study was conducted by Wilson et al. (1982) in Oregon to determine the feasibility of artificially draining moderately well-drained to somewhat poorly drained soils formed in silty alluvium, in order to permit satisfactory operation of OSDSs. Perimeter tile drains installed at a depth of 4 ft (1.2 m) were set back 10 ft (3 m) from disposal trenches and tile drains installed at a depth of 10 ft (1.8 m) were set back 20 ft (6 m) from disposal trenches. Tile drains 6 ft (1.8 m) deep were effective in lowering the ground water elevation approximately 2.5 ft (0.8 m) below the disposal trenches.

Nitrate-nitrogen levels in tile drainage discharge were all below the minimum drinking water standard of 10 mg/liter as mandated by federal water quality standards (USEPA, 1976). Nitrate-nitrogen concentration in drainage from 6 ft (1.8 m)-deep tiles ranged from 0.6 to 4.1 mg/liter, while nitrate-nitrogen concentrations in the drainage from 4 ft (1.2 m) deep tiles ranged from 0.9 to 4.7 mg/liter. Ammonium concentrations of the tile drainage were not monitored.

Uebler (1984) monitored movement of nitrogen in soil water below absorption trenches over a twelve-month period in North Carolina. The site was located in a Cecil sandy loam, and the soil texture at the base of the trenches was clay. Saprolite (i.e., highly weathered bedrock) located at a depth of 12 inches (30 m) below the adsorption system resulted in a perched water table. Nitrogen concentrations in soil water below adsorption trenches were compared at loading rates of 0.18, 0.28, and 0.37 gallons/ft^2/day (7.5, 11.3, and 15 liters/m^2/day). In addition, nitrogen concentrations in soil water below adsorption trenches treated with cement and lime, used to stabilize the adsorptive capacity of the soil, were compared to an unamended trench. The amendments were added to the surface of the soil trench on a basis of 5% by weight. Soil water samples were collected in porous ceramic cups placed at depths of 6 inches (15 cm) and 12 inches (30 cm) below the trenches.

The ammonium-nitrogen concentrations in the soil water below the trenches at the three loading rates were not significantly different. Concentrations of ammonium-nitrogen below the amended trenches ranged from 6.5 to 7.2 mg/liter, and the concentration of ammonium-nitrogen below the unamended or control trench was 11.8 mg/liter. The high ammonium-nitrogen concentration was related to anaerobic soil conditions. Nitrate-nitrogen concentrations in the soil water below the trenches at the three loading rates were also not significantly different but were significantly affected by amendments. Nitrate levels increased by 57 to 67% for the amended trenches as compared with the control trench. The nitrate-nitrogen concentration in soil water below the amended trenches ranged from 10.4 to 11.0 mg/liter, with a nitrate-nitrogen concentration below the control trench of 6.6 mg/liter. Uebler concluded that reduced ammonium and increased nitrate concentrations below the amended trenches reflected an improved soil aeration status.

3.2.4. Lysimeter, sand filter, and column studies.

Several investigators have monitored effluent outflow from lysimeters and sand filters under laboratory and field conditions. Additional studies have utilized soil columns of varying dimensions to examine the fate and transport of OSDS effluent constituents in soils under various controlled conditions which are difficult or costly to determine in situ.

Brown et al. (1978b, 1984) monitored movement of nitrogen from a drainfield distribution line installed in undisturbed field lysimeters in Texas. During a three-year period, leachate was collected at the bottom of the lysimeters by a series of ceramic cups maintained at a potential equivalent to 0.8 bar. During the first 15 months after effluent application began, the concentration of ammonium-nitrate in the leachate ranged from 0.5 to 1.5 mg /liter. Ammonium moved at a rate of 39 inches (100 cm) per year in a sandy loam, 19.7 inches (50 cm) per year in a sandy clay, and 2 inches (5 cm) per year in a clay.

Nitrate-nitrogen concentrations greater than 10 mg/liter were detected in leachate from the sandy loam shortly after the study began. Nitrate-nitrogen concentrations in leachate from the sandy clay and clay did not exceed 10 mg/liter. Nitrate concentration in the leachate decreased during the subsequent year due to increasingly anaerobic soil conditions in all lysimeters caused by high effluent application rates. Cessation of effluent applications to the lysimeters for a six-week period produced aerobic conditions in the soil and increased nitrification. Nitrate-nitrogen concentrations averaged 21 mg/liter in the leachate from the sandy loam, 36.6 mg/liter in the sandy clay, and 18 mg/liter in the clay. Individual samples of leachate contained as much as 125 mg/liter of nitrate-nitrogen. The nitrate-nitrogen concentration decreased after additional effluent applications, which produced less aerobic soil conditions.

Brandes et al. (1975) determined the nitrogen concentration in leachate collected from ten underdrained filter beds of various filtering media in Ontario, Canada. The dimensions of the filter beds were 12 ft x 10 ft (3.6 x 3.0 m) by 30 inches (76 cm) deep. Filter media in five of the beds consisted of sand of varying size fractions. The remaining five beds consisted of various mixtures of sand and limestone, silt, "clayey silt," and red mud. Red mud is a by-product of aluminum extraction from bauxite. The filter beds were ineffective in removal of nitrogen from the effluent. The nitrate-nitrogen concentration in the leachate from the ten filter beds ranged from 15.1 to 32.8 mg/liter. Mean nitrate-nitrogen concentration in the leachate from the five sand filters was 23.8 mg/liter. The lowest nitrate concentration in the leachate occurred in the "silty sand" filter bed.

Brandes (1980) monitored nitrogen concentration in effluent passing through a 3.3 ft (1 m) thick sand filter during a two year period in Ontario, Canada. The OSDS served a house with eleven occupants. The sand filter treatment converted 98% of the ammonia in the effluent to nitrate. A reduction of 46.5% of the combined total (Kjeldahl) nitrogen, nitrite, and nitrate concentration was also found. Adjusting the nitrogen concentration for dilution due to precipitation, Brandes estimated actual nitrogen reduction to be 37.8%. The mean concentration of nitrate-nitrogen in the outflow from the sand filter was 32.2 mg/liter.

Kristiansen (1981a,b) noted that 11% of the total nitrogen in septic tank effluent was removed by passage through 45 inch (115 cm)-deep sand filter trenches at loading rates of 1.6 to 2.4 and 4.7 to 7.1 inches (4 to 6 and 12 to 18 cm) per day. Maximum nitrification occurred in an aerobic zone located 3 to 10.6 inches (7.5 to 27 cm) below the biological clogging or crust zone. Fifty to sixty-five percent of the total applied nitrogen underwent nitrification. Because of aerobic soil conditions and the lack of a suitable energy source in zones containing nitrate, insignificant denitrification occurred. Nitrogen, bound in the microbial biomass, adsorbed to the cation exchange complex, and bound to organic materials in a 65 ft (20 m)-long sand filter, was estimated to be equivalent to approximately three months of nitrogen loading from a single-family OSDS.

Meek et al. (1970) studied the removal of nitrate from 9.8 ft (3.0 m)-long soil columns over a seven-month period. The lower 5.7 ft (1.75 m) of the columns was subjected to anaerobic saturated conditions. Soil texture within the columns ranged from silt loam to clay, with organic carbon content being increased by addition of ground hay to the upper 4 inches (10 cm) of the soil. An initial mean nitrate-nitrogen concentration of 5.2 mg/liter in the soil solution above the saturated zone decreased to a mean nitrate- nitrogen concentration of 0.5 mg/liter and did not exceed a maximum concentration of 1.1 mg/liter after passage through 26 inches (65 cm) or 49 inches (125 cm) of submerged anaerobic soil. Denitrification in the submerged zone was related to nitrate concentration and to the quantity of soluble carbon carried down to the submerged zone in the soil solution. The carbon-rich compounds provided an energy source for denitrifying bacteria.

Hill (1972) reported on nitrate movement of a synthetic effluent applied to six Connecticut soils ranging in texture from sandy loam to silty clay loam. One inch (2.5 cm) of the synthetic effluent, containing 5 mg/liter of nitrate-nitrogen, was applied semiweekly for a two-year period to organic-rich soils packed in columns 3 ft (1 meter) in height. Nitrate concentration in the outflow during the first year of application was 125 to 150% greater than could be accounted for by the nitrogen added. The increased nitrate concentration was attributed to mineralization of organic nitrogen in the soil. After fifteen to eighteen months of effluent application, three soils showed evidence of a 10 to 20% nitrate reduction in outflow attributed to denitrification as a result of anaerobic waterlogged conditions. Only one soil continued to show slight nitrate reductions after two years of effluent application.

Lance and Whisler (1972) indicated that short, intermittent flooding of soil columns with secondary sewage wastewater from Phoenix, Arizona for two days caused no net removal of nitrogen from the wastewater, but did cause transformation of almost all of the nitrogen to nitrate. Longer periods of flooding, ranging from 9 to 23 days, resulted in a net nitrogen removal of 30% from the wastewater. Alternate flooding and drying periods were necessary for consistent nitrogen removal. The removal of nitrogen from the wastewater was related to a combination of several reactions dominated by denitrification.

Tilstra et al. (1972) studied nitrogen removal from secondary wastewater in field and laboratory lysimeters utilizing Detroit Lakes peat in Minnesota. The lysimeters were 46 inches (117 cm) in height and 8 inches (20 cm) in diameter. Leachate from four field lysimeters averaged 0.42 to 2.48 mg/liter of ammonium-nitrogen and 0.49 to

0.65 mg/liter of nitrate-nitrogen under saturated flow conditions. Under unsaturated flow conditions, leachate from two laboratory lysimeters averaged 0.06 to 0.08 mg/liter of ammonium-nitrogen and 36.4 to 42.2 mg/liter of nitrate-nitrogen. They concluded that aerobic, water-unsaturated flow conditions would result in nitrate contamination of field drainage water and the underlying ground water.

Hardisty (1973) and Laak et al. (1974) applied effluent to aerobic, unsaturated soil in experimental soil boxes constructed to simulate drainfield trenches in Connecticut. The passage of effluent through 18 inches (46 cm) of sand or silt resulted in a 30% decrease in the nitrogen concentration of the effluent. Under aerobic unsaturated soil conditions, approximately 80 to 90% of the nitrogen in the effluent was converted to nitrate. The nitrate-nitrogen concentration in the outflow from the soil boxes was approximately 15 mg/liter. Application rate of the effluent was not indicated.

John (1974) applied municipal sewage wastewater from Kelowna, British Columbia, Canada to 8 inch (20 cm)-high soil columns containing soils with varied physical and chemical properties. Semiweekly applications of 2.8 inches (7 cm) of sewage wastewater were applied for seventy-seven days. During the first ten wastewater applications, the nitrate concentration in the leachate or outflow was lower than the nitrate concentration in the sewage wastewater. Subsequent wastewater applications resulted in an increased nitrate concentration in the leachate, exceeding the nitrate concentration of the wastewater. The increase in nitrate concentration of the leachate was related to mineralization of organic nitrogen and nitrification of ammonium. The mean nitrate-nitrogen concentration in the leachate for all soil columns was 27.3 mg/liter.

Magdoff et al. (1974b) reported nearly complete nitrification of septic tank effluent after passage through as little as 12 inches (30 cm) of aerobic, water-unsaturated sandy loam or sand in Wisconsin. The passage of effluent through an additional 12 inches (30 cm) of silt loam under anaerobic conditions resulted in a 32% decrease in nitrogen concentration. This was attributed to denitrification. The lack of a sufficient energy source may have restricted additional denitrification.

Continuous ponding of effluent in the columns initially resulted in a decrease in the nitrogen concentration of the outflow. Following the initial decrease, the nitrogen concentration of the outflow increased gradually until it reached approximately 75% of the nitrogen concentration of the effluent. Nitrification was inhibited under these conditions and was reflected in the higher ammonium concentration of the outflow. Recycling of mineralized nitrogen, mechanical sieving of particulate material by the crust, sorption of organic nitrogen, and resistence of some of the organic nitrogen to degradation were identified as factors which prevented equilibration between the outflow nitrogen concentration and effluent nitrogen concentration.

Several studies have attempted to reduce the nitrate concentration in OSDS effluent by enhancing denitrification. Sikora and Keeney (1975) demonstrated that the nitrate concentration of aerated septic tank effluent could be reduced to very low levels by adding methanol as an energy source for denitrification. A nitrate concentration of 40 to 50 mg/liter in aerated septic tank effluent was reduced to 0.4 mg/liter after passage through 5.5 inches (14 cm) of calcium carbonate-packed columns to which methanol had been added.

Sikora and Keeney (1976) evaluated a sulfur-*Thiobacillus denitrificans* nitrate-removal system. Continuous passage of nitrified septic tank effluent containing 40 mg/liter of nitrate-nitrogen through 25 inch (64 cm)-long soil columns containing *T. denitrificans* bacteria resulted in nearly complete nitrate removal from the effluent. However, sulfate at concentrations as high as 90 mg SO_4/liter was detected in the outflow and may limit applicability of the *T. denitrificans* nitrate reduction systems.

Fetter (1977) applied secondarily treated wastewater from Oshkosh, Wisconsin to two 15 ft (4.6 m) soil columns constructed of calcareous glacial outwash. The glacial outwash contained approximately 30% silt plus clay. The wastewater was applied at a 4 inch (10 cm)/day loading rate for a period of ten weeks. The soil columns were ineffective in removal of nitrogen from the wastewater. Following the growth of nitrifying bacteria, organic nitrogen, ammonium, and nitrite were oxidized to nitrate. The combined concentration of nitrate, nitrite, and ammonium in the wastewater was 16.6 mg N/liter. Nitrate-nitrogen concentrations in the two column outflows were 20.0 and 18.4 mg/liter, indicating that mineralization of organic nitrogen was occurring.

In a study by Stewart et al. (1979) in North Carolina, soil columns which contained a 37 inch (90 cm) water-unsaturated zone overlying a 24 inch (60 cm) saturated zone were dosed twice daily for 220 days with 0.65 inch (1.65 cm) of effluent. Conversion of the various nitrogen forms of the effluent occurred in the unsaturated zone. Near the bottom of the unsaturated zone, the nitrate concentration of the effluent accounted for 87 to 94% of the total nitrogen in columns with a loamy sand texture and for 89 to 91% of the total nitrogen in columns with a sand texture. The use of soil organic matter as an energy source for denitrification in the saturated zone initially resulted in a 93% reduction in nitrate concentration after forty-two days of effluent application. Nitrate reduction dropped to 28% after ninety-three days of effluent application, as a result of depletion of the organic carbon energy source.

Willman (1979) and Willman et al. (1981) examined nitrogen removal from effluent by soil columns during a twenty-three week period in Pennsylvania. Three mineralogically different sand fractions derived from limestone, sandstone, and shale were mixed with varying amounts of clay to produce columns which ranged in clay content from 0 to 12%. Very little nitrogen was removed by the soil columns but the transformation of ammonium to nitrite and nitrate was related to sand type and to clay content. Ammonium was not detected in the outflow from soil columns containing sand derived from shale, because ammonium was adsorbed on the exchange complex and subsequently converted to nitrite and nitrate. Ammonium was detected in outflow from soil columns containing sand derived from limestone and sandstone, with its concentration related to the clay content of the columns. The ammonium concentration in the outflow decreased with time and corresponded to increases in nitrite and nitrate. By the end of the fifteenth week of effluent application, the ammonium and nitrite concentrations in the outflow decreased to zero, whereas the concentration of nitrate in the outflow approached the concentration of ammonium in the effluent.

Lance et al. (1980) applied primary and secondary municipal sewage wastewater from Phoenix, Arizona to 8.3 ft (2.5 m)-long columns containing loamy sand. An alternating schedule of nine days of flooding and five days of drying was used to apply the wastewater. Nitrogen removal from the secondary wastewater by the soil columns averaged 28.5%. Nitrogen removal from the primary effluent by the soil columns averaged 45.6%. The increased nitrogen removal in the primary effluent resulted from increased denitrification due to the higher dissolved organic carbon content of the effluent.

3.2.5. Summary

The predominant forms of nitrogen in septic tank effluent are ammonium-nitrogen (75%) and organic nitrogen (25%). The nitrogen concentration of effluent ranges from 40 to 80 mg/liter, with an average family of four generating about 44 to 73 lbs (20 to 33 kg) of nitrogen per year (Walker et al., 1973b; Siegrist et al., 1976; Harkin et al., 1979).

Transformation, retention, loss or movement of nitrogen in natural soil systems is governed by mechanisms of mineralization, nitrification, denitrification, adsorption, biological uptake, and volatilization. Ground water monitoring studies and laboratory column studies indicate that approximately 20 to 40% of the nitrogen in effluent may be adsorbed or otherwise removed before the effluent reaches ground water (De Vries, 1972; Andreoli et al., 1979; Harkin et al., 1979; Peavy and Brawner, 1979; Lance et al., 1980; Starr and Sawhney, 1980; Laak, 1982).

In a properly sited, designed, constructed, and operated conventional OSDS, nitrification is the predominant N-transformation mechanism for aerobic, water-unsaturated soil beneath the biological clogging mat or crust. Nitrification results in conversion of ammonium and organic nitrogen to nitrate. The soil cation exchange capacity is ineffectual in sorbing nitrate, a soluble anion, so nitrate moves nearly uninhibited to the ground water. In a worst-case situation, the nitrate concentration of percolate beneath a soil absorption system would approximate the total nitrogen concentration of the effluent (Sikora and Keeney, 1976).

Numerous ground water monitoring studies have detected nitrate-nitrogen concentrations exceeding 10 mg/liter at considerable distance from absorption systems. Attenuation of nitrate by dilution is the only mechanism which significantly lowers nitrate-nitrogen concentration in the ground water below conventional OSDSs in aerobic, water-unsaturated soils. The concentration of nitrate in ground water decreases as the nitrate diffuses and is dispersed into surrounding ground water of lower nitrate content. Denitrification within a properly sited, designed, and operated conventional OSDS is unlikely. Under conditions of high water tables or slowly permeable soils, however, nitrate may be denitrified if a biologically useful source of organic carbon is readily available.

Water quality surveys throughout the United States have identified local and regional contamination of ground water and surface water by nitrate derived from OSDSs. Restricting OSDS density (the number of OSDSs per unit land area) lowers the nitrate input from OSDSs to ground water per unit land area. This in turn may effectively control levels of nitrate in ground water.

3.3. Phosphorus

3.3.1. Fate and transport

Phosphorus in septic tank effluent originates from two main sources: detergents with phosphate builders, and human excreta (Sikora and Corey, 1976). The relative contribution of detergent phosphorus will vary with the amount of detergent used and its phosphorus content. Sawyer (1965) and Siegrist et al. (1976) have estimated that detergent accounts for approximately 50 to 75% of the total phosphorus, while human excreta account for approximately 15% of the total phosphorus.

Median total phosphorus concentration in septic tank effluents is approximately 16 mg P/liter and ranges from 11 to 31 mg/liter (Magdoff et al., 1974b; Bernhart, 1975; Otis et al., 1975; Siegrist et al., 1976). The estimated per capita total phosphorus contribution from OSDSs ranges from 1.75 lbs P/yr (0.8 kg P/yr) to 6.6 lbs P/yr (3.0 kg P/yr) (Otis et al., 1975; Siegrist et al., 1976). Approximately 20 to 30% of the total phosphorus in raw wastewater is removed via sludge which accumulates in the bottom of the septic tank (Hardisty, 1973; Laak & Crates, 1978; Laak, 1982).

The principal forms of phosphorus present in septic tank effluents are orthophosphate (85%), and condensed phosphates (meta-, pyro, and tripolyphosphate) and organic phosphorus (15%). Anaerobic digestion occurring in a conventional septic tank converts most of the organic and condensed phosphate in the raw effluent into soluble orthophosphate ($PO_4^{=}$, $H_3PO_3^{=}$, & H_2PO_4) (Jones and Lee, 1977a,b). Magdoff et al. (1974) and Otis et al. (1975) found more than 85% of the total phosphorus in most septic tank effluents to be in the orthophosphate form.

In contrast to the highly mobile nitrate nitrogen ion, (NO_3-N), most phosphate reacts vigorously with soils. It is the major anion to exhibit such behavior in the soil environment (Preul & Schroepfer, 1968). Phosphate ions are removed from the soil solution by the mechanisms of adsorption, chemisorption, precipitation, plant uptake, and biological immobilization. Tofflemire et al. (1973) and Sawhney (1977) have indicated that removal of phosphate ions from dilute solutions is a two-phase process. The initial rapid removal of phosphate was described as occurring generally through physical adsorption and as being 80 to 90% complete in two to five days. Uptake during the subsequent slow reaction phase or at high phosphate concentrations was attributed to sorption and precipitation reactions with aluminum, iron, and calcium.

The relative importance of aluminum, iron, and calcium can be roughly correlated with soil pH (Enfield and Bledsoe, 1975). In acid soils, iron and aluminum phosphates control the equilibrium concentrations of phosphate ions in solution. In basic or alkaline soils, calcium phosphate controls the equilibrium concentration (Polta, 1969; Beek and DeHaan, 1974; Laak et al., 1974; Enfield and Bledsoe, 1975; Hall, 1975; Sikora and Corey, 1976).

In the pH range encountered in most OSDS drainfields, hydroxyapatite is the stable calcium phosphate precipitate. At high phosphorus concentrations, dicalcium phosphate or octocalcium phosphate is formed initially, followed by slow conversion to hydroxyapatite (Lindsey and Moreno, 1960; Magdoff et al., 1974).

Magdoff and Keeney (1976) reported immobilization of 121 ug P/gram of soil and 307 ug P/gram of soil, respectively, for loamy sand and silt loam-textured soil columns to which septic tank effluents were applied.

Walker et al. (1973) determined that phosphorus sorbed by sandy soils beneath OSDSs in Wisconsin ranged from 100 to 300 ug P/gram of soil. Tofflemire and Chen (1977) have reported phosphate sorption capacities as high as 176 mg/gram of soil in isotherm experiments. Magdoff et al. (1974) and Lance (1977) found that phosphate removal from effluent by calcareous sand columns was proportional to the loading rate. Columns with low flow velocities and thus long contact times of solutes with soil particles removed more phosphate than similar columns with higher velocities.

