POTENTIAL FOR CONTAMINATION OF GROUND AND SURFACE WATERS FROM ON-SITE WASTEWATER DISPOSAL SYSTEMS

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TABLE OF CONTENTS

ABSTRACT

On-site wastewater disposal systems (OSWDS) treat wastewater from approximately 30% of the households in Virginia. Over 40,000 new OSWDS permits are applied for each year. Increases in OSWDS of this magnitude are accompanied by increasing potential for ground and surface water pollution. This review is the result of an extensive literature search regarding the cause and effect relationships between OSWDS and ground and surface water pollution. The research objectives were to determine if groundwater pollution from OSWDS is occurring in Virginia, determine the conditions under which degradation is most prevalent, and determine the areas of groundwater and OSWDS research that should receive priority. Over 500 publications were critically evaluated for scientific soundness in regard to methods and interpretations. Literature and research clearly document that Virginia ground and surface wasters are being polluted from OSWDS. The principal pollutants from OSWDS are N, bacteria, and viruses. The extent to which Virginia groundwaters are being polluted by these contaminants is currently unknown. Research is needed to document the extent of groundwater pollution attributable to OSWDS in Virginia. Groundwaters have the greatest potential for pollution if OSWDS occur in high density, or are placed in soils with high water tables and/or coarse textures. Areas within the state that are the most susceptible to pollution should be recognized. The maximum density of OSWDS these recharge basins can safely support in terms of groundwater pollution should be evaluated. Standards for separation distances between OSWDS and high water tables should be established based on sound scientific research. Alternative systems should be developed and evaluated to reduce nitrate levels before leaving an OSWDS.

INTRODUCTION

Virginia relies heavily on on-site wastewater disposal systems (OSWDS) to treat domestic wastewaters. Fifty-eight percent of the localities in Virginia have 60% or more households served by OSWDS (VDH, 1990). The Virginia Division of Health (VDH) receives over 40,000 new OSWDS permit applications each year. Annual increases of this degree strongly suggest that the number of Virginia households using OSWDS will only increase in the future. Centralized wastewater treatment plants in rural areas are not economically feasible due to the distance between households. Therefore, for rural areas, OSWDS is the only cost effective disposal technology available. Suburban areas are also becoming more dependent on OSWDS due to decreases in federal funding for construction of wastewater treatment facilities. In addition, due to the spread of development and the associated pressures, marginal soils are being used for OSWDS.

Increased use of OSWDS is accompanied by an increase in potential pollution to the ground and surface waters by these systems. Public concerns over health and potential degradation of water quality dictate that the use of OSWDS contribute as little as possible to these concerns. Numerous investigators however, have attributed declining water quality to indiscriminate use of OSWDS (Quan et al., 1974; Miller, 1975; Valiela and Costa, 1988; Mancl and Beer, 1982; Bauman and Schafer, 1985; DeWalle and Schaff, 1980; Brooks and Cech, 1979; Sandhu et al., 1979). In addition, 41% of the outbreaks of waterborne disease in the U.S. between 1971 and 1982 have been attributed to OSWDS (Craun, 1984). These studies indicate that pollution from OSWDS is more than a potential problem.

The extent to which OSWDS are contributing to pollution of the ground and surface waters of Virginia is unknown. In order to begin such an assessment, the existing data and research examining the effect of OSWDS on the environment needs to be examined. An evaluation of these data, as well as the associated research methods, will provide the necessary background information needed to: 1) determine if groundwater pollution from OSWDS is occurring in Virginia; 2) determine the conditions under which degradation is most prevalent; and 3) determine the critical areas of groundwater and OSWDS research that should receive priority.

With these goals in mind this literature review was initiated with the following objectives:

1) Critically review the scientific literature pertaining to degradation of ground and surface waters by OSWDS.
2) Identify the areas in which the most potential exists for degradation of Virginia waters as the result of OSWDS.
3) Recommend sampling and research programs to assess the effect of OSWDS on Virginia waters.

ON-SITE WASTEWATER DISPOSAL SYSTEMS (OSWDS)

Approximately 25% of the households in the U.S. use OSWDS to dispose of wastewater (US-EPA Staff, 1980). Most of these systems are conventional type systems using a septic tank and a subsurface soil adsorption field (ST-SAF). The use of OSWDS continues to increase at an alarming rate. Comparisons between the number of households on OSWDS in 1970 versus 1980 (Bureau of Census, 1972; 1983) shows a 26% increase.