Phosphate ions, like ammonium ions, advance through soil as sorption sites undergo stepwise sorption-desorption (Dudley and Stephenson, 1973; Magdoff et al., 1974; Sawhney, 1977). The average depth of phosphorus penetration can be estimated if the loading rate and phosphorus immobilization capacity of the soil are known.

Ellis and Erickson (1969) attempted to predict the amount of time required to saturate a soil volume with phosphorus from an OSDS. Ellis and Erickson (1969) applied Langmuir adsorption theory to phosphorus uptake for an array of Michigan soils as measured in the laboratory. They then used these estimates to predict that phosphorus saturation of soil below an OSDS could occur in six months to six years. Sikora and Corey (1976) estimated that depth of phosphorus penetration in sandy Wisconsin soils would be about 20 inches (50 cm) per year, while in finer-textured soils it could be as little as 4 inches (10 cm) per year.

Sawhney and Hill (1975), Sawhney (1977), and Sawhney and Starr (1977) have indicated that regeneration of phosphorus adsorption sites as a result of slow precipitation reaction mechanisms causes the soil's adsorption capacity to be greater than that determined by equilibrium adsorption measurements. However, regeneration rates of phosphorus adsorption sites of soils under different conditions are unknown (Sawhney and Hill, 1975). Sikora and Corey (1976) have indicated that subjecting a previously well-aerated, non-calcareous soil to reducing conditions will result in desorption of phosphate ions. Much of the phosphorus in acid soils is bound to ferric iron (Fe^{+++}), which is converted to soluble ferrous iron (Fe^{++}) under reducing conditions. The phosphorus is then released to the soil solution where a new equilibrium is established with aluminum- and/or calcium-bound phosphates (Bouma, 1979).

Hill (1972) reported that a fluctuating water table may disturb the phosphate equilibrium, causing phosphorus to go into solution and be transported. However, Ellis and Childs (1973) and Jones and Lee (1977a,b) have indicated that phosphate adsorption may occur below the water table.

The importance of sorption and precipitation reactions is readily demonstrated when one examines the chemical characteristics of ground waters. High phosphorus content in ground water is rarely observed (Jones and Lee, 1977a,b). The total phosphorus content of ground water often ranges from 0.005 mg to 0.1 mg P/liter (Stumm and Morgan, 1970; Dudley and Stevenson, 1973; Tofflemire et al., 1973; Enfield and Bledsoe, 1975).

Although the addition of phosphorus to ground water poses little or no health hazard (Jones and Lee, 1977), the phosphorus content of surface waters is frequently the limiting factor for the growth of algae and aquatic weeds (Holt et al., 1970). Hutchinson (1957) states that most relatively uncontaminated lakes have surface waters containing 0.01

to 0.03 mg P/liter. Algae growth is severely limited by phosphorus concentrations below 0.01 mg P/liter (Taylor, 1967). In contrast to nitrate, phosphate is relatively immobile in ground water due to the mechanisms of sorption and precipitation (Sikora and Corey, 1976). However, very dilute concentrations of phosphorus in ground water may be of sufficient concentration to cause contamination of surface waters.

3.3.2. Water quality surveys

La Valle (1975) investigated various domestic sources associated with stream phosphate levels in twenty-four watersheds near Windsor, Ontario, Canada. A multiple regression analysis revealed that 76% of the spatial variability in orthophosphate content of streams could be explained by the presence of OSDSs. Wherever high stream phosphate concentrations were observed, a large percentage of the residences were served by OSDSs.

3.3.3. Ground water monitoring studies

Hansell (1968) and Polkowski and Boyle (1970) monitored phosphate migration in ground water adjacent to an OSDS. The total phosphorus concentration of ground water sampled 15 ft (4.6 m) downgradient from the drainfield was 0.04 mg/liter. The majority of ground water samples from monitoring wells surrounding the drainfield at distances of 20 to 90 ft (6 to 29 m) had total phosphorus concentrations of less than 0.01 mg/liter. They concluded that the soil at the site removed essentially all of the phosphorus present in the effluent.

Wall and Weber (1970) reported phosphorus concentrations in ground water which were above background levels for ten of fourteen sites investigated near Canal Lake, Ontario, Canada. Soils at the sites ranged in texture from sand to clay loam and had water tables at depths from 18 to 42 inches (46-107 cm) below the soil surface. The mean phosphorus concentration in ground water for all sites was 0.11 mg P/liter but ranged from 0.01 to 2 mg P/liter. The distance from drainfields to monitoring wells and the depth of monitoring wells was not indicated.

Ellis (1971) attempted to assess the impact of OSDSs on phosphate contamination of Gull Lake, Michigan. The input of phosphorus to ground water from fertilizers made the determination very difficult. Only three of twelve locations monitored revealed phosphate movement greater than 20 ft (6.1 m) from the point of effluent discharge. Maximum detected phosphate movement was 30 ft (9.1 m). Ellis stated that until the soil become saturated with phosphorus, nearly 98% of the phosphorus would be adsorbed by the soil. In five samples collected from tile lines draining toward Gull Lake, Ellis and Erickson (1969) found phosphorus concentrations which varied from 0.028 to 0.07 mg P/liter.

A study of six OSDSs (four cesspools, two septic tank/drainfields) in Nassau and Suffolk Counties, New York by the New York State Department of Health (1972) concluded that essentially complete reduction of phosphorus occurred within the distances monitored. Reductions in total phosphorus ranging from 97 to 99.9% were noted within a distance of 31 to 80 ft (9.1 to 24.4 m) from the OSDSs. A reduction of 34 to 52% of the total phosphorus concentration in effluent occurred after passage through 18 inches (46 cm) of unsaturated soil below an OSDS at one site. The total phosphorus concentration of ground water 65 ft (19.8 m) from another OSDS ranged from 0.03 to 0.6 mg/liter. A greater reduction of total phosphorus concentration in ground water was detected in soils with finer texture and water-unsaturated soil conditions.

Bouma et al. (1972) monitored phosphorus concentration in ground water adjacent to numerous OSDSs throughout Wisconsin. The phosphorus concentration in ground water adjacent to the OSDSs was similar to background phosphorus concentrations in the majority of systems monitored. Phosphorus movement was detected in only two of twenty-four OSDSs examined. Soil texture, depth to water table, and age of the system were found to influence the phosphate concentration in ground water. A dissolved inorganic phosphorus concentration approaching 2.0 mg/liter was detected in a monitoring well 10 ft (3.0 m) from a twelve-year-old system constructed in loamy sand. During periods of a seasonally high water table, the phosphorus concentration was as high as 9.5 mg/liter in a monitoring well adjacent to an eight-year-old OSDS constructed in peat and sand. The phosphorus concentration decreased to less than 0.05 mg/liter as the water table receded.

Lee (1972, 1976) monitored movement of nutrient-enriched ground water into Lake Sallie, Minnesota using seepage meters installed in the lake bottom. The highest concentration of phosphate in seepage inflow was found along the shore where OSDSs were used. Water collected from the seepage meters initially contained highly variable concentrations of orthophosphate ranging from 0.21 to 1.15 mg PO_4/liter. The mean orthophosphate concentration in seepage inflow was 0.15 mg PO_4/liter after 317 gallons (1200 liters) of flow-through. Lee (1976) estimated that ground water inflow along the 2600 ft (800 m) shoreline transported 81.4 lbs (37 kg) of phosphorus per year into the lake.

Brandes (1972) monitored movement of phosphorus in ground water from an OSDS located approximately 27 ft (8.2 m) from Lake Chemong, Ontario, Canada. The OSDS was constructed in sandy loam fill material containing "stones and boulders." The depth to the water table ranged from 5 to 7 ft (1.5 to 2.1 m). Phosphorus concentration in ground water sampled at distances of 5 and 22 ft (1.5 and 6.7 m) from the absorption system contained 14.5 and 11.6 mg P/liter, respectively. A surface water sample collected from the lake contained 0.09 mg P/liter. Brandes concluded that the OSDS constructed on "loose fill material" was contributing to lake pollution.

Dudley and Stephenson (1973) investigated phosphorus concentrations in ground water surrounding OSDSs at eleven sites in Wisconsin. Phosphorus contamination of ground water was noted for three systems constructed in coarse sands and gravels. A phosphorus concentration of 7 mg/liter was detected in ground water sampled 5 ft (1.5 m) from a seepage pit and decreased to 1 mg/liter at a distance of 40 ft (12.2 m) downgradient. The water table was located 17 ft (5.2 m) below the system. Another site had a mean phosphorus concentration of 2.7 mg/liter in ground water sampled 30 ft (9.1 m) from the system. At a third site, a mean phosphorus concentration of 1.6 mg/liter was detected in ground water adjacent to the system. The water table was located 50 ft (15.2 m) below the soil surface.

The remaining sites had finer-textured soils, so phosphorus concentrations were consistently less than 0.5 mg/liter at all distances and depths. Greater than 99% removal of phosphorus from effluent was indicated for several sites at distances of 15 to 30 ft (4.6 to 9.1 m) from the systems. Dudley and Stephenson concluded that, although phosphorus enrichment of ground water occurred in several systems constructed in coarse sands and gravels, effective phosphorus removal occurred for systems constructed in finer-textured soils.

Sanborn (1973) monitored fluctuations in total phosphorus concentrations of ground water adjacent to two OSDSs. The fluctuations in phosphorus concentration of the ground water were related to changing soil moisture contents and water table depths. Phosphorus in the effluent was immobilized by the soil surrounding the drainfield during periods of low soil moisture associated with a receding water table. During periods of high soil moisture associated with a high water table, phosphorus in the effluent was immobilized to a lesser extent, or previously adsorbed phosphorus was desorbed. Sanborn's results suggest that phosphorus retained in soil under one set of conditions may be removed and transported when these conditions change.

Ellis and Childs (1973) and Childs et al. (1974) monitored nineteen OSDSs surrounding Houghton Lake, Michigan. The soils at sixteen sites ranged in texture from sand to loam and three sites had soils developed in peat or muck. Depth to the water table ranged from 1 to 5 ft (0.3 to 1.6 m) below the soil surface. Phosphorus ranging in concentration from 2 to 7 mg PO_4/liter moved downgradient 20 to 100 ft (6.1 to 30.5 m) at six of the sites investigated. Phosphorus movement generally occurred at sites with loamy sand soils and for water table depths ranging from 2.5 to 3.5 ft (0.8 to 1.3 m) below the soil surface. The phosphorus was generally concentrated in the upper 4 ft (1.2 m) of the ground water. Phosphorus movement was not detected in thirteen of the sites monitored.

Childs (1974) estimated the annual phosphorus load transported into Houghton Lake from all sources to be 1000 lbs (455 kg), based on ground water sample analyses, an area water budget, and net flow calculations. The 1000 lbs (455 kg) of phosphorus represents 2.5% of the estimated 40,000 lbs (18,182 kg) of phosphorus discharged annually by local residents via OSDSs. Childs concluded that the majority of phosphorus discharged through OSDSs is retained by soils and sediments within the zone of saturation.

Missimer Associates Inc. (1976) monitored ground water quality at four sites on Sanibel Island, Florida. Two of the sites investigated were located in subdivisions served by OSDSs and the remaining two sites were located in undeveloped areas. Soil at the sites was identified as a Canaveral sand with a water table at a depth of 12 to 36 inches (30 to 90 cm) below the soil surface. At a site adjacent to West Rock Lake, the orthophosphate concentration in ground water ranged from 0.02 to 0.03 mg P/liter in background wells and from 0.86 to 4.0 mg P/liter in wells adjacent to or downgradient from the drainfield. Total phosphorus concentration in background wells ranged from 0.05 to 0.07 mg P/liter, with values from 1.3 to 5.2 mg P/liter in wells adjacent to or downgradient from the drainfield. A second OSDS, monitored with only two wells, which were located upgradient from the OSDS, had orthophosphate and total phosphate concentrations below levels detected at two undeveloped sites. Missimer Associates Inc. concluded that contaminants from OSDS effluent were moving into the surface water bodies of Sanibel Island via the shallow ground water aquifer.

Viraraghavan and Warnock (1976a,b,c) reported phosphate concentrations in groundwater below an OSDS ranging from 0.25 to 21.6 mg P/liter for a study in Ottawa, Ontario Canada. Soil at the site was described by visual examination as consisting of 2 ft (0.6 m) of sandy clay, overlying clay with less sand at a depth of 2 to 5 ft (0.6 to 1.5 m). The range in phosphate concentration was related to uneven effluent distribution in the drainfield and to variation in water table depth. Phosphorus concentration in ground water beneath the OSDS reflected a 25 to 50% reduction when compared to effluent phosphorus concentration. Phosphorus concentration in ground water downgradient from the OSDS was not investigated.

The concentration of phosphorus in ground water around two OSDSs located in soils with perched water tables was monitored in Virginia by Reneau and Pettry (1976). Soils at the sites had textures of sandy clay loam or clay loam beneath the absorption system and had slowly permeable soil horizons below a depth of 3.3 ft (1 m). Soluble orthophosphate with a mean concentration of 0.42 mg P/liter was the predominant phosphorus species for the forty-six ground water samples. Orthophosphate concentration of the ground water decreased from a mean of 5.5 mg P/liter in the effluent to 1.05 mg P/liter at a distance of 6 inches (15 cm) from the drainfield at one site. The orthophosphate concentration in ground water continued to decrease with distance from the drainfield and was reduced to 0.01 mg P/liter at 39 ft (12 m).

During periods of low water tables at the second site, Reneau and Pettry concluded that phosphorus moved primarily in the vertical direction until a perched water table or restrictive layer was encountered. Subsequent ground water flow would be primarily in the horizontal direction. Orthophosphate concentration in the ground water at a depth of 56 to 60 inches (142 to 152 cm) decreased from 11.8 mg P/liter in the drainfield tile to 0.91 mg P/liter, 0.21 mg P/liter, and less than 0.01 mg P/liter at distances of 6 inches (15 cm), 10 ft (3 m), and 44 ft

(13.5 m), respectively, from the drainfield. Transport of phosphorus to the permanent ground water table was restricted by a plinthic horizon at one site and by a slowly permeable horizon at the other site.

Beek et al. (1977) reported that surface flooding, with raw domestic sewage effluent, of a sandy soil in the Netherlands over a fifty-year period resulted in an accumulation of phosphates in the upper 20 inches (50 cm) of the soil. The soil effectively removed up to 96% of the total phosphate and orthophosphate in the effluent during a two-year monitoring study, in spite of the fact that the soil already contained appreciable quantities of adsorbed phosphorus from previous effluent applications.

A four-year monitoring study of an OSDS in Wisconsin by Jones and Lee (1976a,b, 1979) detected no phosphorus transport from the OSDS to ground water in a glacial outwash sand. A monitoring well established 50 feet (15.2 m) downgradient from the drainfield had the same total phosphate and orthophosphate concentrations as background monitoring wells. Orthophosphate concentration in nearly all of the ground water samples was less than 0.01 mg P/liter. Phosphate concentration in ground water adjacent to the OSDS was not determined.

Sawhney and Starr (1977) determined the phosphorus concentration of soil solutions obtained with tensiometers and suction probes at different distances below and beside a six-year old OSDS drainfield trench in Connecticut. The morphological and physical properties of the soil at the site were not indicated. Phosphorus concentrations of soil solution collected 6 inches (15 cm) below the trench approached the phosphorus concentration of the effluent, indicating that phosphorus sorption sites had been saturated to that depth. Phosphorus concentrations ranging from 0.38 to 2.5 mg P/liter at a depth of 12 inches (30 cm) below the trench indicated that the soil at this depth continued to remove a large portion of the phosphorus for the effluent. The mean phosphorus concentration of the soil solution sampled at the depth of 24 inches (60 cm) below the trench was 0.35 mg P/liter. Concentrations ranged from 0.02 to 0.6 mg P/liter. Alternating the effluent discharge between different drainfield trenches over six-month periods resulted in regeneration of sorption sites, likely due to slow precipitation of stable phosphorus compounds at the various depths and to increased removal of phosphorus from the soil solution. Sawhney and Starr (1977) concluded that a minimum of 24 inches (60 cm) of unsaturated soil above a seasonally high or perched water table was necessary to prevent undesirably large additions of phosphorus to the ground water.

Tilchin et al. (1978) detected phosphorus concentrations as high as 11 mg/liter and 3 mg/liter in ground water 17 ft (5.2 m) from two OSDSs located in poorly drained organic soils of Michigan. Monitoring wells 70 ft (21.3 m) from the systems detected phosphorus concentrations of 1 to 2 mg/liter. A phosphorus concentration of 1 mg/liter was detected in ground water adjacent to an overloaded OSDS located in a well-drained sandy loam. The phosphorus concentration decreased to 0.4 mg/liter in ground water 27 ft (8.2 m) from the system. Monitoring wells surrounding an additional six OSDSs did not yield ground water phosphorus concentrations which were above background concentrations.

Peavy and Brawner (1979) and Peavy and Groves (1978) monitored phosphate concentrations in ground water adjacent to an OSDS located in Montana. Phosphate concentrations ranged from 0.18 to 0.94 mg PO_4/liter in ground water sampled downgradient from the drainfield. Background phosphate concentration of the ground water was 0.13 mg PO_4/liter. They concluded that the soil was effective in removal of phosphorus from effluent, and that little phosphate from OSDS effluent reached the ground water. However, once enrichment of the ground water had occurred, substantial further removal of phosphorus ceased.

Reneau (1979) monitored the phosphorus concentration in shallow ground water adjacent to three OSDSs in Virginia. The systems were located in somewhat poor to poorly drained soils which were artificially drained by a nearby agricultural drainage tile. Decreasing concentration of phosphorus in the ground water between the drainfields and the tile drainage system were described by logarithmic equations. As much as 87% of the observed variation in phosphorus concentration could be attributed to distance of the monitoring wells from the drainfields. Passage of effluent through approximately 26 ft (8 m) of soil under water-unsaturated conditions resulted in a 99% reduction in phosphorus concentration of the ground water. Under conditions of saturated flow, passage of effluent through 78 ft (30 m) of soil was required before comparable reductions were noted. The phosphorus concentration in ground water 60 inches (152 cm) from the three drainfields ranged from 3.7 to 6.78 mg P/liter. Water in a tile drain located 39 ft (11.9 m) from a drainfield contained 0.19 mg P/liter, while water in the same drainage tile located 63 ft (19.3 m) from another drainfield contained 0.05 mg P/liter.

Harkin et al. (1979) monitored performance of thirty-three randomly selected mound systems in Wisconsin. They reported that orthophosphate was not retained in the mound fill to any appreciable degree. Texture and particle size analysis of the mound fill was indicated for only fourteen mound systems. Twelve systems had sand-textured mound fills with contents of silt plus clay that ranged from 1.96 to 8.37% by weight. Two mound systems were constructed with loamy sand. Lack of orthophosphate retention in the mound fill was related to high effluent dosing rates (up to 1.05 gal/ft^2), which resulted in rapid flow rates through the sand, and to bacterial slime coatings, which blocked sorption sites on soil particles. Ground water sampled at four of the mound systems had total phosphorus concentrations that ranged from 0 to 1.8 mg/liter, with orthophosphate concentrations that ranged from 0 to 0.45 mg P/liter. Total phosphorus and orthophosphate concentrations in ground water from background wells were not reported.

Erickson and Bastian (1980) monitored phosphorus concentrations in ground water adjacent to OSDSs installed at rest stops in Michigan. Ground water in a water table located 25 to 30 ft (7.6 to 9.1 m) below a conventional septic tank-drainfield system did not contain orthophosphate in concentrations greater than background levels. Another system consisting of a septic tank and a 3 to 5 ft (0.9 to 1.5 m)-deep sand filter effectively removed 70% of the total phosphorus from the effluent. However, intensive use caused overloading of the system with a subsequent breakthrough of phosphorus.

Rea and Upchurch (1980) reported orthophosphate concentrations of 7 mg P/liter in ground water sampled 6.5 ft (2 m) beneath a seepage pit examined in Florida. The seepage pit extended 5 ft (1.5 m) below the water table, resulting in direct discharge of effluent into the ground water. Orthophosphate concentrations in the ground water decreased to 1.5 mg P/liter and to less than 1 mg P/liter at lateral distances of 40 ft (12.2 m) and 60 ft (18.3 m), respectively, from the seepage pit.

Whelan and Barrow (1980) noted that the amount of phosphorus measured in the soil below an OSDS may reflect the phosphorus absorbing characteristics of the soil rather than the phosphorus concentration of the soil water. Soil solution phosphorus concentrations under unsaturated flow conditions exceeded 10 mg P/liter to a depth of 23 ft (7 m) below a seepage pit in a sandy Australian soil, despite a low soil inorganic phosphorus concentration.

Carlile et al. (1981) and Cogger and Carlile (1984) monitored phosphorus concentrations of ground water at seventeen OSDSs. Fifteen of the seventeen sites were at least seasonally saturated by high water-tables. The remaining two sites had water tables at least 12 inches (30 cm) below the base of the absorption system. The highest phosphorus concentration in the ground water was detected in the most heavily loaded systems. Six of the OSDSs had total ground water phosphorus concentrations exceeding 1.0 mg P/liter in monitoring wells located 5 ft (1.5 m) from the drainfields. Soils which were seasonally saturated generally had lower total phosphorus concentrations in ground water sampled 5 ft (1.5 m) from the drainfields than did continuously saturated soils. No difference in the total phosphorus concentration of ground water was detected at distances of 25 ft (7.6 m) and 50 ft (15.2 m) under either seasonally or continuously saturated soil conditions. The total phosphorus concentration of ground water sampled 50 ft from the drainfields was generally less than 0.5 mg P/liter.

Water and Air Research (1981) evaluated surface water and ground water quality adjacent to OSDSs near Lakes Cannon, Gordon, and Ned in Polk County, Florida. At the Lake Cannon site, total dissolved phosphorus concentrations in ground water sampled approximately 150 ft

(46 m) downgradient were 0.10 mg P/liter in a well 8.4 ft (2.6 m) deep, and 0.15 mg P/liter in a well 14.2 ft (4.3 m) deep. Total dissolved phosphorus concentrations in the ground water approximately 275 ft (84 m) from the OSDS decreased to 0.02 and 0.03 mg P/liter for shallow and deep monitoring wells, respectively. A total dissolved phosphorus concentration of 0.17 mg P/liter was detected in ground water adjacent to the OSDS at the Lake Gordon site. Total dissolved phosphorus concentrations in ground water sampled approximately 300 ft (96 m) from the OSDS were 0.15 mg P/liter for a monitoring well 6.5 ft (2 m) deep and 0.02 mg P/liter for a monitoring well 17.4 ft (5.3) deep.

At the Lake Ned site, total dissolved phosphorus concentrations in ground water sampled downgradient from the OSDS ranged from 0.01 to 0.05 mg P/liter. Water and Air Research (1981) also noted that the small number of ground water samples analyzed for phosphorus and the high concentration of phosphorus detected in background wells at the Lake Cannon and Lake Gordon sites made it difficult to assess accurately OSDS impact on phosphorus contamination of ground water and surface water in this study.

Gilliom and Patmont (1983) monitored eight OSDSs adjacent to Pine Lake, Washington. The OSDSs, ranging in age from 20 to 40 yrs, were located in permeable soils with a perched water table overlying compact glacial till. Ground water monitoring wells were installed at variable distances from the drainfields, ranging from 33 to 169 ft (10 to 50 m), and at depths ranging from 16 to 51 inches (41 to 130 cm). The mean phosphorus concentration in ground water not affected by OSDS effluent was 0.028 mg P/liter, ranging from 0.007 to 0.071 mg P/liter. The mean phosphorus concentration of ground water sampled downgradient from the

OSDSs was 0.036 mg P/liter, and ranged from 0.005 to to 0.324 mg P/liter. Effluent from the old OSDSs was reaching the lake in perched ground water flowing through the permeable soil overlying the less permeable glacial till. However, movement of more than 1% of the effluent phosphorus into the lake was rare.

Uebler (1984) monitored movement of phosphorus in soil water below absorption trenches over a twelve-month period in North Carolina. The site was located in a Cecil sandy loam, and the soil texture at the base of the trenches was clay. Saprolite (highly weathered bedrock) located at a depth of 12 inches (30 cm) below the absorption system resulted in a perched water table. Concentrations of phosphorus in soil water below absorption trenches were compared at loading rates of 0.18, 0.28, and 0.37 gallons/ft^2/day (7.5, 11.3, and 15 liters/m^2/day). In addition, phosphorus concentrations in soil water below absorption trenches treated with cement and lime, used to stabilize the adsorptive capacity of the soil, were compared with those below an unamended trench. The amendments were added to the surface of the soil trench on a 5% by-weight basis. Soil water samples were collected using porous ceramic cups placed at depths of 6 inches (15 cm) and 12 inches (30 cm) below the trenches.

Soluble phosphorus concentrations in soil water beneath the trenches were unaffected by loading rates and soil amendments, due to the high phosphorus sorption capacity of the clay. There was a greater than 95% reduction in soluble phosphorus concentrations of soil water as compared to untreated wastewater. Mean soluble phosphorus concentration of soil water for all rates and amendments was 0.56 mg/liter. Soluble phosphorus concentrations increased from 0.1 mg/liter to 0.9 mg/liter during the twelve-month sampling period. Uebler attributed the apparent increase of soluble phosphorus with time to sampling technique. Soil water samples were collected in ceramic cups which may have initially sorbed phosphorus as samples were collected. The phosphorus sorption sites of the ceramic cups would eventually become satisfied, however, leading to an apparent increase in concentration of soluble phosphorus with time. Uebler (1984) made no reference to the possibility of saturation of such sorption sites with time.

3.3.4. Sand filter and column studies

Brandes et al. (1975) determined the phosphorus concentration in leachate collected from ten underdrained filter beds. The filter media in five of the beds consisted of sand of varying size fractions. The remaining five filter beds consisted of various mixtures of sand and limestone, silt, "clayey silt," and red mud. Red mud is a by-product of aluminum extraction from bauxite. Total phosphorus removal from the effluent in the five sand-filter beds was found to be a function of grain size. Very coarse sand removed only 1% of the total phosphorus, while medium sand removed 48% of the total phosphorus from the effluent.

The addition of limestone, silt, "clayey silt," and red mud improved removal of phosphorus from the effluent by filter beds. The percentage of phosphorus removed from the effluent ranged from 73 to 88% for the amended filter beds. Total phosphorus concentration in the leachate ranged from 1.2 to 3.7 mg P/liter after passage through 30 inches (76 cm) of amended filter-bed material.

Brandes (1980) determined the phosphorus concentration in effluent passing through a 3.3 ft (1 m)-thick sand filter during a two-year period in Ontario, Canada. The OSDS served a house with eleven occupants. The sand filter treatment removed 42.2% of the total phosphorus and 30% of the orthophosphate from the effluent. The mean concentration of phosphorus in the leachate from the sand filter was 8.2 mg P/liter of total phosphorus and 7.6 mg P/liter of orthophosphate.

Tofflemire and Chen (1977) examined phosphate removal from municipal sewage wastewater in New York using three intermittent sand filters which contained approximately 2 ft (0.61 m) of sand. Average phosphate removal ranged from 18 to 56% at loading rates of 10 to 1.15 gallons/ sq.ft./day (3.52 to 0.41 liters/sq.m./day). The average phosphate removal by eighteen sand filters sampled in New York was 36%.

De Vries (1972) applied secondary wastewater containing 6 to 9 mg P/liter to 30 inch (75 cm)-long soil columns for 240 days at a rate of 8 inches (20 cm) per day for two hours. The concentration of phosphorus in the outflow reached 0.1 mg P/liter in the column filled with 0.1 to 0.25 mm soil particles, and 5.8 mg P/liter in the column filled with 0.25 to 5.0 mm soil particles.

Hill (1972) examined adsorption of phosphate applied to undisturbed cores of six Connecticut soils ranging in texture from sandy loam to silty clay loam. Soil cores measuring 12 inches (30 cm) in diameter and 3.3 ft (1 m) in height were treated semiweekly with one inch of a synthetic effluent containing 8 mg PO_4/liter for a two-year period. The soil cores retained 100% of the applied phosphate during the two-year study. Virtually all of the phosphate was adsorbed in the A horizon of the soils, while lesser amounts of phosphorus accumulated in the B horizon of several soils.

Tilstra et al. (1972) studied phosphorus removal from secondary wastewater in field and laboratory lysimeters in Minnesota utilizing Detroit Lakes peat. Under saturated flow conditions, leachate from four field lysimeters had a phosphorus concentration of 0.3 to 2.0 mg P/liter. A general trend of increasing phosphorus concentration in the leachate with time was noted. An initial reduction of 92% of the phosphorus in the wastewater was reduced to 76% after several months of wastewater application. Phosphorus removal from wastewater by peat columns under water-unsaturated flow conditions in the laboratory ranged from 95 to 99% of the phosphorus in the wastewater and did not decrease during several months of wastewater application.

Hardisty (1973) and Laak et al. (1974) applied effluent to aerobic unsaturated soil in experimental soil boxes constructed to simulate drainfield trenches in Connecticut. The passage of effluent through 18 inches (46 cm) of soil resulted in a 30% decrease in the phosphate concentration of the effluent. They concluded that passage of effluent through 18 inches (46 cm) of sand or silt was insufficient to remove phosphate adequately from the effluent. The application rate of the effluent was not indicated.

John (1974) applied municipal sewage wastewater to 8 inch (20 cm)-high soil columns containing soils with varied physical and chemical properties in British Columbia, Canada. Semiweekly applications of 100 ml of wastewater were carried out for seventy-seven days. Initially, the soil columns removed all of the phosphorus from the wastewater. Subsequent application of wastewater resulted in an average retention of 88 to 99% of the applied phosphorus in the soil columns.

The variation in retention of phosphorus by the soil columns was related to soil pH and to iron and aluminum concentrations.

Magdoff et al. (1974) examined phosphorus removal from effluent by soil columns which were constructed of 24 inches (60 cm) of sand or calcareous sandy loam glacial till overlying 12 inches (30 cm) of silt loam. The phosphorus concentration in the effluent was decreased by 29% at a depth of 22 inches (55 cm) and by an additional 30% at the point of outflow. The removal of 59% of the phosphorus from the effluent resulted in a phosphorus concentration ranging from 11 to 14 mg P/liter in the outflow. The formation of a crust or biological clogging zone at the top of the column resulted in continuous ponding of the effluent and further reduction in its phosphorus concentration. There was a 55% decrease in the concentration of phosphorus between the 0 and 22 inch (0 and 55 cm) depths and an additional 36% decrease between the 22 inch (55 cm) depth and the point of outflow, for a total phosphorus removal of 91%. The phosphorus concentration in the outflow from the crusted columns ranged from 2 to 6 mg P/liter.

Fetter (1977) applied secondarily treated wastewater to two 15 ft (4.6 m) soil columns constructed of calcareous glacial outwash. The glacial outwash contained approximately 30% silt plus clay. The wastewater was applied at a 4 inch (10 cm)/day loading rate for a period of ten weeks. Reductions of orthophosphate and total phosphorus concentrations in the wastewater were in excess of 99% for both soil columns. Initial total phosphorus concentrations of 5.37 and 5.16 mg P/liter in the wastewater were reduced to 0.049 and 0.044 mg P/liter, respectively, in the two soil columns. Movement of phosphorus was restricted to the top 20 inches (50 cm) of the soil columns.

Lance (1977) investigated the removal of phosphate from secondary sewage wastewater by 8.3 ft (2.5 m)-long columns containing calcareous loamy sand. Infiltration rates in the soil columns ranging from 6 to 19.7 inches (15 to 50 cm)/day were controlled by the packing density of the columns. All columns were loaded on a schedule of nine days of flooding, alternated with five days of drying. Phosphate removal from the secondary effluent by the soil columns was inversely proportional to the infiltration rate of the soil. Orthophosphate concentration in the outflow from the columns was below 1 mg PO_4/liter for over 200 days of effluent application. The concentration then increased, and finally leveled off at different concentrations depending on the infiltration rate. Thus, the orthophosphate concentration seemed to be controlled by the travel time of the effluent in the soil column. Removal of 75 to 80% of the orthophosphate in secondary wastewater continued when the infiltration rate of the soil was less than 6 inches (15 cm)/day.

To assess the potential pollution of ground water from OSDSs, sorption capacities of various soils were determined by Sawhney (1977) over an extended period of time and related to phosphorus movement through soil columns using solutions having phosphorus concentrations similar to those of effluent. No phosphorus was detected, for example, in column outflow until approximately 50 to 60 pore volumes of the phosphorus solution had passed through the columns of fine sandy loam and silt loam. Sawhney concluded that, while most soils should effectively remove phosphorus from effluent, ground water under OSDSs installed in soils with low phosphorus sorption capacity may contain high concentrations of phosphorus after prolonged use.

Soil columns which contained a 37 inch (90 cm) water-unsaturated zone overlying a 24 inch (60 cm) saturated zone were dosed twice daily for 220 days with 0.65 inch (1.65 cm) of effluent in a study by Stewart et al. (1979) in North Carolina. The aerobic, water-unsaturated portions of the soil columns were effective in removal of orthophosphates and polyphosphates from the effluent. Column outflow and soil solution samples generally contained less than 1 mg P/liter.

Willman (1979) and Willman et al. (1981) examined phosphorus removal from effluent by soil columns during a twenty-three week period. Three mineralogically different sand fractions derived from limestone, sandstone, and shale were mixed with varying amounts of clay to produce soil columns which ranged in clay content from 0 to 12%. Virtually all of the orthophosphate and total phosphate was removed from the effluent after passage through the 24 inch (60 cm) soil columns, with the exception of the clay-free sand column derived from sandstone. The outflow from the clay-free sand column derived from sandstone reached a maximum orthophosphate concentration of 10 mg PO_4/liter, three weeks after effluent applications began. The orthophosphate concentration in the outflow subsequently decreased to very low levels during the remainder of the study. The total phosphate concentration in the outflow from the clay-free sand column increased steadily throughout the study period until it reached the concentration of phosphate in the effluent.

Hill and Sawhney (1981) examined phosphorus movement of simulated wastewater effluent containing 12 mg P/liter under aerobic and anaerobic conditions in a large undisturbed field lysimeter in Connecticut. The concentration of phosphorus in leachate discharged at depths of 18 inches (45 cm) and 30 inches (75 cm) over a two and one half year period was related to the number of effluent applications and to the aeration status of the fine sandy loam soil. The average phosphorus concentration of leachate discharged from the 18 inch (45 cm) depth was 0.1 mg P/liter shortly after the experiment began. After one year of effluent application the phosphorus concentration in leachate ranged from 0.2 to 1 mg P/liter. The phosphorus concentration of leachate rose to 1.4 mg P/liter after one and one half years of effluent application. A reduction in phosphorus concentration of the leachate occurred after temporary cessation of effluent applications. Subjecting the system to anaerobic conditions resulted in a phosphorus concentration of 2.5 mg P/liter in the leachate. At the 30 inch (75 cm) depth, the phosphorus concentration in the leachate rarely exceeded 0.1 mg P/liter during the first year of effluent application. Phosphorus concentration in the leachate ranged from 0.5 to 0.7 mg P/liter after one and one half years of effluent application. Hill and Sawhney concluded that application of effluent to soils over a long period of time reduced the phosphorus sorption capacity of the soil, thus increasing the transport of phosphorus to ground water. Periodic resting of the system resulted in increased sorption of phosphorus. Anaerobic soil conditions enhanced mobility of phosphorus.

3.3.5. Summary

The predominant forms of phosphorus in septic tank effluent are orthophosphate (85%), and condensed phosphates (meta-, pyro-, and tripolyphosphate) and organic phosphorus (15%). The phosphorus concentration of effluent ranges from 11 to 31 mg/liter, and the median phosphorus concentration is about 16 mg/liter. A family of four generates about 1.75 to 6.6 lbs (0.8 to 3.0 kg) of phosphorus per year (Bernhart, 1975; Otis et al., 1975; Siegrist et al., 1976).

Phosphorus is retained or immobilized in natural soil systems by the mechanisms of adsorption, chemisorption, precipitation, and biological uptake. Ground water monitoring studies and laboratory column studies indicate that very limited phosphorus transport occurs in aerobic, water-unsaturated soils, and reduction in total phosphorus content of effluent in the soil commonly ranges from 85 to 95% or more. Under conditions of proper siting, design, construction, and operation of OSDSs, the likelihood of significant phosphorus transport to ground water and surface water is small. Phosphorus transport is likely to occur, however, in coarse-textured, non-calcareous, sandy soils that are low in organic matter with shallow depth to water tables, or in shallow soils over fractured or solution-riddled bedrock.

Phosphorus derived from OSDSs has been detected above background levels in numerous studies of ground water adjacent to OSDSs under conditions of saturated flow due to high water tables or high hydraulic loading rates. However, phosphorus concentrations in ground water are found to decrease with distance from OSDSs, because phosphorus is capable of undergoing sorption and precipitation in the ground water zone (Ellis and Childs, 1973; Childs et al., 1974; Jones and Lee, 1977a,b, 1979).

Very low concentrations of phosphorus in ground water may be sufficient to cause contamination of surface water (Holt et al., 1970). Documented cases of contamination of surface water by OSDS-derived phosphorus have been reported where OSDSs are located within close proximity [i.e., less than 100 to 150 ft (31 to 46 m)] to the surface water, or where drainage tile or drainage ditches intercept ground water before phosphorus sorption, precipitation, or uptake is complete.

3.4. Chlorides, Sulfates, Sodium, Detergent MBAS, and Toxic Organics

3.4.1. Fate and transport

Mean concentrations of chloride (Cl^-), sulfate ($SO_4^=$), and sodium (Na^+) ions in septic tank effluent are in the ranges 49 to 183, 20 to 50, and 45 to 100 mg/liter, respectively (Robeck et al., 1962; Polkowski and Boyle, 1970; Hill, 1972; Viraragahavan and Warnock, 1975; Brandes, 1980; Brown et al., 1978).

The soluble chloride ion is highly mobile and is not significantly adsorbed or exchanged (Schmidt, 1972; Miller and Wolf, 1975). McKee and Wolf (1963) and Polkowski and Boyle (1970) have indicated that dilution by ground water is the mechanism responsible for reduction of chloride concentrations. Because of its mobility in soils, chloride is frequently used as a tracer in ground water studies (Viraraghavan, 1977). Since chloride and nitrate are very mobile and both are present in OSDS effluent, a correlation between the two has often been used to document contamination of water supplies (Hill, 1972; Dudley and Stephenson, 1973; Peavy and Brawner, 1979).

Except for the regeneration phenomenon ascribed to phosphorus sorption (Sawhney and Hill, 1975), soils possess a finite exchange capacity (anion exchange capacity and cation exchange capacity) and therefore have a more or less fixed capacity to remove ions such as sulfate and sodium from effluent (Miller and Wolf, 1975). As a result, the reduction in various species present in the effluent decreases with time. Hill (1972) reported that effluent leached through soil columns initially removed 90% of the applied sulphate. Subsequently, only 40% of the added sulphate was removed during the second year, and all sulphate passed through the columns during the third year of application. A similar response was indicated for sodium adsorption. Bouwer and Chaney (1974) indicated that the ionic composition of the percolated effluent will be the same as that of the septic tank effluent after equilibrium between the cation exchange complex and the soil solution is reached.

Methylene blue active substances (MBAS) found in OSDS effluents are primarily anionic surfactants from commercial detergent formulations (Reneau and Pettry, 1975b). Surfactants comprise 10 to 30% of most modern commercial detergent formulations (Miller and Wolf, 1975). Linear alkylate sulfonate (LAS), a biodegradable surfactant, has replaced the branched chain alkyl benzene sulfonate (ABS) in recent years.

Klein and McGauhey (1965) reported that septic tank treatment removed only 9 to 15% of LAS in raw wastewater. Because LAS is not biodegradable under anaerobic conditions, only 0.6 to 2.5% of the LAS removal was attributed to biodegradation. Krishna Murti et al. (1966) reported that LAS adsorption capacities for an array of soils ranged from 5 to 48 μg/gram of soil.

The mechanisms for removal of MBAS include biodegradation and adsorption. Factors which promote aerobic conditions and unsaturated flow (i.e., low flow rates and prolonged travel times) in the soil promote biodegradation of MBAS (Polkowski and Boyle, 1970). Klein and McGauhey (1965) indicated that ABS was found to be 5 to 8 times as resistant to degradation as LAS.

The adsorption of MBAS is influenced by such soil properties as texture, mineralogic composition, clay species, organic matter content, sesquioxide content, ionic species and concentration, pH, exchangeable acidity, and formation of a biologically active clogging layer (Renn and Barada, 1959; Wayman, 1963; Robeck et al., 1964; Krishna Murti et al., 1966; Reneau and Pettry, 1975b; Miller and Wolf, 1975; Viraraghavan, 1977). Reneau and Pettry (1975b) indicated that adsorption of LAS retards movement and allows more time for biodegradation to occur.

Toxic and non-biodegradable organic compounds, chlorinated hydrocarbons, trichloroethylene (TCE), and methyl chloroform (MC) are commonly found in homeowner-administered septic tank cleaners or additives (Petura, 1981; Kaplan, 1983). Trichloroethylene and methyl chloroform have a higher specific gravity than water and are only slightly soluble in water. If allowed to contact ground water, TCE and MC sink to the bottom of the water phase, making wells that extend to considerable depth more prone to contamination than shallow wells. Federal law recognizes that hazardous substances can be manufactured and used for treating septic systems (Kaplan, 1983).

Sulfuric acid, sodium hydroxide, and peroxide are compounds that may be added to OSDSs by professional septic tank cleaners. Kaplan (1983) has indicated that use of these compounds may result in increased salinity of ground water. Bicki and Wright (1982) noted that sulfuric acid and peroxide treatment of soil samples from a biological crust sampled below a 35 year old cesspool in Rhode Island released 121 to 291 mg/liter of phosphorus that had been bound in the crust, after only 24 hours of contact time in soil columns.

Toxic substances such as pesticides, solvents, and compounds containing heavy metals discharged to OSDSs also have the potential to contaminate ground water.

3.4.2. Water quality surveys

Water quality surveys throughout the United States have identified contamination of ground water and surface water by OSDSs. Campenni (1961) reported on a survey of twenty-five water wells in a housing subdivision in Portsmouth, Rhode Island served by OSDSs. All but one of the wells tested revealed contamination by detergent MBAS ranging in concentration from 0.15 to 5 mg/liter. Coliform bacteria and elevated nitrate levels were also found in several wells.

Nichols and Koepp (1961) determined water quality of 2167 samples from private water supplies in Wisconsin. Detergent MBAS originating from OSDSs were detected in 32% of the samples and ranged in concentration from a trace to 10 mg/liter.

Morrill and Toler (1973) examined the effect of increasing density of OSDSs on stream water quality in seventeen small drainage basins in Massachusetts. Increasing density of OSDSs resulted in increasing concentrations of chloride and dissolved solids in stream flow.

A survey of 217 rural water supplies in Chesterfield County, South Carolina reported 46% of the water supplies to be contaminated with arsenic (Sandhu et al., 1975, 1976) that originated from detergents in OSDS effluents. The presence of arsenic in detergents and its potential pollution hazard have been discussed by Angino et al. (1970) and Sandhu et al. (1981).

Weimer (1980) indicated that owner-administered toxic septic system cleaners were a major source of organic solvents, such as trichloroethylene, in wells on Long Island, New York.

Wolf and Boateng (1983) reported contamination of two public water wells by chlorinated hydrocarbons in Lakewood, Washington. The source of the hydrocarbon contamination was traced to an OSDS serving a laundry and dry cleaning establishment.

3.4.3. Summary

Mean concentrations of chloride, sodium, and sulfate ions in septic tank effluent range from about 49 to 183, 45 to 100, and 20 to 50 mg/liter, respectively. Mean concentrations of linear alkylate sulfonate (LAS), an anionic surfactant in detergent, range from about 1.2 to 6.5 mg/liter (Robeck et al., 1962; Polkowski and Boyle, 1970; Viraraghavan and Warnock, 1975; Brown et al., 1978; Brandes, 1980; Whelan and Titamnis, 1982).