Several soil factors effect the efficiency of an OSWDS and restrict the usage of these soils for wastewater disposal by conventional means. Most of these factors have bearing on the percolation rates of wastewater into soil, or the amount of soil available for treatment of the STE before groundwater is reached. Soils that are shallow to restrictive layers such as fragipans or bedrock, have extremely high or low percolation rates, or are subjected to elevated water tables should not be used for waste disposal by conventional means. Only 32% of the nation's soils however, are suitable for conventional OSWDS (SCS Engineers, 1983). These facts suggest that effective use and evaluation of alternative OSWDS may be as important as similar research and study of conventional systems.

CONVENTIONAL OSWDS

A conventional OSWDS consists of a septic tank and subsurface soil adsorption field (ST-SAF). The design and operation of conventional OSWDS have been adequately described by US-EPA Staff (1980) and Cotteral and Norris (1969). Septic tanks are buried, enclosed receptacles that receive the raw domestic wastewater. Domestic wastewater can be divided into either black (toilet wastewater) or grey (bath and sink wastewater) wastewaters, but most often both are combined in the same septic tank.

The purpose of the septic tank is to initiate wastewater treatment. Solids settle to the bottom of the septic tank. Anaerobic microbes decompose much of the suspended organic material, and utilize some of the nutrients such as N and P. Microbes in the septic tank also have an antagonistic role. These microbes compete with the enteric bacteria and viruses in the domestic wastewater, reducing their numbers prior to release of the STE.

Most conventional systems use gravity to move STE from the septic tank to the distribution box and drain lines. Distribution boxes divide the effluent among the perforated distribution drain lines buried in the ST-SAF. The STE finally moves through the soil where it is filtered, purified, and biodegraded.

The composition of STE varies depending upon the household (Table 1). The STE components with the most potential as ground and surface water pollutants are N, P, bacteria, viruses, and organics. Heavy metals, cations, anions, and surfactants may also be of some concern as pollutants in certain systems.

WASTEWATER MOVEMENT IN SOIL

The degree of purification and filtration of wastewater by soil is primarily dependent upon the hydraulic characteristics of a soil. Hydraulic character directly effects travel time, and thus wastewater-soil interactions. In general, the slower the wastewater moves through the soil the more likely soil-wastewater interactions will occur leading to increased filtration and purification. Bouma et al. (1972) and Otis (1982) thoroughly reviewed the theory and concepts of water movement in soils.

Movement of water through soil is dependent upon the type and distribution of voids in a soil, and the amount and frequency of water applied. Void type and distribution is directly related to the arrangement of soil particles. The arrangement of soil particles and associated voids is termed, soil structure (Brewer, 1976).

Table 1. Characteristics of Septic Tank Effluent (STE) from 8 Studies. All Concentration are in Mg L^{-1} Except for pH, Bacteria (FC, TC) and Coliphages. Bacteria and Coliphage Concentrations are Expressed in 10^5/100 ml and Coliphage 100 ml^{-1}, respectively.

	Laak, 1970	Brandes et al., 1975	Brooks et al., 1984	Brown et al., 1978	Cogger et al., 1988	Effert et al., 1985	Mitchell et al., 1982	Otis et al., 1975
	n^*=1	n=1	n=3	n=1	n=1	n=1	n=1	n=6
pH	8.0	--	--	7.4	--	--	--	--
DO	0	--	0	--	--	0	--	--
BOD	189	122	258	--	--	127	--	158
COD	585	233	641	--	--	310	--	358
TOC	129	--	--	94.9	--	--	260	--
SS	--	84.8	--	--	--	53	270	--
TSS	124	--	168	--	--	--	--	55
Total-P	--	10	20.5	--	--	19.5	12	15.3
Total-N	--	--	60.8	29.8	--	41.5	--	52.2
NH_4^+	--	21.6	49.2	24.7	28	--	16	35.6
Organic-N	--	5.0	11.5	5.8	--	--	--	--
Na	--	--	--	--	174	--	--	--
Cl-	--	--	--	--	97	--	--	--
K	--	--	--	--	18	--	--	--

*n = number of septic tanks sampled.

Soil structure can be divided into primary and secondary components. Primary structure is related to the arrangement of primary soil particles (i.e., sands, silt, and clay) and the voids that occur between the particles (packing voids). Packing void size depends on the size of the primary particles. Sand particles have the largest packing voids, and clay particles have the smallest.

Secondary structure is the arrangement of soil aggregates, called peds, and the voids that separate these peds. Voids associated with secondary structure include vughs, channels, and planar voids (Brewer 1976, Bullock et al., 1985). Vughs are small (<1 mm) irregularly shaped discontinuous pores within peds. Channels can be continuous or discontinuous and occur within a large ped, or form the natural boundary between two peds. Most planar voids are continuous and form the boundary between two large soil peds. (Most soil studies are only concerned with secondary structure. Therefore, the term soil structure is synonymous with secondary soil structure, and soils without secondary soil structure are often termed structureless.)