Chloride anions are highly mobile and are not adsorbed or exchanged in soil to a significant degree. Reduction of chloride concentration in ground water is by dilution. Soils possess a finite cation and anion exchange capacity and therefore have a more or less fixed capacity to remove sodium and sulfate ions from effluent. Reduction of sodium and sulfate ions from effluent by soil decreases with time as sorption sites become satisfied, resulting in increased levels of these ions in ground water (Hill, 1972; Bouwer and Chaney, 1974; Miller and Wolf, 1975). Aerobic, water-unsaturated soil conditions promote biodegradation and adsorption of LAS (Klein and McGaughey, 1965; Reneau and Pettry, 1975b).

Several ground water monitoring studies have reported movement of chloride, sodium, and sulfate into ground water under aerobic, water-unsaturated flow conditions. Under water-saturated flow conditions, extensive movement of LAS has been reported (Reneau and Pettry, 1975b).

Several water quality surveys have detected elevated chloride, sodium, sulfate, and LAS concentrations in ground water associated with

OSDSs. Restricting OSDS density may effectively control levels of these contaminants in ground water.

Contamination of ground water has resulted from disposal of household products containing toxic organics or heavy metals into OSDSs, and from treatment of OSDSs with "septic tank cleaners" that contain toxic wastes (Weimar, 1980; Petura, 1981; Kaplan, 1983).

3.5. Bacteria

3.5.1. Fate and transport

The fate of microorganisms in effluent as they contact the soil is an important consideration for OSDSs. How free the ground water is of human pathogens depends principally on the survival of the organisms in the soil and on the degree of retention by the soil (Gerba et al., 1975).

On-site sewage disposal system effluent may contain bacteria, viruses, protozoa, and helminths pathogenic to humans (Burge and Marsh, 1978). The occurrence of these organisms in effluent reflects the combined infection and carrier status of residents utilizing the OSDS (Berg, 1978).

Indicator organisms such as total coliforms, fecal coliforms, and fecal streptococci are enumerated most often in OSDS effluents, because the task of detecting all possible pathogens is complex and costly. It is assumed that the fecal bacteria in the OSDS effluents are the survivors of the intestinal flora, and that counts of total coliforms, fecal coliforms, and fecal streptococci can be used to reflect the possible presence of human pathogens (Bouma, 1979; Hagedorn et al., 1981). While it is a useful method, the indicator organism approach may prove to be inaccurate in some instances (Berg, 1978; Geldreich, 1978). Since pathogens are not always present in feces, the presence of fecal organisms in water does not indicate the presence of pathogens.

Ziebell et al. (1975b) enumerated selected fecal bacterial populations in effluents from five septic tanks and observed mean population densities of 3.4×10^6 organisms/100 ml for total coliforms, 4.2×10^5

organisms/100 ml for fecal coliforms, 3.8×10^3 organisms/100 ml for fecal streptococci, and 1.0×10^4 organisms/100 ml for Pseudomonas aeruginosa. Polkowski and Boyle (1970) reported total coliform densities exceeding 9.3×10^6 organisms/100 ml. Brandes (1980) reported fecal coliform densities exceeding 1.4×10^6 organisms/100 ml for septic tank effluents. Septic tank pre-treatment does little to alter the bacterial concentration of raw wastewater (Clements and Otis, 1980). However, Otis et al. (1975) and McCoy and Ziebell (1975) noted a decrease in the fecal coliform concentrations of aerobically treated septic tank effluents.

Enteric bacteria are removed from effluent in the soil by the mechanisms of filtration, sedimentation, adsorption, and natural organism die-off. Filtration appears to be the main mechanism for removal of enteric bacteria from effluent (Gerba et al. 1975). The removal of bacteria from percolating effluent by filtration and sedimentation is a function of the bacterial concentration of the effluent, soil texture, formation of a biological mat or crust layer, soil moisture tension, and hydraulic loading rate (Lance, 1978).

Lance (1978) found that reduction of fecal coliforms by filtration was proportional to the concentration of fecal coliforms applied to the soil. Butler et al. (1954) reported that removal of bacteria from percolating wastewater through a given depth of soil was inversely proportional to the particle size of the soil. They reported that sandy loam columns removed significantly more coliform organisms than did sand columns.

Formation of a biological clogging mat within the first few centimeters of the soil below a drainfield has been found to be an effective mechanism for filtration of fecal organisms. Bouma et al. (1972) and Ziebell et al. (1975a) indicated that the greatest reduction in fecal organisms in the effluent occurred in the biological clogging mat or crust at the interface of the drainfield trench and the natural soil. Column studies by Butler et al. (1954) and Krone et al. (1958) also found removal of bacteria from percolating wastewater to be highest in the first few centimeters of travel through sand. Removal was associated with the formation of a biological clogging mat or crust. In sandy soils such crusts may be beneficial by decreasing the rate of flow of effluent through the soil.

The removal of bacteria from percolating effluent also has been shown to be a function of soil moisture content. Moisture content below saturation is associated with lower hydraulic conductivity (i.e., lower rates of flow) because water flow is then restricted to smaller diameter soil pores (Bouma et al., 1972). Unsaturated flow results, therefore, in increased travel time and improved soil-effluent contact and filtration (Bouma et al., 1972; Bitton et al., 1974; Bouma, 1979). Similarly, low hydraulic loading rates promote unsaturated flow, thereby improving filtration of effluent. Ziebell et al. (1975a) reported that significantly better removal of fecal bacteria was achieved when effluent was applied at a rate of 5 cm/day (2 inches/day) to sand as compared with a 10 cm/day (4 inches/day) application rate.

Adsorption is also a factor in the removal of bacteria by soil. Bacteria are electrically charged colloidal particles that possess a net negative charge at their surface (Gerba et al., 1975). Factors that reduce the repulsive forces between the clay particles and bacteria allow adsorption to proceed.

The types and amounts of clay mineral species, nature of the cations saturating the soil, pH and ionic composition of the effluent, concentration of bacteria, and bacterial type affect adsorption (Bitton et al., 1974). Hendricks et al. (1979) reported that bacterial adsorption obeys first-order kinetics and can be described using the Langmuir isotherm model, a technique commonly employed in the study of soil-water-solute interactions.

Factors affecting survival of enteric bacteria in soils include soil moisture content, temperature, pH, organic matter content, antagonism from soil microflora, and bacterial type (Gerba et al., 1975; Parsons et al., 1975; Burge and Marsh, 1978; Lance, 1978).

Gerba et al. (1975) reported that survival of enteric bacteria under adverse conditions seldom exceeds 10 days, while Romero (1970) and Gerba et al. (1975) reported that survival under favorable conditions (high moisture content, low temperature, alkaline pH, high organic matter content) may extend beyond 100 days. Comprehensive information on survival periods of several pathogens in soil-wastewater systems is presented in Parsons et al. (1975) and in Morrison and Martin (1976).

3.5.2. Disease outbreaks

On-site sewage disposal systems rank highest among wastewater treatment techniques in the total volume of wastewater discharged to ground water (Miller and Scalf, 1974; Geraghty and Miller, 1978). Under conditions of inadequate design, construction, siting, operation, and maintenance, OSDSs are the most frequently reported source of ground water contamination (Geraghty and Miller, 1978). On the basis of five regional EPA studies (Fuhriman and Barton, 1971; Scalf et al., 1973;

Miller et al., 1974; Miller et al., 1977; Van der Leeden et al., 1974), Miller and Scalf (1974) indicated that OSDSs were key potential sources of ground water contamination in the northeastern, northwestern, southeastern, south central, and southwestern United States. Miller and Scalf (1974) and Miller et al. (1977) reported that bacterial contamination of water wells by OSDSs was the second most common reason for well replacement in the southeastern United States.

The microbial contamination of ground water is a serious problem that can result in outbreaks of waterborne diseases (Keswick and Gerba, 1980). Craun (1979, 1981) reported that from 1946 to 1977 there were 264 outbreaks and 62,273 cases of illness related to contaminated ground water. On-site sewage disposal systems were implicated in 42% of the outbreaks and 71% of the illnesses.

Disease outbreaks traced or attributed to bacterial or viral contamination of ground water by OSDSs include: hepatitis (Vogt, 1961; Garibaldi et al., 1972; Berg, 1973; Craun, 1974, 1979; McCabe and Craun, 1975), cholera (Craun, 1974), typhoid (Kingston, 1943; McCabe and Craun, 1975; Craun et al., 1976; Craun, 1979, 1981; McGinnis and DeWalle, 1983), poliomyelitis (Mack et al., 1972; Van der Velde and Mack, 1973), shigellosis (Weissman et al., 1976; Miller et al., 1977), and gastroenteritis (Kingston, 1943; Davis and Morens et al., 1979; Stephenson, 1970; Weissman et al., 1976; Morens et al., 1979; Craun, 1979, 1981).

One of the three largest outbreaks of waterborne disease in the U.S. involving contaminated ground water during the period 1946 to 1977 occurred in Florida (Craun, 1979). Approximately 1200 cases of acute gastrointestinal illness occurred in Richmond Heights, Dade County during 1974. Epidemiologic investigation and a dye tracer study disclosed that a public water well was contaminated by an OSDS approximately 125 ft (38 m) from the well (Weissman et al., 1976). Fluorescein dye introduced into the OSDS was detected 9 hours later in the public water supply.

3.5.3. Water quality surveys

Woodward et al. (1961) reported extensive ground water contamination by OSDSs in thirty-nine communities surrounding Minneapolis-St. Paul, Minnesota. The OSDSs were installed in soils overlying till, sand and gravel, and/or jointed, fractured or solution-riddled limestone. A survey of 63,000 individual wells revealed that 48% of the wells had elevated nitrate-nitrogen concentrations (11% contained greater than 10 mg/liter of nitrate-nitrogen) and 22% of the wells contained measurable quantities of detergent MBAS. Twenty to fifty percent of the wells in various communities were contaminated by bacteria. Well depth, population density, age of the community, and soil and hydrologic characteristics were factors identified as influencing contamination of ground water by OSDSs.

Davis and Stephenson (1970) reported that 51% of 194 private water supplies in Bartow County, Georgia were contaminated with bacteria. They implicated OSDSs as the principal source of the contamination. A more detailed water quality survey of 760 wells from Wayne, Coweta, Bartow, and Mitchell Counties, Georgia indicated that forty-three samples contained greater than 10 mg/liter of nitrate-nitrogen, 300 samples contained coliform bacteria, and 114 contained fecal coliform. On-site sewage disposal systems were implicated as the principal source of the contamination (McDermott, 1971).

Contamination of rural water supplies by OSDSs constructed in shallow soils overlying fractured or jointed bedrock have been noted by Freethey (1969), Million (1970), and Waltz (1972). Seven of twenty-eight sites sampled by Freethey (1969) in north-central Colorado were contaminated by *E. coli* bacteria. Fifty-three of eighty-five sites sampled by Million (1970) in Larimer County, Colorado were also found to be contaminated by *E. coli* bacteria. Waltz (1972) indicated that the orientation of jointed bedrock surfaces affected the travel path of OSDS effluent and resulted in bacterial contamination of wells.

DeMichele (1974) sampled water quality in the Murderkill River of Kent County, Delaware over a six-year period prior to installation of a regional wastewater treatment plant. The fecal coliform concentration in stream samples ranged from 20 to 16,000 organisms/100 ml, with 45% of the samples having concentrations greater than 200 organisms/100 ml. Fecal streptococcus concentrations ranged from 10 to 11,300 organisms/100 ml, with 90% of all samples having concentrations greater than 200 organisms/100 ml. On-site sewage disposal systems discharging raw wastewater were identified as the source of the bacterial contamination.

Water quality samples from streams and ditches in rural Henry County, Indiana were found to be contaminated by fecal bacteria (Bowers, 1979). Fecal coliform counts as high as 3.9×10^6 MPN[1]/100 ml in stream and ditch samples were reported. Seventy-eight percent of the soils in the county were described as having severe limitations for conventional

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[1]MPN - Most probable number expresses a statistical "count" of the number or quantity of microorganisms over a given area or within a given volume (Romero, 1970).

OSDSs. Improperly functioning OSDSs were identified as the probable source of the contamination.

The North Carolina Division of Environmental Management conducted investigations along the coastal plain of southeastern North Carolina to assess the impact of OSDS density on water quality of tidal estuaries and their tributary fresh water creeks (Duda and Cromartie, 1982). Twenty-two sampling locations in six drainage basins were chosen where different OSDS densities were utilized. Within the drainage basins studied, 45 to 70% of OSDS sites had soils with severe limitations for conventional OSDSs. A significant positive correlation was found between levels of both fecal and total coliform bacteria in waters draining coastal residential areas and increased densities of OSDSs. Densities of more than 0.15 systems per acre resulted in shellfish bed closure due to mean coliform levels exceeding 100 MPN/100 ml.

In a study of selected coastal areas of Wakulla County, Florida, Williams et al. (1981, 1982) concluded that OSDSs were the principal source of elevated fecal coliform densities in surface water. Significant differences in fecal coliform densities in surface water were noted between developed and undeveloped areas. In the Spring Creek - Stuart Cove area, surface water of canals lacking residential development had fecal coliform densities of less than 35 organisms/100 ml. Surface water in canals adjacent to residential development had mean coliform densities ranging from 87 to 162 organisms/ml. Total coliform and fecal coliform densities as high as 22,908 and 5,128 MPN/100 ml, respectively, were sampled in the study area. Bacterial densities associated with naturally occurring fauna were demonstrated to be, at times, comparable to levels of human-induced pollution. High water tables and close proximity of OSDSs to surface waters were identified by Williams et al. (1981, 1982) as the primary causes of inadequate treatment by OSDSs. The contributions of urban runoff and surfacing of effluent from failing OSDSs to elevated coliform densities in surface waters were not identified.

The U.S. Environmental Protection Agency (1975) conducted tracer dye studies and monitored bacteriological quality of surface water at Punta Gorda and Big Pine Key, Florida, and at Atlantic Beach and Surf City, North Carolina to document movement of OSDS effluent into canal waters. The OSDSs were located approximately 50 ft (15.2 m) from the canals. The land elevation was approximately 5 ft (1.5 m) above mean sea level and building sites were created from fill material obtained from canal excavations. Rhodamine WT, a tracer dye, was added to septic tanks through house drains. An automatic water sampler located approximately 1 ft (30 cm) from the bottom of the canal sampled water downgradient from OSDSs. Additional water samples for bacterial analysis were collected 1 ft (30 cm) below the water surface. Dye was detected in canal water sampled 25 hours after it was introduced into a septic tank at Punta Gorda, Florida. Dye was not detected in canal water sampled over a 110 to 150 hour sampling period at two sites at Big Pine Key, Florida. At Atlantic Beach, North Carolina, dye was detected in canal water 4 hours and 60 hours, respectively, after introduction into two septic tanks. Results of the dye studies indicated that in several cases effluent was rapidly transmitted to canal waters.

Violations of applicable coliform standards were recorded in canal waters along housing developments utilizing OSDSs in both Florida and North Carolina (U.S. Environmental Protection Agency, 1975). Canal age and number of OSDSs along the canal correlated with coliform densities, as did location within the canal. Coliform densities were much higher at canal sampling stations than at background stations. Log mean total coliform densities in background samples at Punta Gorda, Florida were 203 colonies/100 ml. Log mean total coliform densities ranged from 436 to 871 colonies/100 ml in water samples from an undeveloped canal, and ranged from 176 to 1809 colonies/100 ml in water samples from a developed canal. Coliform densities in canal water at Big Pine Key, Florida were relatively low. Log mean total coliform densities in background water samples at Big Pine Key were less than 10 colonies/100 ml. Log mean total coliform densities in water samples were less than 10 colonies/100 ml for an undeveloped canal and ranged from 14 to 32 colonies/100 ml in water samples from a developed canal.

Sampling stations located at the end, middle, and mouth of a canal with a high density of OSDS installations at Atlantic Beach, North Carolina had log mean total coliform densities of 3400, 400, and 360 colonies/100 ml, respectively. OSDS effluent entering a canal system at Surf City, North Carolina, which was traced with dye introduced into the septic tank, was found to contain fecal coliform in excess of 2.4×10^6 colonies/100 ml.

The U. S. Environmental Protection Agency (1975) concluded that OSDSs may be viewed as an acceptable form of domestic waste treatment in the context of rural development where the quality of ground and surface water can be protected. In contrast, coastal canal development maximizes housing unit density and proximity to surface water bodies, and thus eliminates the safeguards inherent in rural environments.

3.5.4. Ground water monitoring studies

Crosby et al. (1968) determined the concentrations of total coliforms, fecal coliforms, and fecal enterococci in soil samples collected beneath a heavily loaded OSDS serving 100 occupants of a nursing home near Spokane, Washington. The site was located in glacial outwash sediments. Total coliform and fecal enterococcus counts in the soil were found to be quite high at a depth of 1 ft (30 cm) below the drainfield. Bacterial counts ranged from 9100 to 160,900 MPN/100 grams soil for total coliforms, 4900 to 49,000 MPN/100 grams soil for fecal coliforms, and 900 to 100,000 MPN/100 grams soil for enterococci.

Large reductions in bacterial counts were noted for soil samples collected 5 ft (1.5 m) beneath the drainfield. However, three of six samples contained appreciable quantities of fecal coliforms, and one sample contained enterococci. Total coliforms, fecal coliforms, and enterococci were detected in several soil samples from a depth of 21 ft (8 m) beneath the drainfield, but the authors suspected contamination of the samples during the sampling procedure.

Polkowski and Boyle (1970) monitored the coliform concentration of ground water adjacent to a large OSDS. The OSDS served a high school in Dane County, Wisconsin. Soil at the site was characterized as several feet of silty clay loam underlain by outwash sand and gravel. The water table was approximately 48 inches (122 cm) below the soil surface. Coliform counts for various ground water samples adjacent to the OSDS yielded erratic results. The coliform counts were zero, or very low and sporadic, for the majority of the wells. However, ground water samples from one monitoring well located approximately 100 ft (30 m) from the drainfield had coliform counts as high as 1170 organisms/100 ml. On one sampling date, fifteen of twenty-four monitoring wells detected quantities of coliform bacteria ranging from 24 to 730 organisms/100 ml. The authors concluded, nevertheless, that the site was relatively effective in removal of coliform bacteria from OSDS effluent.

Romero (1970) reviewed much of the early research on coliform movement from drainfields and pit privies. He indicated that coliform bacteria could be transported much farther when they directly entered the ground water than when they passed first through a water-unsaturated zone. Caldwell (1937, 1938) monitored movement of Bacillus coli from pit privies at three sites where privies were dug into the water table. Bacillus coli was transported distances of 10, 25, and 80 ft (3.0, 7.6 and 24.4 m) prior to the formation of a biological clogging mat or crust at the bottoms of the pit privies. In a latrine above the water table, no movement of bacteria was detected.

Wall and Weber (1970) reported total coliform concentrations above background levels in ground water samples from eleven of fourteen OSDS sites investigated near Canal Lake, Ontario, Canada. Soils at the sites ranged in texture from sand to clay loam and had water tables at depths ranging from 18 to 24 inches (46 to 60 cm) below the soil surface. Total coliform counts at the eleven OSDS sites ranged from 3.6×10^3 MPN/100 ml to 2.4×10^6 MPN/100 ml. The distance from the drainfield to the monitoring wells, and the depth of the wells, was not indicated.

Brandes (1972) monitored movement of indicator organisms in ground water from three OSDSs located at distances ranging from 27 to 53 ft (8.2 to 16.2 m) from Lake Chemong, Ontario, Canada. The OSDSs were constructed in sandy loam and silt loam fill materials containing stones and boulders. Depth to the water table ranged from 5 to 7 ft (1.5 to 2.1 m).

Effluent in the septic tank contained total and fecal coliform concentrations of 8 x 10^6 and 4.68 x 10^6 organisms/100 ml, respectively. Total and fecal coliform concentrations in ground water sampled 5 ft (1.5 m) from the absorption system contained 8 x 10^6 and 2.37 x 10^6 organisms/100 ml, respectively. At distances of 22 and 34 ft (6.7 and 10.4 m), total coliform concentrations were 6.4 x 10^5 and 1.15 x 10^5 organisms/100 ml, respectively. Fecal coliform concentrations were 1500 and 100 organisms/100 ml, respectively, at distances of 22 and 34 ft (6.7 and 10.4 m) from the absorption system. Brandes concluded that effluent treatment by the "loose fill material" was inadequate to protect the lake from bacterial contamination.

Vecchioli et al. (1972) monitored the movement of bacteria from municipal wastewater injected into the Magothy aquifer at Bay Park, Long Island, N.Y. Although the injected wastewater had substantial total coliform, fecal coliform, and fecal streptococcus densities, no fecal coliform and fecal streptococcus bacteria, and only nominal total coliform bacteria were detected in ground water sampled 20 ft (6 m) from the point of injection.

Allen and Morrison (1973) studied the movement of sewage-amended water and *Bacillus stearothermophilis*, a tracer bacterium, to evaluate the potential for ground water contamination from OSDS effluent in soils underlain by fractured crystalline bedrock. Direction and movement of effluent were controlled largely by the orientation of the major bedrock fracture sets. At one site the tracer bacterium was transported along bedrock fractures to a horizontal distance of 94 ft (28.7 m) in 24 to

30 hours. Similar studies using sewage amended water demonstrated that while fecal bacteria decreased slightly during percolation through bedrock fractures, total coliform densities were generally unchanged.

Converse et al. (1975) reported on the treatment of effluent applied to an OSDS mound using a pressure distribution system. Total coliform, fecal coliform, and fecal streptococcus concentrations of 2.8×10^6, 2.7×10^5, and 2×10^3 organisms/100 ml, respectively, in septic tank effluent were reduced to 2500, 310, and 45 organisms/100 ml, respectively, after passage through 2 ft (60 cm) of mound fill. Fecal streptococci and fecal coliforms were not detected in 22% and 33%, respectively, of the mound drainage samples. No indicator organisms were detected in the clay subsoil located 12 to 22 inches (30 to 55 cm) below the base of the mound after two years of operation. Unsaturated flow through 24 inches (60 cm) of mound fill and 12 to 22 inches (30 to 55 cm) of clay subsoil was effective in removal of indicator organisms from septic tank effluent.

Reneau et al. (1975) examined the bacteriological quality of surface and subsurface waters from a 198 acre (80 ha) watershed in Spotsylvania County, Virginia which contained improperly functioning OSDSs. Within the watershed, approximately seventeen percent of the OSDSs were located in soils suitable for proper OSDS functioning. Forty-one percent of the OSDSs were located in marginal soils which had periodic failures during periods of high soil moisture. Forty-two percent of the OSDSs were located in unsuitable soils which failed persistently throughout the year.

Subsurface water samples generally contained lower counts of total and fecal coliforms than surface water samples. There was also a trend of decreasing total and fecal coliform counts with distance from the drainfields. Surface waters from areas influenced by failing OSDSs had a large range in total and fecal coliforms. Sites not subjected to runoff had no detectable fecal coliforms in four of nine observations, whereas at sites where runoff occurred, total and fecal coliform densities were in excess of 2.4×10^7 MPN/100 ml. Most of the surface waters in the watershed were found to exceed the recommended bacteriological standard for surface waters used for recreation or drinking water.

Reneau and Pettry (1975a) monitored movement of total and fecal coliforms from OSDS effluent in three soils in Virginia considered to be marginally suited to OSDSs due to fluctuating seasonal water tables and/or restricting horizons. Total and fecal coliform counts above the plinthic horizon of a Varina soil were reduced from an average of 11×10^6 MPN/100 ml and 1.3×10^6 MPN/100 ml, respectively, in the distribution box, to 11×10^3 MPN/100 ml and 2.5×10^3 MPN/100 ml, respectively, at a distance of 20 ft (6.1 m) down-gradient from the drainfield. Within the plinthic horizon, the total coliform count decreased to 150 MPN/100 ml at a distance of 20 ft (6.1 m) downgradient. No fecal coliforms were detected below the plinthic horizon at any distance sampled.

The largest reduction in both total and fecal coliforms occurred during the initial 60 inches (1.5 m) of travel in a Goldsboro soil. The average total and fecal coliform counts decreased from approximately 4×10^6 NPM/100 ml in the drainline to 5×10^5 MPN/100 ml and 2×10^5 MPN/100 ml, respectively, at the 60 inch (1.5 m) depth. Fecal coliform counts decreased to less than 3 MPN/100 ml at a vertical distance of 13.8 ft (4.2 m) below the drainfield.

There were significant decreases in total and fecal coliform densities with distance from the drainfield both above and below the fragipan of a Beltsville soil. The fecal coliform counts were reduced from an average of approximately 15 x 10^3 MPN/100 ml near the drainfield to less than 300 MPN/100 ml at distances of 41 ft (12.5 m) and 92 ft (28 m) downgradient from the drainfield. The total and fecal coliform counts were greatly reduced below the fragipan as compared to the counts detected above the fragipan. In general, the total coliform counts below the fragipan were less than 400 MPN/100 ml, and no fecal coliforms were detected. The authors concluded that effluent movement was predominately in a horizontal direction above the restrictive horizons, and that only a limited possibility of contamination of the permanent ground water table would be expected at these sites.

Ziebell et al. (1975a) determined the concentration of indicator bacteria in the soil below an OSDS drainfield located in a sandy soil in Wisconsin. The indicator organisms were found to decrease to very low numbers after travel through a relatively short distance of unsaturated soil. The fecal streptococcus, fecal coliform, and total coliform concentrations in the biological clogging layer or crust zone were 5.4 x 10^4, 4.6 x 10^6, and 2.3 x 10^7 organisms/100 grams soil, respectively. At a depth of 3 inches (8 cm) below the base of the trench and at a lateral distance of 12 inches (30 cm) from the trench, the fecal streptococcus, fecal coliform, and total coliform concentrations were less than 2.0 x 10^2, 1.7 x 10^4, and 2.3 x 10^4 organisms/100 grams, respectively. Twelve inches (30 cm) below the base of the trench, the fecal streptococcus, fecal coliform, and total coliform concentrations were reduced to less than 200, 200, and 600 organisms/100 grams, respectively.

Ziebell et al. (1975b) studied indicator organisms in outflow from the base of a 24 inch (60 cm)-thick mound fill. Twenty-two percent, thirty-three percent, and six percent of the samples tested were negative for the presence of fecal streptococci, fecal coliforms, and total coliforms, respectively. The detection of Pseudomonas aeruginosa in two samples confirmed the presence of pathogens in the outflow. Additional treatment of the effluent by the natural soil was necessary for further purification.

Missimer Associates Inc. (1976) examined the bacteriological quality of ground water adjacent to two OSDS sites located on Sanibel Island, Florida. Fecal streptococci were detected in only two of eight ground water monitoring wells, at concentrations of 4.6×10^3 and 5.0×10^3 organisms/100 ml at one site adjacent to West Rock Lake. One of the wells in which fecal streptococci were detected was located upgradient of the OSDS. Fecal coliforms were not detected in any of the ground water monitoring wells, but total coliform concentrations detected in six of the eight wells ranged from 1.6×10^3 to 1.4×10^4 organisms/100 ml. Two ground water monitoring wells located upgradient from a second OSDS failed to detect total coliforms or fecal coliforms in ground water adjacent to the system. Fecal coliform counts of 100 organisms/100 ml were detected in ground water adjacent to the OSDS. The authors concluded that bacterial analyses of ground water at the two sites were inconclusive with regard to bacterial contamination of ground water from OSDSs.

Viraraghavan and Warnock (1976a,b,c) determined the bacteriological character of ground water beneath an OSDS drainfield in Ottawa, Ontario, Canada. Considerable reductions in the concentration of indicator organisms were noted in the effluent after treatment by the soil surrounding the drainfield. However, relatively high concentrations of indicator organisms were detected in ground water below the drainfield. The concentrations of total coliforms, fecal coliforms, and fecal streptococci in ground water were 5.4×10^5 organisms/100 ml, 6.6×10^2 organisms/100 ml, and 1.3×10^5 organisms/100 ml, respectively. The concentration of Pseudomonas aeruginosa was 2 organisms/100 ml. The concentrations of total coliforms, fecal coliforms, and fecal sprepto-cocci in ground water increased with time as loading continued, and decreased when loading was discontinued. The depth from which ground water samples were collected below the drainfield was not indicated.

Hagedorn et al. (1978) used antibiotic-resistant fecal bacteria to monitor the movement of enteric bacteria discharged into a simulated drainfield under saturated soil conditions in Oregon. The soil at the site was somewhat poorly drained and the soil texture below the simulated drainfield ranged from silty clay loam to clay loam. Streptomycin-resistant Streptococcus faecalis and Escherichia coli isolated from raw sewage were added to gravel-packed pits constructed to simulate drainfields.

The bacteria survived in appreciable numbers throughout two thirty-two-day sampling periods, but the numbers of bacteria decreased with time after inoculation and with distance from the inoculation pit. At both sites, the inoculated bacteria were continually detected in ground water sampled 9.8 ft (3 m), 16.4 ft (5 m), and 49 ft (15 m) from the inoculation pit. The populations of _S. faecalis_ and _E. coli_ in the various ground water monitoring wells reached maxima during intervals closely associated with the rise of the water table following major rainfall periods. Bacteria were not detected in ground water sampled 98 ft (30 m) from the simulated drainfield. The authors concluded that thirty-two days was insufficient time for movement of bacteria to the 98 ft (30 m) monitoring wells.

McCoy and Hagedorn (1979) monitored the survival and movement of three isolates of _E. coli_ in a moderately well-drained soil in Oregon. The three isolates of _E. Coli_ were labeled by their resistence to sodium azide and separately to novobiocin, nalidixic acid, and tetracycline. The site was located on a hillslope and the water table was between 22 and 34 inches (55 to 85 cm) below the soil surface. A clay-textured subsoil at a depth of 14 to 24 inches (35 to 60 cm) was underlain by saprolite (highly weathered bedrock) which graded into $CaCO_3$-cemented sandstone at a depth of 39 inches (100 cm). The _E. coli_ were introduced into the soil by three perforated horizontal lines located at depths which corresponded to the A, B, and C horizons. The resistance-labeled _E. coli_ population in the soil environment was found to drop below detection limits within 96 hours of introduction. The maximum temporal concentration of _E. coli_ in the ground water decreased with distance downslope from the injection line. Initially, large reductions in bacterial numbers were noted as the bacterial population entered the soil. However, once the organisms reached a highly conductive zone, relatively long distances were necessary for the further reduction of bacterial densities. _E. coli_ counts exceeding 10^4 cells/ml and 10^2 cells/ml were detected in monitoring wells located 34.5 ft (10.5 m) and

67 ft (20.4 m), respectively, from the injection line. McCoy and Hagedorn concluded that effluent-borne pathogens in these circumstances have the opportunity for rapid transport either horizontally to surface receiving waters or vertically to aquifers through saturated recharge pathways.

McCoy and Hagedorn (1980) monitored the movement of resistance-labeled _E. coli_ through a saturated soil profile which represented a transition between the Steiwer and Hazelair soil series in Oregon. The site was located on the concave footslope segment of the hillslope. Three separately distinquishable _E. coli_ strains (McCoy and Hagedorn, 1979) were introduced into three perforated horizontal lines installed upslope from the transition soil and positioned in the A, B and C horizons of the soil. Bacterial translocation patterns varied with depth of _E. coli_ introduction in the monitoring wells sampled 8.2 ft (2.5 m) downslope. However, as the bacteria were transported downslope, flow pathways converged and differences due to introduction depth disappeared. Surfacing of bacteria to the upper level of the zone of saturation was noted in monitoring wells located 16.4 ft (5 m) downslope. Transitory channel flow or pipe flow under saturated conditions transported _E. coli_ in a manner which bypassed interaction with the soil matrix. _E. coli_ concentrations in excess of 10^3 cells/ml were detected in outflow from a rodent channel which surfaced 33 ft (10 m) downslope from the introduction line.

Rahe et al. (1978) studied the movement of an antibiotic-resistant strain of _E. coli_ in two soils with artificially maintained high water tables which were located on a hillslope in Oregon. The _E. coli_ was introduced into a Dixonville silty clay loam soil and Hazelair silt loam soil by injection lines located at depths which corresponded to the A, B and C horizons of the two soils. Transport of *E. coli* differed at both sites with respect to movement rates, horizons in the soil profile through which major translocation occurred, and the relative number of bacteria transported over time. The bacteria were transported much more slowly and in greatly lowered numbers in the Hazelair soil as compared to the Dixonville soil, and the portion of the profile through which the greatest movement occurred was also different. Translocation of the *E. coli* at the Hazelair site was restricted predominately to the 5 inch (12 cm) and 12 inch (30 cm) depths, while the *E. coli* were transported at the 18 inch (45 cm), 32 inch (80 cm), and 43 inch (110 cm) depths in the Dixonville site. *E. coli* injected into the A and B horizons of the Dixonville soil were recovered in detectable numbers after two hours in monitoring wells located 49 ft (15 m) from the point of injection. *E. coli* injected into the A horizon of the Hazelair soil were recovered in detectable numbers after twelve hours in monitoring wells located 66 ft (20 m) from the point of injection. The rapid movement of bacteria through saturated soil was associated with high rates of water flow through macropores in these hillside soils.

Harkin et al. (1979) monitored performance of 33 randomly selected mound systems in Wisconsin. Twenty systems were located on sites with seasonally high water tables, six systems were on sites with shallow depth to bedrock, and seven systems were on sites having slowly permeable soils with or without high water tables. The depth to the water table was not indicated.

The mound systems at sites with shallow depth to bedrock were constructed with 2 ft (60 cm) of mound fill and the remaining systems were constructed using 1 ft (30 cm) of mound fill. The average uniformity coefficient of the mound fill was 5.7 and ranged from 1.8 to more than 100. Uniformity coefficient, a measure of filter sand, is the quotient obtained by dividing the maximum diameter of 60% (by weight) of the soil particles by the maximum diameter of 10% (by weight) of the soil particles (Spangler and Handy, 1973). Texture and particle size analysis of the mound fill were indicated for only fourteen mound systems. Twelve systems had sand-textured mound fills with contents of silt plus clay that ranged from 1.9 to 8.4% by weight. Two mound systems were constructed with loamy sand-textured mound fill. The effluent dose rate averaged 0.74 gal/ft^2/dose and ranged from 0.14 to 1.05 gal/ft^2/dose.

Averaged over all mound systems, total coliform and fecal coliform concentrations were reduced by 99.5% and 99.9%, respectively, before the effluent reached the natural soil. Coliform travel into the natural soil was minimal. Only one system, located on a coarse-textured soil with a high water table, had an appreciable concentration of coliforms in the natural soil. In general, the mound fill was less effective in reduction of fecal streptococci and _Pseudomonas aeruginosa_. Movement of fecal streptococci and _P. aeruginosa_ in mound fill was related to the small size of fecal streptococci (i.e., 0.6 to 1.0 μm diameter), the mobility of _P. aeruginosa_, and high effluent loading rates (i.e., 0.14 to 1.05 gal/ft^2/dose). High concentrations of fecal streptococci and _P. aeruginosa_ at the natural soil surface were thought to result from filtration or regrowth. Mound systems with low dosing rates, and fill material with low uniformity and finer texture, most effectively reduced the bacterial concentration of effluent before the effluent reached the natural soil. Mound systems employing gravity distribution systems or high rate pressure dosed distribution systems, highly uniform fill materials, and coarse-textured materials may not be effective in treatment of septic tank effluent to prevent ground water contamination when the underlying natural soil is coarse-textured and has a high water table.

Harkin et al. (1979) monitored ground water quality adjacent to four of the mound systems over a four-month period. The mounds were constructed in soils ranging in texture from loamy sand to silt loam, with shallow permanent water tables. The exact depth to the water table was not indicated. Ground water monitoring wells were located at minimum distances of 10 to 31.5 ft (3 to 9.6 m) and maximum distances of 55 to 60.5 ft (16.8 to 18.5 m) from the absorption bed. The minimum distance corresponded to the toe of the mound fill. Each site had only 3 to 5 wells, which sampled only the upper 12 inches (30 cm) of the ground water.

None of the four systems monitored showed contamination that could be associated with significant transfer of bacteria from effluent to ground water. Ground water had total and fecal coliform concentrations ranging from less than 2 to less than 20 MPN/100 ml, fecal streptococcus concentrations ranging from 0 to less than 100 MPN/100 ml, and _P. aeruginosa_ concentrations ranging from 0 to less than 20 MPN/100 ml. Harkin et al. (1979) noted that detection limits of the fecal streptococcus analyses were only sensitive enough to report fecal streptococcus concentrations as less than 100 MPN/100 ml.

Peavy and Groves (1978) and Peavy and Brawner (1979) monitored fecal coliform concentrations in the upper 12 inches (30 cm) of the shallow ground water adjacent to an OSDS in Montana. The OSDS was located in a soil with a water table approximately 48 inches (122 cm) below the soil surface. Fecal coliform counts in ground water adjacent to the OSDS were quite erratic. [Similar results have been reported by Carlile et al. (1981) and Cogger and Carlile (1984).] The only ground water location which had a positive count for more than fifty percent of the sample tests was located just under the drainfield. The occurrence of positive tests did not follow a predictable pattern. The authors concluded that the infrequent occurrence and low number of organisms indicated a lack of significant movement of fecal coliforms with ground water at this site.

Reneau (1978) studied the effect of artificial drainage on movement of coliform bacteria from three OSDSs into somewhat poorly to very poorly drained soils in Virginia. The drainage tile line was located at distances ranging from 38 to 63 ft (11.6 to 19.2 m) from the three OSDSs. Total and fecal coliform counts during periods of high water table decreased rapidly in ground water as effluent moved from the disposal areas toward the artificial drainage system. The decrease in total and fecal coliform counts, as a function of distance from the drainfield, could be estimated by logarithmic equations. Fecal coliform densities were approximately 10^5 MPN/100 ml in ground water adjacent to the drainfields but decreased to 10 to 1000 MPN/100 ml in ground water sampled 5 ft (1.5 m) from the drainage tile. The water intercepted by the drainage tile contained elevated total and fecal coliform densities. Water intercepted by the drainage tile immediately downgradient from the drainfields ranged in coliform counts from 290 to 870 MPN/100 ml. Dilution reduced the fecal coliform count in the outfall to less than

200 MPN/100 ml. The fecal coliform count in control wells was less than 3 MPN/100 ml. The author noted that while the drainage outflow water would meet the bacteriological standards of the Virginia State Water Control Board for Subclass B Waters, increasing OSDS density would result in unacceptable levels of total and fecal coliforms in tile drain outlets or ditches.

Viraraghavan (1978) monitored the transport of indicator microorganisms downslope from a drainfield distribution line in Ottawa, Ontario, Canada. The seasonally high water table was at a depth of 6 inches (15 cm) below the drainfield distribution line during the ground water sampling period. A general decline in the levels of total coliforms, fecal coliforms, fecal streptococci, and standard plate count were noted in ground water sampled at increasing distances away from the drainfield. However, the concentration of microorganisms in the ground water was not reduced to acceptable levels even in ground water sampled 50 ft (15.3 m) from the drainfield. The author concluded that insufficient unsaturated flow occurred to adequately treat the OSDS effluent at this site.

Carlile et al. (1981) and Cogger and Carlile (1984) evaluated the performance of fifteen OSDSs located in soils with seasonally high water tables. Thirteen of the fifteen systems examined were at least seasonally saturated by high water tables. The remaining two sites had water tables at least 12 inches (30 cm) below the bottom of the absorption system. Effluent treatment was poorest around systems that were continuously saturated with ground water. An average fecal coliform concentration of 1700 MPN/100 ml was detected in ground water monitoring wells located 5 ft (1.5 m) from the drainfields. Systems that were only seasonally saturated had average fecal coliform concentrations of 560 MPN/100 ml in ground water sampled 5 ft (1.5 m) from the drainfield. Under continuously saturated soil conditions, median fecal coliform concentrations of approximately 175 MPN/100 ml in ground water sampled 5 ft (1.5 m) from the drainfields decreased to approximately 40 MPN/ 100 ml in ground water sampled 25 ft (7.6 m) from the drainfields. Under seasonally saturated soil conditions, median fecal coliform concentrations decreased from approximately 55 MPN/100 ml in ground water sampled 5 ft (1.5 m) from the drainfields to approximately 15 MPN/100 ml in ground water sampled 25 ft (7.6 m) from the drainfields.

Stewart and Reneau (1981) examined the spatial and temporal variation in fecal coliform densities within and around two OSDSs located in soils having a high seasonal water table. Fecal coliform movement at both sites was almost entirely limited to periods when the water table was at or above the drainfield trench bottoms. At one site, the water table gradient was influenced by a drainage ditch approximately 70 ft (21.4 m) from the drainfield. The movement of fecal coliform bacteria was primarily in the horizontal direction toward the drainage ditch, with limited penetration of organisms to a depth of 10 ft (3.1 m). The reduced downward movement of bacteria was attributed to the slower permeability and increased filtering capacity of a relatively fine-textured subsoil and to the water table gradient created by the drainage ditch. The average concentration of fecal coliforms in ground water sampled 10 ft (3.1 m) beneath the drainfield was less than 3 MPN/100 ml. Elevated fecal coliform concentrations near the drainfield decreased to less than 5 MPN/100 ml at a lateral distance of 33 ft (10 m) in the direction of ground water flow.

Bacterial movement away from the drainfield at the second OSDS site was both horizontal and vertical. The absence of a measureable water table gradient and of higher fecal coliform densities in ground water sampled in deep wells (10 ft, 3.0 m) compared with that of shallow wells (5 ft, 1.5 m) indicated that the predominant direction of flow was vertical. The average fecal coliform concentration in ground water sampled 10 ft (3 m) below the drainfield was 10 MPN/100 ml, but on several occassions extremely high fecal coliform concentrations were detected. The extent of lateral movement of fecal coliforms was 16 to 20 ft (5 to 6 m) from the drainfield. The authors concluded that effective drainage created a water table gradient at one location and resulted in horizontal flow of effluent, whereas the lack of a discernable water table gradient at another location resulted in downward flow and subsequent contamination of the shallow ground water.

Wilson et al. (1982) monitored the movement of effluent from eight OSDSs located in moderately well to somewhat poorly drained soils that were artificially drained by a perimeter drainage system. Tile drains located 20 ft (6 m) from the drainfield trenches were installed 6 ft (1.8 m) below the soil surface. The 6 ft (1.8 m)-deep tile was effective in lowering the water table elevation approximately 2.5 ft (0.8 m) below the disposal trench. Discharge from the perimeter tile outlet had a wide variation in total and fecal coliform concentrations during the monitoring period. Total coliform concentrations in the perimeter drain outlet ranged from 470 to 2380 organisms/100 ml, with a weighted mean total coliform concentration of 1468 organisms/100 ml. The fecal coliform concentration in the perimeter drain outlet ranged from 47 to 484 organisms/100 ml, with a weighted mean fecal coliform concentration of 202 organisms/100 ml.

3.5.5. Lysimeter, sand filter, and column studies

Brown et al. (1978b, 1979) monitored movement of fecal coliform bacteria and bacterial viruses from a drainfield distribution line installed in field lysimeters in Texas. During a three-year period, leachate was collected at the bottom of the lysimeters by a series of ceramic cups maintained at a potential equivalent to -0.8 bar. During the first year of effluent application, only 23 of 421 leachate samples tested positive for the presence of fecal coliforms. The majority of the positive tests occurred very near the beginning of the study and were thought to have resulted from contamination of the samples. Composite leachate samples collected during the second year of effluent application tested positive for the presence of fecal coliforms in only 3 of 133 samples. Fecal coliforms in the soil were concentrated primarily at the gravel-soil interface below the drainfield distribution line in Miller clay. In Norwood sandy clay, fecal coliform counts as high as 750 organisms/gram of soil were detected at the gravel-soil interface. Fecal coliforms were also detected in the soil beside the drainfield and to a depth of 35 inches (90 cm) below the drainfield tile line. The discontinuous distribution of fecal coliforms around the drainfield was related to movement of coliforms along root channels and cracks or voids in the soil.

In only 2 of 230 samples were coliphages detected in the leachate from the lysimeters after one year of effluent application. The presence of coliphages in two leachate samples was detected only after the effluent was enriched with a concentration of E. coli f_2 bacteriophage equivalent to one thousand times the concentration normally found in the septic tank effluent. At the time that the bacteriophage-enriched effluent was applied to the lysimeter containing Miller clay, the soil was dry and had large cracks. The coliphages detected in the leachate may have moved along cracks or root channels rather than through intrapedal pores in the Miller clay. Coliphages were detected at the soil-gravel interface below the drainfield distribution line in greater concentrations at a depth of 12 inches (30 cm) below the drainfield distribution line in Norwood sandy clay than in the other soils. In addition, a few coliphages were detected in soil samples collected 40 inches (100 cm) below the drainfield tile line. Survival of coliphages in the soil was limited to three or four weeks after effluent application ceased. No difference in survival or mobility of coliphages was found between the soils studied. Brown et al. (1978b, 1979) concluded that under water-unsaturated flow conditions, 48 inches (120 cm) of any of the soils tested appeared to be sufficient to minimize the possibility of ground water contamination by fecal coliforms or coliphages from OSDS effluent.

Brandes et al. (1975) determined the total and fecal coliform concentrations in effluent outflow from ten underdrained filter beds. Five filter beds, 30 inches (76 cm) in depth, were constructed from coarse-textured materials ranging from fine sand to pebbles. In addition, five filter beds were constructed using fine sand to which varying amounts of limestone, "clayey silt," or red mud had been added. As the sand size in the filter bed decreased, total and fecal coliform counts in the outflow decreased. Total and fecal coliform counts in the effluent outflow from the bed of pebble-sized filter material were 1.013×10^7 and 1.19×10^6 organisms/100 ml, respectively. Total and fecal coliform counts in the fine sand filter bed outflow were 2.338×10^6, and 3.3×10^4 organisms/100 ml, respectively. Additions of limestone, "clayey silt," and red mud to the fine sand resulted in even greater reductions in coliform bacteria counts of the outflow. Total coliform counts in the outflow were reduced to less than 2.5×10^4 organisms/100 ml for three of the five amended filter beds. Fecal coliform counts were reduced to less than 1×10^4 organisms/100 ml in outflow from three of the five filter beds.

Otis et al. (1975) reported on the bacteriological quality of effluent from septic tank and aerobic treatment units after passage through intermittent sand filters. The sand filters consisted of parallel beds with 24 inches (60 cm) of medium sand overlying 8 inches (20 cm) of gravel. A mean fecal coliform concentration of 2310 organisms/ml in the septic tank effluent was reduced to a mean concentration of 111 organisms/ml in the outflow from one filter bed and was reduced to 42 organisms/ml in the outflow from the adjoining filter bed. A fecal streptococcus concentration of 20.7 organisms/ml in the septic tank effluent was reduced to mean concentrations of 4.8 organisms/ml and less than 1 organism/ml in the filter beds.

For the aerobic treatment unit, fecal coliform and fecal streptococcus concentrations of the effluent were 26 organisms/ml and 144 organisms/ml, respectively. Sand filter treatment of the aerobic effluent resulted in a decrease in the fecal coliform concentration of the outflow from one filter bed and an increase in the fecal coliform concentration of the outflow from the other filter bed. Fecal coliform concentrations in the outflow after sand-filter treatment were 10.7 and 30.6 organisms/ml. Fecal streptococcus concentrations in the outflow from both sand filters following aerobic treatment decreased to 2.6 and 7.3 organisms/ml.

Brandes (1980) monitored the concentration of total and fecal coliform organisms in effluent passing though a 3.3 ft (1 m)-thick underdrained sand filter during a two-year study. The average concentrations of total and fecal coliforms in the septic tank effluent were 3.7×10^7 organisms/100 ml and 7.4×10^5 organisms/100 ml, respectively. Effluent collected beneath the sand filter contained total and fecal coliforms counts of 428 and 25 organisms/100 ml, respectively. Brandes indicated that the low level of coliform organisms in the outflowing effluent was related to the presence of limestone gravel surrounding the distribution system. In a previous study, Brandes et al. (1975) noted that calcium in the form of limestone, when added to sand filters, reduced the concentration of coliform organisms in outflow from sand filters by enhancing adsorption and increasing die-off of coliform organisms.

Willman (1979) and Willman et al. (1981) examined the removal of fecal bacteria from effluent by soil columns. Three mineralogically different sand fractions derived from limestone, sandstone, and shale were mixed with varying amounts of clay to produce soil columns which ranged in clay content from 0 to 12%. A reduction of fecal indicator organisms from approximately 10^6 organisms/ml to less than 10^3 organisms/ml was obtained after passage through 24 inches (60 cm) of the various sand-clay mixtures. In columns containing no added clay, the sand derived from limestone and shale removed the largest number of both fecal coliforms and fecal streptococci. However, for columns containing 6 or 12% clay, the sand derived from sandstone removed the largest number of fecal coliforms. The authors concluded that sand type and clay content influenced the removal of fecal bacteria from effluent.

Stewart el al. (1979) did not detect fecal coliforms in column outflow after passage through 37 inches (90 cm) of aerobic, water-unsaturated soil. Soil columns of sand and loamy sand were dosed twice daily for 220 days with 0.65 inch (1.65 cm) of effluent containing 6.9×10^4 organisms/100 ml. Analysis of soil samples from the columns revealed no penetration of fecal coliforms beyond a depth of 12 inches (30 cm) from the point of effluent application.

3.5.6. Summary

Indicator organisms are commonly enumerated in effluent and ground water because the task of detecting all types of bacteria is complex and costly. Counts of total coliforms, fecal coliforms, and fecal streptococci are thought to reflect the presence of human pathogens in effluent and ground water (Berg, 1978).

The concentration of indicator organisms can be quite variable. Ziebell et al. (1975a) reported total coliform concentrations of 3.4×10^6 organisms/100 ml, fecal coliform concentrations of 4.2×10^5 organisms/100 ml, and fecal streptococcus concentrations of 3.8×10^3 organisms/100 ml in septic tank effluent.

Fecal bacteria are removed from effluent in soil by the mechanisms of filtration, sedimentation, adsorption, and natural die-off. The biological clogging mat or crust that commonly forms within the first few inches of the soil below an absorption system has been found to be an effective barrier to bacterial transport (Bouma et al., 1972). The removal of indicator organisms from effluent is also a function of the soil water/effluent flow regime. Transport of indicator organisms under water-unsaturated flow conditions is generally restricted to about 3.3 ft (1 m) (Brown et al., 1979; Hagedorn and McCoy, 1979). Movement of indicator organisms over much longer distances has been reported under water-saturated flow conditions (Romero, 1970; Viraraghavan, 1978).

Several ground water monitoring studies have reported contamination of ground water originating from OSDSs under conditions of saturated flow, high effluent loading rates, and shallow depth to seasonal high water tables or fractured or jointed bedrock.

Bacterial contamination of water wells by OSDSs is the second most common reason for well replacement in the southeastern United States (Miller and Scalf, 1974; Miller et al., 1977).

3.6. Viruses

3.6.1. Fate and transport

Viruses are obligate intracellular parasites ranging in size from 0.02 to 0.26 microns. In comparison, the smallest bacterium is approximately 0.30 microns in size (Cookson, 1974). Viruses behave as electrically charged collodial particles in the soil environment (Sproul, 1974, 1975; Bitton, 1975). Most viruses have a protein coat which surrounds the nucleic acid, the infectious part of the virus. The reactions of viruses in the soil environment are characteristic of the protein in the coat (Sproul, 1975).

Viruses and other pathogens are not part of the normal fecal flora. They occur in effluent in varying numbers that reflect the combined infection and carrier status of the residents utilizing the OSDS (Berg, 1973). As such, concentration of viruses can be quite variable. Clark et al. (1964) estimated that the concentration of viruses in domestic wastewater was 7000 plaque forming units (PFU)/liter. Berg (1973) reported recoveries of 32 to 107 PFU/liter in domestic wastewater from several midwestern states. Vaughn and Landry (1977) reported virus concentrations in raw wastewater entering a septic tank ranging from 0 to 2365 PFU/liter in Upton, New York. Yeager and O'Brien (1977) reported a virus concentration of 2500 PFU/ml in a septic tank, and Hain and O'Brien (1979) reported that virus concentrations ranged from 1600 to 3700 PFU/liter in three septic tanks sampled in New Mexico. The number of viruses that constitute a disease-producing dose varies, but it has been shown that one PFU is capable of producing human infection (Katz and Plotkin, 1967).

Virus removal or inactivation in the soil may be effected by any of the mechanisms of filtration, precipitation, adsorption, biological enzyme attack, and natural die-off (Sproul, 1974, 1975; Gerba et al., 1975). The small size of the virus, and its surface properties, suggest that removal from effluent is primarily through adsorption to soil particles, rather than through filtration as with bacteria (Scheuerman et al., 1979).

Many of the soil properties that affect sorption of bacteria also affect adsorption of viruses. Drewry and Eliassen (1968) and Drewry (1973) found that virus adsorption generally increases with increasing cation exchange capacity, clay content, silt content, and glycerol retention capacity. These results indicate that virus adsorption may be closely related to specific surface area of the soil (Lance, 1978). Drewry and Eliassen (1968), Gerba et al. (1975), and Bitton et al. (1979) have indicated that sandy soils retain fewer viruses than fine-textured soils. However, the pH, ionic composition of the soil solution, clay mineral species, and presence of other chemical or organic constituents may overide the effect of surface area in some soils (Lance, 1978). Hydrogen ion concentration (pH), for example, has a strong influence on virus stability as well as on adsorption and elution. Generally, low pH favors virus adsorption while high pH results in decreased adsorption or increased elution of adsorbed viruses (Drewry and Eliassen, 1968; Gerba et al., 1975).

Carlson et al. (1968) and Lefler and Kott (1974) showed that adsorption increased as the concentration of cations increased. Maximum adsorption was ten times greater with divalent cations present than with monovalent cations present at the same solution concentration. Carlson et al. (1968) also noted that desorption of viruses occurs when the ionic concentration of the soil solution is lowered. Duboise et al. (1976) and Wellings et al. (1975a) have demonstrated that rainwater has the ability to desorb viruses.

Carlson et al. (1968), Jakubowski (1969), and Young and Burbank (1973) have demonstrated that clay mineral species vary in their ability to adsorb and retain viruses. Carlson et al. (1968) found that, under similar ionic concentrations, kaolinite and montmorillinite adsorbed more viruses than did illite.

Drewry (1973) reported that virus adsorption capacities decreased as organic content increased, and Scheuerman et al. (1979) indicated that humic and fulvic acids in soil solutions interfered with virus adsorption. Bitton (1975) reported that iron oxides are effective in virus adsorption.

The mechanism of adsorption of viruses by soil has been studied by Cookson (1967, 1969) and Cookson and North (1967) using bacteriophage T_4 as a model. They indicated that the adsorption process was reversible, obeyed the Langmuir isotherm, and was probably diffusion limited. Additional studies by Drewry and Eliassen (1968) and Burge and Enkiri (1978) have indicated that adsorption of bacteriophage to soils conforms to the Freundlich isotherm, which differs from the Langmuir isotherm in that there is no upper boundary constraint on amount adsorbed.

Additional factors affecting the removal of viruses from OSDS effluent include the virus concentration, adsorption characteristics of the various viruses, temperature, soil moisture regime, and loading rate.

Viruses vary widely in size, shape, structure, and probable iso-electric point (Bitton et al., 1979). Goyal and Gerba (1979) indicated that virus retention in soils varies with virus type. Viruses which have different proteins in their protein coat have different adsorption characteristics (Sproul, 1975). Hori et al. (1970) noted a lesser adsorption of poliovirus type 2 than that obtained by Tanimoto et al. (1968) for a T_4 bacteriophage under similar adsorption conditions for two soils in Hawaii. Bacteriophages frequently have more complicated structures than animal viruses, and could be expected to have different adsorption characteristics (Sproul, 1975).

The removal of viruses from percolating effluent has also been shown to be a function of soil moisture content and flow regime. Rapid movement during saturated flow results in short travel times, limited soil-solution contact, and poor retention (Bouma et al., 1972; Bouma, 1979). Green and Cliver (1975) reported increased removal of viruses in sand columns under unsaturated flow conditions compared to saturated flow conditions. Robeck et al. (1962) found that increasing the flow rate through sand columns decreased the percentage of viruses retained by the sand. Lance et al. (1976) found that increasing the flow rate through loamy sand did not affect virus adsorption until a critical breakthrough rate was reached.

Monitoring of viruses under field conditions has been accomplished through the development of relatively sophisticated and sensitive techniques capable of detecting low numbers of viruses in large volumes of ground water and in soil samples (Bitton et al., 1979).

3.6.2. Ground water monitoring studies

Parsons (1973) added laboratory cultured Poliovirus type 1 to a septic tank in Dade County, Florida to determine if enteric viruses could survive conditions in the septic tank and be transmitted by effluent to the ground water. Poliovirus was not recovered in the effluent from the septic tank fifteen days after inoculation, indicating die-off of the seeded poliovirus or reductions in concentration such that the recovery technique could not detect the viruses. During a six-month investigation, no viruses were detected in ground water from a limited number of monitoring wells at five sites.

Wellings et al. (1975b) indicated that attenuated or laboratory adapted virus strains have differing sensitivities to various physio-chemical agents than do so-called wild strains (i.e., those excreted by man) and may succumb quickly when exposed to septic tank environments. Wellings et al. (1975b) evaluated critically the study by Parsons (1973) and concluded that use of free polio vaccine virus, which had been attenuated, and small sample size (0.1% of the septic tank effluent volume) may have resulted in the lack of virus detection in the effluent and ground water.

Vaughn and Landry (1977) monitored virus concentrations in raw wastewater and in septic tank effluent from an OSDS installed at an apartment complex of the Brookhaven National Laboratory, Upton, New York. Virus concentrations in raw wastewater sampled monthly over an eleven month period ranged from 0 to 2365 PFU/liter. Viruses were isolated in the septic tank effluent on only one occasion, and the virus concentration in that instance was 2.6 PFU/liter.

Harkin et al. (1979) monitored performance of 33 randomly selected mounds in Wisconsin. Twenty systems were on sites with seasonally high water tables, six systems were on sites with shallow depth to bedrock, and seven systems were on sites with slowly permeable soils with and without high water tables. Over a nine-month period, analyses of 78 effluent samples from septic tanks showed no evidence of viruses. Thereafter, samples for viral analysis were taken only from systems which served families with young children. Viruses were detected in only one septic tank during the two-year monitoring program, recurring in three samples collected over a four-month period. The virus was traced to a child in the family who was being immunized against poliomyelitis with oral Sabin vaccine. No virus was detected in the soil below the system.

Hain and O'Brien (1979) studied the survival of enteric viruses in septic tanks and the survival and movement of enteric viruses in ground water and soils surrounding OSDS drainfields near Las Cruces, New Mexico. Viable enteric viruses were isolated from four properly functioning septic tanks. The number of viruses isolated ranged from 1600 to 3700 PFU/liter. No attempt was made to identify the viruses isolated, but different plaque morphologies suggested at least four types were present.

Poliovirus type 1, often used as a model for enteroviruses, was introduced into a septic tank and remained viable in contact with septic tank effluent for longer than the 24 hours necessary to pass through a septic tank and into an OSDS drainfield. Fifty percent of the poliovirus type 1 introduced into the septic tank was inactivated after 24 hours, and 95% was inactivated after 20 days. Once introduced into the drainfield, poliovirus remained viable for extended periods of time and moved with ground water flow. Ground water samples which tested positive for the presence of coliform bacteria were concentrated and assayed for virus. Viruses capable of infecting Hela cell monolayers were isolated from each coliform-positive ground water sample. The concentrations of viruses ranged from 1 to 7 PFU/240 ml for the ground water samples. In contact with the ground water, poliovirus was rapidly inactivated, but contact with soil seemed to have a protective effect on virus activity. The rate of inactivation of poliovirus was more rapid in the ground water environment than in the septic tank. Virus counts were reduced approximately 90% in 24 hours in ground water-viral survival experiments. Infective viruses were recovered after more than 30 days from moist soil cores kept in the laboratory at room temperature.

Data for poliovirus is often assumed to be representative of all viruses. However, Yeager and O'Brien (1977) have shown that Coxsackievirus B-1 has survived longer in septic tanks than poliovirus, and Bertucci et al. (1977) reported that Echovirus 11 is less sensitive than poliovirus to inactivation during anaerobic sludge digestion.

Stramer and Cliver (1981) monitored the survival and movement of Poliovirus type 1 added to septic tanks. The OSDSs were located near a lake in Wisconsin. As much as 99.9 to 99.99% of the initial Poliovirus type 1 concentration added to the septic tanks was inactivated in the septic tanks. However, enough viruses migrated to the drainfields to be detected in ground water monitoring wells and in the lake.

The survival and movement of Bovine Enterovirus type 1 (BE-1), was monitored at four OSDS sites in North Carolina by Sobsey et al. (1981).

The OSDSs were located in soils with sand or sandy loam textures and seasonally high water tables. The depth to the seasonally high water table varied from 8 inches (20 cm) to 58 inches (147 cm) below the soil surface. The seasonally high water table was less than 24 inches (60 cm) below the drainfield distribution line at three of the sites and on occasion immersed the distribution lines in ground water at these sites. BE-1, used as a model for enteroviruses, was inoculated into two septic tanks and was found to persist longer than the residence time of the wastewater in the septic tank. BE-1 was inoculated into the distribution box of two of the systems in order to achieve rapid discharge of the virus into the drainfield. Initial concentrations of 10^5 PFU/ml were reduced to 10 PFU/ml within three to five weeks after inoculation into the septic tank or distribution box. The BE-1 was isolated from ground water samples within one to three days after it was inoculated into the septic tank or distribution box. Monitoring wells located 5 to 115 ft (1.5 to 35 m) from the drainfields detected BE-1 at several sampling times. The lowest frequency of virus isolation in monitoring wells was found at the site with the greatest vertical separation between the water table and the drainfield distribution line.

Virus counts ranging from 8 to 80 PFU/liter were isolated in ground water sampled 5 ft (1.5 m) from the distribution lines. Rapid movement of viruses was noted by the isolation of 908 PFU/liter of BE-1 from ground water collected 115 ft (35 m) from the nearest distribution line only two days after inoculation. Sobsey et al. (1981) concluded that OSDSs provided ineffective treatment for removal of added BE-1 under saturated flow conditions in soils with sand and sandy loam textures.

Vaughn et al. (1983) studied the movement of coliform organisms and enteroviruses into ground water below a large OSDS on Long Island, N.Y. The OSDS served 120 apartments and consisted of a 3650 gallon (13,800 liter) septic tank, three distribution pools, and forty leaching pools, or seepage pits, whose bottoms were 11.8 ft (3.6 m) above the water table.

Using the center of the system as the point of origin, ten monitoring wells that penetrated the water table to a depth of 5 to 6 ft (1.5 to 1.8 m) were installed at lateral distances from the center ranging from 5 to 198 ft (1.5 to 60.4 m). In addition, five deep wells which penetrated the water table to a depth of 19.7 to 59 ft (6 to 18 m) were installed at lateral distances ranging from 5 to 220 ft (1.5 to 67 m). Coliform organisms, although occurring in high concentrations in the distribution pools (10^5 to 10^8 PFU/100 ml), were rarely detected in ground water from monitoring wells located beyond a distance of 5 ft (1.5 m) from the system.

Virus isolates were recovered from each of the downgradient wells on at least one occasion. In general, virus concentrations varied with monitoring well distance from the effluent source. The virus concentration in ground water from shallow monitoring wells located 5 to 15 ft (1.5 to 4.6 m) from the system ranged from 0.002 to 10.8 PFU/liter and from 0.002 to 0.05 PFU/liter in wells located 150 ft (45.7 m) and 198 ft (60.4 m) from the system. By the end of the first six months of sampling, it had become apparent that viruses were being transported through the upper 6 ft (1.8 m) of the aquifer. Three of the five deep-ground water monitoring wells tested positive for the presence of enteroviruses. The presence of viruses in ground water sampled from depths of

19.7 and 59 ft (6 and 18 m) below the surface of the water table was related to mixing of deep ground water with contaminated shallow ground water during drawdown caused by sampling. Similar effects would be expected during normal usage of any water supply well. Viruses were recovered in a deep well sample located 200 ft (67 m) downgradient from the source. The authors concluded that 11.8 ft (3.6 m) of unsaturated sandy soil was ineffective in preventing contamination of the shallow ground water aquifer by this large-scale OSDS.

3.6.3. Column studies

Laak and McLean (1967) reported persistence of added Sabin-attenuated Poliovirus-3 in oxidation tank wastewater and in column outflow after passage through sand, sandy loam, and "garden soil" from Toronto, Ontario, Canada. Sand columns were 2 inches (5.5 cm) in diameter and 12.6 inches (32 cm) long, and sandy loam and "garden soil" columns were 2 inches in diameter and 5.5 inches (14 cm) long. Each column received a single effluent application containing the equivalent of approximately 100,000 PFU/ml of attenuated poliovirus. Additional aeration tank effluent was subsequently added and the outflow examined. Viruses persisted in the aeration tank effluent and in the sand columns for up to seven days. Viruses were detected in the outflow from the sand columns one to three hours after the virus-enriched effluent was added. A decline in the virus content of the outflow was noted for the next four days, and no viruses were detected after 120 hours. Viruses were detected in the outflow from the sandy loam columns after seven hours and were recovered for up to 96 hours. Viruses were detected in the "garden soil" columns after three hours and reached a maximum concentration after seven hours. Viruses were recovered from the outflow for 72 hours. Laak and McLean (1967) concluded that poliovirus-3 was not eliminated by filtration through sand, sandy loam, or "garden soil," and that the decrease in virus titres in the outflow resulted mainly from dilution.

Green and Cliver (1975) monitored the removal of Poliovirus type 1 from 3 or 5.7 inch (7.7 or 14.6 cm)-diameter by 24 inch (60 cm)-high soil columns constructed of medium sand in Wisconsin. The removal of viruses from the septic tank effluent by sand columns was found to be related to conditioning or ageing of the system, dose rate, and temperature. Columns to which effluent had been applied prior to application of poliovirus were found to retain less virus and to allow greater depth of penetration of virus than fresh sand columns. The authors concluded that virus retention by the soil beneath an OSDS would decrease with time as the system aged.

A sand column receiving 850 ml of effluent containing greater than 10^5 PFU of poliovirus daily for a year contained only 1 PFU of poliovirus in the outflow. Increasing the dose to 8.5 liters of effluent containing 4.6×10^5 PFU/ml resulted in a concentration of 4.5×10^2 PFU/ml in the outflow the following day. Unsaturated and saturated flow conditions in the soil corresponded to loading rates of 850 ml and 8.5 liters per day, respectively. At a temperature of 46°F (8°C), the sand was less retentive and the virus was inactivated more slowly, if at all, than at temperatures of 68 to 72°F (20 to 22°C).

Lance et al. (1982) compared adsorption and movement of enteroviruses and coliforms in soil columns. Echovirus type 1, Echovirus type 29, and Poliovirus type 1 seeded in secondary sewage wastewater were applied to 98 inch (250 cm)-long columns of loamy sand. Water samples from various column depths showed that the adsorption patterns of Echo 29 and Polio 1 were quite similar. Ninety percent of the Echo 29 and Polio 1 viruses were adsorbed in the top 0.8 inch (2 cm) of the soil. Fewer Echo 1 viruses were adsorbed near the soil surface, with only seventy-seven percent of the viruses adsorbed in the top 0.8 inch (2 cm) of the soil. Below a depth of 16 inches (40 cm), the adsorption pattern of Echo 1 virus resembled that for Polio 1 and Echo 29 viruses. Virus movement through the soil column roughly paralleled coliform movement but no survival studies were conducted. At a depth of 51 inches (130 cm), both viruses and fecal coliforms were reduced to less than 0.1% of their concentration in the wastewater. Neither viruses nor coliforms were detected in the column outflow at the 98 inch (250 cm) depth.

3.6.4. Summary

Viruses occur in effluent in varied concentrations that reflect the combined infection and carrier status of the residents utilizing the OSDS (Berg, 1973). Reported recoveries of viruses in septic tank effluent range from 0 to 7000 plaque forming units (PFU)/liter (Clark et al., 1974; Yeager and O'Brien, 1977; Hain and O'Brien, 1979; Vaughn, 1983). The public health hazard posed by enteric viruses in effluent is difficult to assess due to the inapparent nature of many viral infections and the difficulty encountered in isolating small numbers of viruses from water supplies (Hain and O'Brien, 1979). Moreover, the number of viruses that constitute a disease-producing dose varies, although it has been shown that one viral plaque forming unit is capable of producing human infection (Katz and Plotkin, 1967).

Viruses are removed or inactivated in natural soil systems by the mechanisms of adsorption, filtration, precipitation, biological enzyme attack, and natural die off. Laboratory column studies indicate that virus adsorption by soil generally increases with increasing cation exchange capacity, clay content, specific surface area, and ionic composition of the soil solution. Low soil pH, low soil moisture content, and low effluent loading rates also increase virus retention by soil.

Several ground water monitoring studies have reported transport of viruses to ground water from OSDSs under conditions of saturated or near-saturated flow due to high water tables or high effluent loading rates (Hain and O'Brien, 1979; Stramer and Cliver, 1981; Sobsey et al., 1981; Vaughn et al., 1983). Additional ground water monitoring studies of OSDSs operating under water-unsaturated soil conditions are necessary before the fate and transport of viruses in soil surrounding OSDSs can be evaluated.

4. DEPTH TO SEASONAL HIGH WATER TABLE

Proper functioning of an OSDS is achieved only if a sufficient volume of soil is available to absorb the effluent and to purify it (Bouma, 1979). A vertical separation distance of 24 inches (60 cm) between the bottom of the absorption system and the seasonally high water table is stipulated by Chapter 10D-6 of the Florida Administrative Code (State of Florida, 1983). A minimum separation distance of 24 inches (60 cm) is also suggested by the U.S. Department of Housing and Urban Development (1978).

The Design Manual - Onsite Wastewater Treatment and Disposal Systems (Clements and Otis, 1980), and Miller and Wolf (1975) recommend a water-unsaturated soil depth beneath the absorption system of from 24 to 48 inches (60 to 120 cm). Walker et al. (1973b), Bouma (1979), and Clements and Otis (1980) indicate that the depth of water-unsaturated soil necessary to prevent ground water contamination should be determined by the permeability of the soil. Soils with relatively rapid permeabilities, such as sand and loamy sand, may require a greater depth of water-unsaturated soil below the absorption system than soils with relatively slow permeabilities. Ground water monitoring studies by Carlile et al. (1981) and Cogger and Carlile (1984) found that the concentration of effluent contaminants in ground water decreased as the vertical separation distance between the bottom of the absorption system and the seasonally high water table increased.

A 24 inch (60 cm) depth of water-unsaturated soil below the absorption system may be inadequate to prevent bacterial and viral contamination of ground water. Ziebell et al. (1975b) indicated that low levels of fecal indicator organisms did pass through 24 inches (60 cm) of "sandy soil" fill material in a mound system in Wisconsin. Detection of Pseudomonas aeruginosa, a pathogenic bacteria, in effluent at the contact between the mound fill and the natural soil led them to conclude that additional treatment by the natural soil was necessary for purification of effluent. Kristiansen (1981c,d) reported fecal coliform concentrations ranging from 100 to 10,000 organisms/100 ml in outflow from 30 inch (75 cm)-deep sand filters receiving septic tank effluent. The relatively high concentration of fecal coliform bacteria in the outflow, and fecal coliform concentrations of up to 1000 organisms/gram of sand at a depth of 24 inches (60 cm), indicate that passage of effluent through 24 inches (60 cm) of sand may be insufficient to prevent bacteriological contamination of ground water.

Wilson et al. (1982) created a minimum separation distance of 24 inches (60 cm) between the bottom of the absorption system and the seasonally high water table by installing perimeter tile drains around six OSDSs located in Oregon. The soils were moderately well- to somewhat poorly drained, and developed in silty alluvium. The perimeter tile drainage system was located at a lateral distance of 19.7 ft (6 m) from the OSDSs. Total and fecal coliform levels in the tile drainage outflow varied widely during two 4-month monitoring periods. Mean total coliform and fecal coliform concentrations in the tile drainage outflow ranged from 435 to 2843 organisms/ml, and 10 to 484 organisms/ml, respectively.

The fate and transport of bacteria and viruses from septic tank effluent and municipal wastewater has frequently been examined with 24 inch (60 cm)-long soil columns. Viruses are commonly detected in outflow from 24 inch (60 cm) soil and sand columns, indicating that additional soil treatment is necessary for effluent purification (Robeck et al. 1962; Drewry and Eliassen, 1968; Sproul, 1973; Bitton, 1975; Green and Cliver, 1975; Lance, 1978). Research is needed to evaluate the adequacy of 24 inches (60 cm) of water-unsaturated soil above the seasonal high water table for effective removal of chemical and biological contaminants from effluent in Florida soils and climatic conditions.

A water-unsaturated soil depth of 36 inches (90 cm) between the bottom of the absorption system and the seasonally high water table is commonly recommended for acceptable chemical and biological treatment of septic tank effluent in order to protect ground water quality (Bouma et al., 1972, 1975a,b; Beatty and Bouma, 1973; Sproul, 1973; Bouwer, 1974; Bouma, 1975a,b, 1979; Brown et al., 1978a,b, 1979, 1984; Hagedorn and McCoy, 1979; Boyle and Otis, 1979; Russell and Axon, 1979, 1980; Hansel and Machmeier, 1980; Wicks and Erickson, 1982).

Bouma et al. (1972) reported effective removal of fecal indicator organisms and phosphorus from septic effluent by OSDS seepage beds installed in sand and coarse sand in Wisconsin. Water-unsaturated soil beneath the crusted seepage beds extended to a depth of 36 inches (90 cm) above the seasonal high water table. Brown et al. (1978b, 1984) indicated that fecal coliforms and coliphages were removed from septic tank effluent after passage through approximately 3.3 ft (1 m) of soils ranging in texture from sandy loam to clay in undisturbed soil lysimeters. Hagedorn and McCoy (1979) noted that fecal organisms generally moved less than 3.3 ft (1 m) when water-unsaturated flow conditions prevailed. Magdoff et al. (1974b) found that 36 inch (90 cm)-long soil columns constructed of 24 inches (60 cm) of sand or sandy loam soil over 12 inches (30 cm) of silt loam soil were effective in removal of fecal coliforms from septic tank effluent. In-situ ground water monitoring studies and laboratory column studies support a recommended water-unsaturated soil depth of 36 inches (90 cm) between the bottom of the absorption system and the seasonally high water table.

A greater depth of water-unsaturated soil below the absorption system can result in a greater degree of effluent purification. The Manual of Septic Tank Practice (U.S. Public Health Service, 1967) indicates that a minimum of 48 inches (120 cm) of water-unsaturated soil is necessary to achieve purification of effluent. A 48 inch (120 cm) depth of water-unsaturated soil between the bottom of the absorption system and the seasonally high water table has been recommended to maintain high quality ground water and protect public health (Romero, 1970; Bouwer, 1974; Viraraghavan and Warnock, 1976a,b; Viraraghavan, 1977; Crane and Moore, 1984). Plews (1977) reported that a majority of states surveyed throughout the Unted States require a separation distance of 48 inches (120 cm) between the bottom of the absorption system and the seasonally high water table. However, several coastal states require separation distances of less than 24 inches (60 cm), because only a very small percentage of all soils in some coastal states would be acceptable for conventional OSDSs having separation distances greater than 24 inches (60 cm).

The requirement that ground water not occur within 24, 36, or 48 inches (60, 90, or 120 cm) of the bottom of the absorption system requires a knowledge of the seasonal high ground water elevation at a particular site. Seasonal high water tables can be determined with ground water observation wells or estimated by examining the soil mottling pattern, other morphologic features, landscape position, and vegetation (Simonson and Boersma, 1972; Bouma, 1973, 1975b, 1979; Vepraskas et al., 1974; Rathbun, 1979; Fredrickson, 1980). Bouma (1979) noted that ground water observations prior to OSDS installation are often made when the ground water is below the maximum ground water elevation. Installation of an OSDS without accurate knowledge of the seasonal high water table can result in a decreased separation distance due to rising water tables and decreased effluent treatment.

Discharge of effluent from an OSDS drainfield may result in mounding or raising of the ground water beneath the system. Mounding of the underlying ground water can reduce the unsaturated soil depth through which the effluent moves, and thus decrease effluent treatment (Dudley and Stephenson, 1973; Healy and Laak, 1974; Finnemore et al., 1983; Vaughn et al., 1983; Chan and Sykes, 1984).

Soils containing impermeable or slowly permeable soil horizons commonly develop perched water tables during periods of seasonally high rainfall (Reneau and Pettry, 1975a; Hagedorn et al., 1981). Soils with percolation rates slower than 1 inch/hr (2.5 cm/hr) commonly have perched water tables within 24 inches (60 cm) of the soil surface due to perching of infiltrative effluent above slowly permeable subsoil horizons. Perched water tables can result in saturated flow of effluent, reduced treatment of effluent, lateral flow, and/or transport of the effluent to the soil surface.

Alternative systems such as mounds or fills have been used and/or recommended to increase the separation distance between the bottom of the absorption system and the seasonal high water table or bedrock

(Bouma et al., 1972, 1975a,b; Magdoff et al., 1974b; Small Scale Waste Management Project, 1978; Guthrie and Latshaw, 1980; Hansel and Machmeier, 1980).

Artificial drainage by subsurface tile drains or drainage ditches can also increase the water-unsaturated soil depth below an absorption system, and has been shown to improve the hydraulic functioning of OSDSs in soils with slowly permeable soil horizons or in soils subjected to seasonally high water tables (Reneau, 1978, 1979; Hagedorn et al., 1981; Wilson et al., 1982).

5. ON-SITE SEWAGE DISPOSAL SYSTEM DENSITY

OSDS density (i.e., the number of OSDSs per unit land area) is one of the most important parameters influencing local and regional contamination of ground water (Scalf et al., 1977; U.S.E.P.A., 1977). Increasing density of OSDS installations decreases the dilution of effluent constituents and increases potential contamination of ground water.

Perkins (1984) noted that most regulatory agencies controlling OSDS installations depend on setback distance requirements between wells and OSDSs to provide adequate dilution and attenuation of chemical and biological contaminants and thus to prevent contamination of private drinking water supplies. In order to comply with established distance requirements for separation of OSDSs and private water wells, lots must have a minimum linear dimension greater than the minimum setback distance. If a municipal or community water supply exists, minimum lot size is commonly decreased and may be limited only by the area necessary to locate the dwelling and OSDS. Municipal water supplies may alleviate the concern for contamination of private water supply wells by OSDSs, but can result in increased contamination of local or regional ground water by allowing increased OSDS density. Perkins (1984) indicated that population density ultimately determines the effluent load per unit of land area and the concentration of contaminants in ground water.

Woodward et al. (1961) reported a correlation between rural population density and well contamination near Coon Rapids, Minnesota. An area with a population density of 0.54 persons/acre (1.33 persons/hectare) had two percent of its private water wells contaminated with nitrate, while an area with a population density of 2.7 persons/acre

(6.7 persons/hectare) had more than 29 percent of private water wells contaminated with nitrate.

Miller (1972) recommended that house lot-size requirements in Delaware be increased from 0.5 acre (0.2 hectare) to 2.0 acres (0.8 hectare) after a water quality survey indicated that 25 percent of the water wells in the shallow water table aquifer had nitrate-nitrogen concentrations of 4.5 mg/liter (i.e., twice background levels). Nitrate-nitrogen concentrations in ground water from areas with well-drained soils and lot sizes ranging from 0.25 to 0.5 acres (0.1 to 0.2 hectares) were as high as 31 mg/liter (Miller, 1975).

Walker et al. (1973) estimated that a maximum OSDS density of 2 OSDSs/acre (4.9 OSDSs/hectare) would be necessary to insure adequate ground water dilution of nitrate-nitrogen to concentrations below 10 mg/liter in loamy sand soils at four sites in Wisconsin. The density of OSDSs was estimated on the basis that a family of four would generate 73 lbs (33 kg) of nitrate-nitrogen per year. Nitrate contributions from OSDS effluent to ground water in sands were estimated to be approximately equal to those from natural sources (i.e., rainfall and decomposition of organic matter) when one OSDS was located on six acres of land.

Morrill and Toler (1973) indicated that the contribution of OSDS to dissolved-solids load or soluble salt concentration in streams draining seventeen small drainage basins near Boston, Massachusetts could be predicted on the basis of OSDS density. OSDS density ranged from 0 to 900 OSDS/mile^2 (0 to 348 OSDS/km^2). In the range of housing densities observed, the dissolved solids concentration in stream flow was found to increase by 10 to 15 mg/liter per 100 houses per square mile.

Pitt et al. (1974a, 1975) monitored ground water quality near Homestead, Florida in an area with OSDS densities of 4 OSDSs/acre (9.8 OSDSs/hectare) and 1 OSDS/acre (2.5 OSDS/hectare). Slightly higher concentrations of sodium, total coliforms, fecal coliforms, and fecal streptococci were detected in ground water of the higher density area.

Geraghty and Miller (1978) collected 865 ground water samples from 54 wells on Long Island, New York and correlated nitrate concentration with OSDS density. A nitrate-nitrogen concentration in ground water of 10 mg/liter or more was detected in fifty percent of the ground water samples when OSDS density exceeded 2.8 OSDSs/acre (6.9 OSDSs/hectare). Densities of less than 1.25 OSDSs/acre (3.1 OSDSs/hectare) resulted in less than ten percent of the ground water samples containing nitrate-nitrogen concentrations of 10 mg/liter or more.

Konikow and Bredehoeft (1978) developed a computer simulation model to evaluate the effects of OSDS density on water quality of the Rio Grande alluvial aquifer in New Mexico. They concluded that steady state levels of nitrate in ground water may not be reached for many decades, and that the effect of lot size on nitrate concentrations in ground water is not necessarily a linear function. Predicted nitrate-nitrogen concentrations of ground water after 10 years of OSDS effluent applications in the Rio Grande Valley were 60 mg/liter below 0.25 acre (0.1 hectare) house lots and 35 mg/liter under 1.2 acre (0.5 hectare) house lots. Nitrate concentration in ground water was dependent on lot size, ground water mixing, street orientation with respect to ground water flow direction, and ground water velocity.

Ford et al. (1980) reported that nitrate contamination of ground water was associated with increased housing density in unsewered residential areas of Jefferson County, Colorado. Contamination of ground water with nitrate-nitrogen concentrations exceeding 20 mg/liter was associated with OSDS densities exceeding 1 OSDS/acre (2.5 OSDS/hectare) and with well setback distances of 100 ft (30 m) or less.

Duda and Cromartie (1982) and Everette (1982) related closure of shellfish harvesting beds to density of OSDSs along the coast of North Carolina. They examined the bacteriological quality of surface water from tidal estuaries and tributary freshwater creeks with different OSDS densities in four coastal watersheds. No industrial or point-source discharges were located in the watersheds, and all residential development utilized OSDSs. The watersheds ranged in size from 0.2 to 1.35 miles^2 (0.32 to 2.2 km^2). On-site sewage disposal system density ranged from 0.08 to 0.52 OSDSs/acre (0.20 to 1.28 OSDSs/hectare). A highly significant correlation was found between bacterial levels in surface water and increasing density of OSDSs. On-site sewage disposal system densities greater than 0.17 OSDSs/acre (0.42 OSDSs/hectare) resulted in closure of shellfish harvesting beds in the watersheds examined. Forty-five to seventy percent of the OSDSs were estimated to be located in soils with severe limitations for on-site sewage disposal.

Trela and Douglas (1978) developed a model to estimate OSDS density which would prevent nitrate-nitrogen concentration in ground water from exceeding 10 mg/liter below sandy soils in the New Jersey Pine Barrens. The minimum land area or lot size was 0.2 acres (0.08 hectares) per capita or 0.8 acres (0.32 hectares) per household, assuming a family of four.

Brown (1980) and Tateman and Lee (1983) modified the model proposed by Trela and Douglas (1978) and calculated a minimum land area or lot size needed to prevent nitrate-nitrogen concentration in ground water from exceeding 10 mg/liter. Brown (1980) determined that a minimum land area of 0.84 acres (0.34 hectares)/household was necessary in Texas. Tateman and Lee (1983) calculated that a land area of 0.25 acres (0.1 hectares)/capita or 1 acre (0.4 hectares)/household was necessary to achieve the same result in Delaware.

Holzer (1975), Peavy and Brawner (1979), and Starr and Sawhney (1980) recommended that OSDS density should not exceed an average of one system per acre on well-drained soils, and Olivieri et al. (1981) suggested that maximum overall OSDS density should be one OSDS per 1.4 acres (0.6 hectares) in order to maintain high-quality ground water and to protect public health.

Russell and Axon (1979, 1980) utilized OSDS density to evaluate areas where use of OSDSs could prevent potential health hazards in the Loxahatchee River Environmental Control District (i.e., southeast Martin County and northeast Palm Beach County, Florida). Russell and Axon (1979) indicated that ten out of sixteen study areas had OSDS densities exceeding 5 OSDSs/acre (12.4 OSDSs/hectare) in at least a portion of the study areas.

Harkin et al. (1979) indicated that nineteen mound systems monitored in Wisconsin supplied considerably less nitrate to ground water than did conventional OSDSs, due to enhanced denitrification. A family of four was estimated to contribute 43 lbs (19.7 kg) of nitrogen per year to ground water, considerably less than the 73 lbs (33 kg) of nitrogen contributed by conventional OSDSs (Walker et al., 1973). Harkin et al. (1979) calculated that mound system densities of greater than 1.3 per acre (3.2 mound systems/hectare) would be necessary to reach an overall 10 mg/liter nitrate-nitrogen concentration in ground water.

The Florida Department of Environmental Regulation (1979) determined OSDS densities for the forty-one counties in Florida which are not included in designated Section 208 planning areas (Table 4). The number of OSDSs in each county was obtained from either 1970 census data (U.S. Dept. of Commerce, 1970) updated with data from the Florida Dept. of Health and Rehabilitative Services (1979), or from survey estimates submitted by county sanitarians, whichever was larger. Land acreage was adjusted to developable land acreage by excluding wildlife management areas, national parks, wildlife refuges, etc. On-site sewage disposal system density ranged from 0.303 OSDSs/acre in Monroe County to 0.0057 OSDSs/acre in Hamilton County (Table 4). The OSDS densities are below those recommended by Betz (1975), Holzer (1975), Peavy and Brawner (1979), and Starr and Sawhney (1980) to prevent deterioration of ground water by OSDSs. However, these counties generally represent the rural areas of the state without major population centers. Considerably higher OSDS densities exist in Dade, Broward, and Hillsborough Counties. The Florida Department of Environmental Regulation (1979) identified Duval, St. Johns, Dixie, Marion, Lake, Highlands, Indian River, Martin, and Monroe Counties as areas in the non-designated Section 208 area where OSDSs may impact on ground water quality based on OSDS density and septic tank soil suitability ratings (Soil Conservation Service, 1978) for the soil associations found in the counties. Some of the counties identified by the Florida Department of Environmental Regulation as having a potential to contaminate ground water are located in high recharge areas of the Florida aquifer (Stewart, 1980), and therefore have particular potential for affecting water quality.

Table 4. ON-SITE SEWAGE DISPOSAL SYSTEM DENSITIES AND SOIL LIMITATION RANKINGS FOR 41 FLORIDA COUNTIES (Fla. Dept. of Environ. Reg., 1979).

County	No. of Septic Tanks	Developable Land Acres	Septic Tank Density Tanks/Acre	Septic[a] Tank Density Rating	Soil[b] Limitation Ranking	Combined[c] Ranking Scores	Total[d] Ranking
Alachua	22,192	529,665	.0419	9	33	42	22
Baker	3,500	253,794	.0138	26	8	34	13
Bradford	4,955	187,968	.0264	16	25	41	20
Calhoun	2,207	280,962	.0079	36	28	64	34
Citrus	19,406	299,606	.0648	3	38	41	20
Clay	15,000	308,848	.0486	6	30	36	16
Columbia	7,245	298,821	.0182	22	22	44	23
Dixie	3,500	157,275	.0223	18	13	31	10*[e]
Duval	70,000	486,144	.1440	2	17	19	6*
Flagler	3,000	291,978	.0103	31	3	34	13
Franklin	1,784	308,308	.0058	40	19	59	32
Gadsden	6,216	307,407	.0202	20	31	51	28
Gilchrist	4,000	221,632	.0181	23	41	64	34
Gulf	2,882	280,703	.0103	32	12	44	23
Hamilton	1,727	302,016	.0057	41	32	73	40
Hernando	10,573	261,185	.0405	10	37	47	26
Highlands	15,333	526,960	.0291	14	4	18	5*
Holmes	3,885	307,952	.0126	28	27	55	31
Indian River	12,321	324,096	.0380	11	1	12	3*
Jackson	7,850	591,136	.0133	27	35	62	33
Jefferson	2,200	305,091	.0072	38	14	52	30
Lafayette	2,000	264,940	.0076	37	29	66	37
Lake	30,806	532,727	.0578	5	5	10	2*
Levy	9,000	594,824	.0151	24	24	48	27
Liberty	1,900	230,021	.0083	35	16	51	28

continued

Table 4. cont'd.

County	No. of Septic Tanks	Developable Land Acres	Septic Tank Density Tanks/Acre	Tank Density Rating	Soil[b] Limitation Ranking	Combined[c] Ranking Scores	Total[d] Ranking
Madison	3,000	449,856	.0067	39	36	75	41
Marion	42,909	731,190	.0587	4	20	24	9*
Martin	14,887	344,455	.0432	8	9	17	4*
Monroe	12,261	40,465	.3030	1	2	3	1*
Nassau	7,942	323,765	.0245	17	15	32	11
Okeechobee	5,610	497,280	.0113	30	10	40	18
Putnam	19,995	457,375	.0437	7	26	33	12
St. Johns	10,274	372,670	.0276	15	7	22	8*
St. Lucie	12,646	371,339	.0341	13	6	19	6*
Sumter	9,558	278,670	.0343	12	23	35	15
Suwannee	5,440	436,980	.0125	29	40	69	39
Taylor	4,964	559,729	.0089	34	11	45	25
Union	1,808	94,299	.0192	21	18	39	17
Wakulla	4,401	204,413	.0215	19	21	40	18
Walton	6,170	426,909	.0145	25	39	64	34
Washington	3,500	372,725	.0094	33	34	67	38

a) Septic tank density ranking is based on decreasing septic tank density.

b) Soil limitation rating is based on decreasing weighted average septic tank suitability ratings for soil associations in the counties.

c) Combined ranking scores is the sum of septic tank density ranking and soil limitation ranking.

d) Total ranking is an overall numerical ranking based on decreasing combined ranking score.

e) The * indicates counties identified by Florida Department of Environmental Regulation as potential areas for deterioration of ground water outside designated Section 208 areas.

6. GROUND WATER IN FLORIDA

Florida lies entirely within the Coastal Plain physiographic province and ground water region (Miller et al., 1977). Virtually all of Florida is underlain by underground sources of drinking water (i.e., aquifers containing water with dissolved solids concentrations of less than 10,000 mg/liter) which are capable of yielding at least small quantities of water to wells (Hand and Jackman, 1982). Deep artesian aquifers and shallow water table aquifers provide over 90% of the states' potable water supply.

The four major developed aquifers in Florida are referred to as the Floridan, the Biscayne, the Shallow, and the Sand and Gravel aquifers. The Floridan aquifer, the most extensive and widely used aquifer in the state, supplies water to all but southernmost and westernmost Florida. In some areas, the Floridan aquifer is at or near land surface, while in other areas it is as much as 1000 ft (305 m) below land surface (Hyde, 1975). The Floridan aquifer is considered a confined aquifer where it is covered by low permeability "clayey sands" and sandy clay, and is considered an unconfined aquifer where it is thinly veneered by sands or is exposed at the land surface (Knapp, 1978). The unconfined aquifer may have a greater potential for contamination by OSDSs than the confined aquifer, because the unconfined aquifer may be recharged directly by infiltrating effluent. The confined Floridan aquifer is less likely to be contaminated by OSDSs because the overlying, low permeability sediments protect it from surface contamination except in areas where sinkholes are directly connected to the aquifer. Sinkholes that are directly connected to the Floridan aquifer are one of the most rapid means of recharge. Many sinkhole areas have closed-basin topography characterized by internal drainage and some areas have numerous sinkholes that are open from land surface to the Floridan aquifer.

Soils, topography, and landforms have a distinct influence on recharge of the Floridan aquifer (Stewart, 1980). The Central Florida Ridge (Caldwell and Johnson, 1982), containing many lakes, depressions, and sinkholes that do not have surface outlets, is the principal area of natural recharge. The recharge rate generally is low in areas dominated by poorly drained soils, moderate in areas dominated by moderately well-drained soils, and high in areas of permeable, well-drained soils. Some of the highest rates are in areas of well-drained sand ridges of Central and West-Central Florida, which include portions of Orange, Lake, Polk, Pasco, and Hernando Counties (Stewart, 1980).

Prevention of ground water contamination by OSDS effluent in the high recharge areas of the Floridan aquifer may be necessary to protect water quality not only in the high recharge regions but also in other areas that draw their water from the same aquifer. The high recharge areas of the Floridan aquifer along the Central Florida Ridge had some of the highest numbers of OSDS installations in 1983 (Florida Dept. of Health Rehabilitative Services, 1984). The Florida Department of Environmental Regulation (1979) has identified several counties in the high recharge area of the Floridan aquifer as potential areas for ground water contamination by OSDSs, based on OSDS density and soil limitations.

Kreitler and Browning (1983) indicate that limestone aquifers in humid environments that are recharged by percolation through the soil appear to be more susceptible to contamination by OSDSs than are aquifers in subhumid environments that feature thick unsaturated sections and are recharged by streams. Percolation of recharge water through soil can carry contaminants held in the unsaturated zone to the ground water. In arid to semiarid environments with a deep water table aquifer, recharge through soil percolation is less important because of a characteristically thick unsaturated zone and a high rate of evapotranspiration.

Throughout most of the state the uppermost aquifer is contiguous with the land surface, except for a few areas in central peninsular Florida, and the water table commonly lies at very shallow depths below the land surface (Hand and Jackman, 1982). The uppermost aquifer is generally referred to as the surficial aquifer, but in areas where large quantities of ground water have been developed the surficial aquifer is identified as the Biscayne aquifer, the Shallow aquifer, or the Sand and Gravel aquifer.

The Biscayne aquifer underlies an area of about 3000 square miles (7767 square km) in Dade, Broward, and southern Palm Beach Counties. The rapid permeability of the Biscayne aquifer makes it one of the highest yielding aquifers in the world (Yoder et al., 1981). The Biscayne aquifer has rapid interaction between the local water table and surface water, because almost all surface water is ground water which has been exposed by dredging.

Dade, Broward, and Palm Beach Counties have some of the highest numbers of OSDSs in the state. Increased OSDS density has been identified as a cause of chemical and biological contamination of ground and surface water along the Atlantic Coastal Plain (Miller, 1972; Geraghty and Miller, 1978; Duda and Cromartie, 1982). Agricultural fertilization has been identified as the principal source of elevated nitrate levels in ground water of the Biscayne aquifer, but OSDSs were also found to contribute to elevated nitrate levels (Pitt, 1974a,b,c; Pitt et al., 1975).

The Shallow aquifer is present over much of the state, but in most areas it is not an important source of drinking water because a better supply is available from other aquifers. The Shallow aquifer is an important source of water in some Atlantic and Gulf coastal areas, where artesian water is highly mineralized. In rural areas where water requirements are small, the aquifer is tapped by shallow, small diameter, sand point wells. Domestic or small public wells are commonly less than 50 ft (15.2 m) deep (Hyde, 1975).

The Sand and Gravel aquifer is the principal source of water supply in extreme West Florida. This aquifer extends beneath all of Escambia and Santa Rosa Counties and part of western Okaloosa County. Domestic and public well supplies vary from 30 to 500 ft (9 to 153 m) in depth (Hyde, 1975). Commonly, wells which draw water from shallow depths are more prone to chemical and biological contamination from OSDSs than are wells which draw water from deeper aquifers.

Florida's humid climate is characterized by a high average annual rainfall, ranging from about 45 to 60 inches (120 to 150 cm), and by a rainfall distribution pattern that is highly nonuniform in time and space (Butson and Prine, 1968). More than one-half of the annual rainfall generally occurs during the summer, when local thunderstorms sometimes produce 3 inches (7.6 cm) or more of rain over a small area in an hour to two. The general trend in annual distribution of rainfall stipulates wet summer months and relatively dry periods during the rest of the year, with the exception that in the Florida Panhandle a distinct secondary rainy season in late winter often results in the development of two separate wet seasons (summer and late winter) there. Depths to ground water typically decrease during rainy periods and increase during the rest of the year.

Although rainfall amounts are typically high during some months in Florida, relatively high rates of evapotranspiration on well-vegetated sites may tend to reduce maximal rise of ground water in the soil. Smajstrla et al. (1984) used the Penman equation to calculate annual potential evapotranspiration rates (assuming an active plant canopy and moist soil) for 9 locations throughout Florida. Rates ranged from 46 inches (118 cm) at Tallahassee to 55 inches (141 cm) at West Palm Beach. Actual evapotranspiration would be expected to be less than these potential amounts due to periodic occurrence of dry soil conditions. In any event, one can see that the nature and extent of vegetative cover over large areas of land can have an impact on the degree of rise and fall of the water table.

7. SUMMARY

On-site disposal of septic tank effluent is the most common means of domestic waste treatment in rural and unincorporated areas without sewer systems. Over 1.3 million families in Florida are served by on-site sewage disposal systems (OSDSs). These families introduce nearly 170 million gallons (643 million liters) per day into the subsurface environment, making it potentially one of the largest sources of artificial ground water recharge in the state.

Properly sited, designed, constructed, and operated OSDSs provide an efficient and economical alternative to public sewer systems, particularly in rural and sparsely developed suburban areas. Increasing public concern and awareness for environmental quality and public health require effective treatment and disposal of domestic wastes for all homes in unsewered areas. Septic tank effluent contains varied concentrations of nitrogen, phosphorus, chloride, sulfate, sodium, toxic organics, detergent surfactants, and pathogenic bacteria and viruses. Widespread use of conventional[1] OSDSs can result in contamination of ground and surface water if the soil does not effectively treat or purify the effluent before it encounters ground water.

Transformation, retention, loss, or movement of nitrogen in natural soil systems is governed by the mechanisms of mineralization, nitrification, denitrification, adsorption, biological uptake, and volatilization. Ground water monitoring studies and laboratory column studies

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[1] A conventional OSDS is defined as a septic tank and soil absorption trench or bed installed below the natural soil surface.

indicate that approximately 20 to 40% of the nitrogen in effluent may be adsorbed or otherwise removed before the effluent reaches ground water (DeVries, 1972; Andreoli et al., 1979; Harkin et al., 1979; Peavy and Brawner, 1979; Lance et al., 1980; Starr and Sawhney, 1980; Laak, 1982).

In a properly sited, designed, constructed, and operated conventional OSDS, nitrification is the predominant nitrogen-transformation mechanism in aerobic, water-unsaturated soil beneath the biological clogging mat or crust. It results in conversion of ammonium and organic nitrogen to nitrate. The soil cation exchange capacity is ineffectual in sorbing nitrate, a soluble anion, and the nitrate moves nearly uninhibited to ground water.

Numerous ground water monitoring studies have detected nitrate-nitrogen concentrations exceeding 10 mg/liter at considerable distance from absorption systems. Attenuation of nitrate by dilution is the only mechanism which significantly lowers nitrate-nitrogen concentrations in ground water below conventional OSDSs in aerobic, water-unsaturated soils. Denitrification within a properly sited, designed, and operated conventional OSDS is unlikely but, under conditions of high water tables or slowly permeable soils, nitrate may be denitrified if a biologically useful source of organic carbon is readily available.

Water quality surveys throughout the United States have identified local and regional contamination of ground water and surface water by nitrate derived from OSDSs. Restricting OSDS density (the number of OSDSs per unit land area) lowers the nitrate input from OSDSs to ground water per unit land area and may effectively control levels of nitrate in ground water.

Phosphorus is retained or immobilized in natural soil systems by the mechanisms of adsorption, chemisorption, precipitation, and biological uptake. Ground water monitoring studies and laboratory column studies indicate that very limited phosphorus transport to ground water occurs in aerobic, water-unsaturated soils, with reductions in total phosphorus content of effluent in soil ranging from 85 to 95% or more. Phosphorus transport to ground water is likely to occur, however, in coarse-textured, non-calcareous, sandy soils that are low in organic matter, or in shallow soils over fractured or solution-riddled bedrock.

Phosphorus derived from OSDSs has been detected above background levels in ground water adjacent to OSDSs under conditions of saturated flow, due to high water tables or high hydraulic loading rates, in numerous studies. However, phosphorus concentration in ground water is found to decrease with distance from OSDSs, because phosphorus is capable of undergoing sorption and precipitation within ground water (Ellis and Childs, 1973; Childs et al., 1974; Jones and Lee, 1977a,b, 1979). Very low concentrations of phosphorus in ground water may be sufficient to cause contamination of surface water (Holt et al., 1970). Documented cases of contamination of surface water by OSDS-derived phosphorus have been reported where OSDSs are located within close proximity [i.e., less than 100 to 150 ft (31 to 46 m)] to surface water, or where drainage tile or drainage ditches intercept ground water before phosphorus sorption, precipitation, or uptake is complete.

Natural soil systems provide relatively ineffective retention of chloride and sulfate anions, and limited retention of anionic detergent substances and sodium cations. Septic tank effluent can contain appreciable quantities of these chemical constituents. Chloride anions are highly mobile and are not adsorbed or exchanged in soil, so that reduction of chloride concentrations in ground water is by dilution. Soils possess a finite cation and anion exchange capacity and therefore have a more or less fixed capacity to remove sodium and sulfate ions from effluent. Reduction of sodium and sulfate ions from effluent by soil decreases with time as sorption sites become satisfied, resulting in increased levels of these ions in ground water. Aerobic, water-unsaturated soil conditions promote biodegradation and adsorption of anionic detergent substances such as linear alkylate sulfonate (LAS).

Several ground water monitoring studies have reported movement of chlorides, sodium, and sulfate from OSDSs into ground water under aerobic, water-unsaturated flow conditions, and movement of LAS under water-saturated flow conditions. Several water quality surveys have detected elevated chloride, sodium, sulfate, and LAS concentrations in ground water associated with OSDSs. Because dilution is the primary mechanism for reduction of these constituents, restricting OSDS density may effectively control levels of these constituents in ground water.

Contamination of ground water has resulted from disposal of household products containing toxic organic substances or heavy metals into OSDSs, and from treatment of OSDSs with "septic tank cleaners" that contain toxic compounds.

Indicator organisms are commonly enumerated in effluent and ground water because the task of detecting all types of bacteria is complex and costly. Counts of total coliforms, fecal coliforms, and fecal streptococci are thought to reflect the presence of human pathogens in effluent and ground water (Berg, 1978).

Fecal bacteria are removed or otherwise inactivated from effluent in soil by the mechanisms of filtration, adsorption, and natural die-off. The biological clogging mat or crust that commonly forms within the first few inches of the soil below an absorption trench or bed has been found to be an effective barrier to bacterial transport (Bouma et al., 1972). The removal of indicator organisms from effluent is also a function of the soil water/effluent flow regime. Transport of indicator organisms under water-unsaturated flow conditions is generally restricted to about 3.3 ft (1 m) (Brown et al., 1979; Hagedorn and McCoy, 1979). Movement of indicator organisms over much longer distances has been reported under water-saturated flow conditions (Romero, 1970; Viraraghavan, 1978). Several ground water monitoring studies and water quality surveys have reported contamination of ground water and surface water originating from OSDSs under conditions of saturated flow, high effluent loading rates, and shallow depth to seasonal high water tables or fractured, jointed, or solution-riddled bedrock. Bacterial contamination of wells by OSDSs is the second most common reason for well replacement in the southeastern United States (Miller and Scalf, 1974; Miller et al., 1977).

Viruses occur in effluent in varied concentrations that reflect the combined infection and carrier status of the residents utilizing the OSDS (Berg, 1973). Viruses are removed or otherwise inactivated in natural soil systems by the mechanisms of adsorption, filtration, precipitation, biological enzyme attack, and natural die-off. Laboratory column studies indicate that virus adsorption in soil generally increases with increasing cation exchange capacity, clay content, specific surface area, and ionic composition of the soil solution. Low soil pH, low soil moisture content, and low effluent loading rates also increase virus adsorption in soil.

Several ground water monitoring studies have reported transport of viruses to ground water from OSDSs under conditions of saturated or near-saturated flow, high water tables, or high effluent loading rates.

Movement of effluent through 36 to 48 inches (90 to 120 cm) of soil under unsaturated flow conditions is commonly deemed essential to prevent contamination of ground water and surface water. Flow of effluent through water-unsaturated soil results in increased travel time for contaminants, better effluent/soil contact for physical, chemical, and biological processes, and improved effluent treatment by the soil.

A 24 inch (60 cm) separation distance between the bottom of the adsorption system and the seasonal high ground water as required currently by Chapter 10D-6 of the Florida Administrative Code (State of Florida, 1983) may be inadequate to insure proper treatment of septic tank effluent and to prevent contamination of ground water. In-situ ground water monitoring studies and laboratory column studies support the maintenance of a minimum depth of 36 inches (90 cm) of water-unsaturated soil between the bottom of the absorption system and the seasonal high water table.

Conditions of water-saturated flow in soil, high hydraulic loading rates, and shallow depth to seasonal high water tables or highly permeable bedrock have provided documented cases where nitrogen, phosphorus, bacteria, and viruses from septic tank effluent have resulted in contamination of ground and surface water supplies. Proper functioning of an OSDS is achieved only if a sufficient volume of aerobic, water-unsaturated soil is available to absorb the volume of effluent and purify it before it reaches the ground water.

More than one half of the soils in the United States are unsuited for conventional OSDSs. Florida has a particularly high percentage of soils unsuited to conventional OSDSs due to conditions of periodically high water tables, low relief, and/or shallow depth to bedrock. Conventional OSDS can be modified, however, to improve effluent treatment and reduce the potential for ground water contamination. Proper siting, design, construction, and operation of OSDSs is the key to effectively controlling ground water contamination by the various effluent constituents.

8. RESEARCH NEEDS

Research is needed to evaluate the fate and transport of chemical and biological contaminants in OSDS effluent. Research strategies and activities should achieve the following:

(1) Monitor the fate and transport of nitrates, phosphates, heavy metals, toxic organics, pathogenic bacteria, and viruses under water unsaturated flow conditions in and around properly sited, installed, and operated conventional and alternative systems in an array of Florida soil, landscape, and climatic conditions.

A limited number of ground-water monitoring studies and water quality surveys have been conducted in Florida to assess the impact of OSDSs on water quality. One of the fundamental questions that needs to be addressed is "What is the current environmental impact of OSDSs in Florida?"

(2) Evaluate the adequacy of 24 inches (60 cm) of water-unsaturated soil above a seasonal high water table for effective removal of chemical and biological contaminants from effluent under Florida's soil and climatic conditions.

In-situ ground water monitoring studies and laboratory column studies support the maintenance of a minimum soil depth of 36 inches (90 cm) between the bottom of the absorption system and the seasonal high water table. The 24 inch (60 cm) separation distance between the bottom of the adsorption system and seasonal high ground water table may be inadequate to insure proper treatment of septic tank effluent in order to prevent contamination of ground water.

This research should include intensive monitoring studies of conventional OSDS installations that have been installed in conditions such that the seasonal high water table is 24 inches (60 cm), or nearly so, below the absorption trench or bed bottom. This phase of the proposed research should comprise the most intensive part of the overall monitoring program recommended in item (1).

(3) Determine densities and setback distances for OSDSs which are adequate to maintain a high degree of water quality and insure protection for public health in future years.

On-site sewage disposal system density is the primary mechanism for controlling concentrations of nitrate, chloride, sulfate, and sodium in ground water. Phosphorus, bacteria, and viruses derived from OSDSs have been found to travel considerable distances in soil under water-saturated soil conditions. Subsurface tile drainage systems, and drainage ditches and canals located adjacent to OSDSs, intercept effluent, reduce effluent-soil contact, and may result in contamination of surface water.

On-site sewage disposal systems may impact more severely on water quality in some parts of the state than in others due to differences in land-use, soil-water-landscape relationships, OSDS density, and aquifer recharge potentials.

Research in these areas should include both water quality surveys and mathematical analysis, including computer modelling, of the local and regional fate and transport of effluent constituents in a variety of soil, landscape, and land-use conditions.

(4) Improve on the current understanding of seasonal high ground water elevations and fluctuations for an array of Florida soils and landscapes.

Water table monitoring studies should be carried out with statewide centralized coordination. Monitoring sites should be selected to represent the most common soil and landscape types. Scientific analysis of water table and rainfall data should be carried out to predict water table response to variations in rainfall pattern, intensity, and amount at different times of the year. Refinement of understanding of the relationships between seasonal high water tables and soil/vegetative indicators should be undertaken as feasible.

(5) Determine adequate design criteria for alternative OSDSs with particular emphasis on proper selection and handling of fill materials for mounds and built-up lots.

Uniformity coefficient, grain size, texture, clay content, and lithology of fill material have been found to affect effluent treatment in mound systems and sand filters.

Water-unsaturated soil conditions and improved effluent treatment have been found in mound systems which distribute effluent by pressure-dosed distribution systems rather than by gravity systems. These phenomena need investigation in Florida. Also, development or refinement of Soil Conservation Service soil potential ratings for alternative systems such as mounds and built-up or filled lots would aid in proper siting and design of OSDSs.

(6) Determine the proper level of use, maintenance, and management of OSDSs to maintain optimal performance with respect to sewage treatment.

Lack of periodic maintenance can result in premature failure of OSDSs. Septage removal from septic tanks is necessary to maintain sufficient detention time of raw wastewater in the septic tank in order to provide effective pretreatment of effluent before discharge to the soil absorption system. Carry-over of wastewater solids to the absorption system can lead to clogging of the distribution system and infiltrative soil surface, and ultimately to system failure. Maintenance of OSDSs among homeowners is quite varied and in many cases is performed only after a "problem" with system functioning arises. Therefore, OSDS use, maintenance, and management, and their effects on surface and ground water quality, need investigation.

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