Water retention in a soil is governed by the adhesive and cohesive forces between water molecules and soil particles. These forces are often termed the matric potential. Smaller voids hold onto the water tighter, or with greater tension. Therefore, soils which have primarily packing voids or small vughs hold onto water with greater tension than a soil with many large channels or planar voids. In a similar manner, the thinner the layer of water molecules around a soil particle or ped, thegreater the tension on those molecules. These physical laws dictate that insimilar soils, water will move faster through soils with higher moisture contents.

Several examples may be the best way to illustrate these concepts. Two soils are saturated with water so that all voids are filled. Secondary structure is absent from both soils, however, one soil has a clay texture and the other is a sand. As the soils drain, the sand soil will drain faster and retain less water than the clay. These results occur because the larger packing voids in the sand offer less resistance to flow and retain less water than the clay textured soil. In the second scenario, two soils of the same texture are completely saturated with water so that all voids are filled. One soil has only packing voids. The other soil has a strongly expressed structure which is exhibited by a large number of large and continuously linking channels or planar voids. If the soils are allowed to drain, the soil with well developed structure will drain with less resistance and to a greater extent because water will not be held as tight in the larger voids. The soil with the well developed structure will also drain much faster. This occurs because the path between two points in the soil with continuous or connected channels and planar voids is much shorter and less tortuous, than the soil with only packing voids.

Water moves in a soil as a function of the difference in potential energy between two points. Gravity (gravitational potential) and soil water tension (matric potential) are the forces that contribute the largest portion of the potential energy (potential gradient) governing the flow of water in a soil (Otis, 1982). The flow of water in a soil follows

Darcy's Law:

$$Q = KA\,(dh/dz)$$

where:
Q = flow rate;
K = hydraulic conductivity;
A = cross-sectional area;
dh/dz = potential gradient.

Hydraulic conductivity (K) is a constant for a given soil moisture tension, and is characteristic for an individual soil. Thus, hydraulic conductivity is the best parameter for comparing the rates of water movement through soils.

Hydraulic conductivity varies depending upon the soil moisture tension (matric potential) of the soil (Figure 1). All soils have the highest conductivity at zero moisture tension (saturated conditions). As the soils dry, the hydraulic conductivity decreases. Therefore greater movement of pollutants will occur in soils with saturated conditions.

Numerous studies have examined the relationships between various soil properties and characteristics and the hydraulic conductivity of a soil. King and Franzmeier (1981) examined the morphology and saturated hydraulic conductivity (K-sat) of horizons from 28 soils. Hydraulic conductivity data were grouped based on texture, mode and origin of parent material, and horizon development. Texture was the most important factor. Figure 1 illustrates the effect of texture on conductivity of a soil. Saturated hydraulic conductivity (K-sat) is greater for coarser-textured soils (sands) than finer-textured soils (clays).

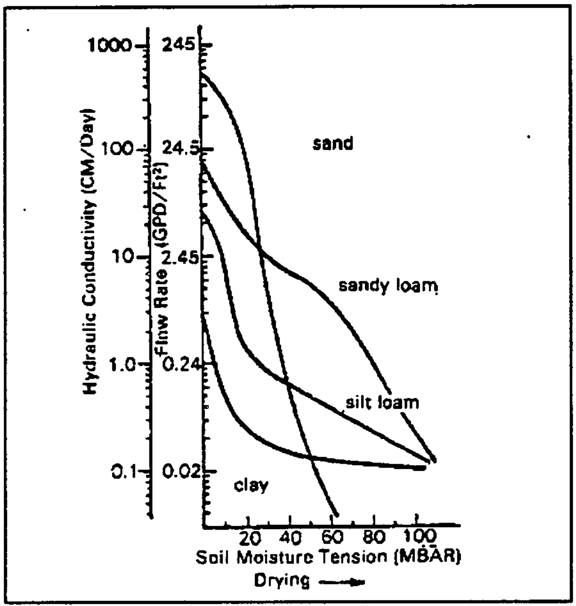

Figure 1. Hydraulic conductivity vs soil moisture retention (after Bouma, 1975)

Puckett et al. (1985) examined the physical soil properties effecting hydraulic conductivity. The soils examined were from the lower coastal plain of Alabama. The K-sat of a soil could be best estimated using the percent clay of the sample. Porosity and bulk density values showed correlations coefficients < 0.7.

Bouma (1975) examined the hydraulic conductivity of 12 established ST-SAF with textures ranging from coarse sand to clay. These ST-SAF could be grouped into four hydraulic conductivity classes according to texture. These classes were: 1) sands; 2)sandy loams and loams; 3) silt loams and some silty clay loams; and 4) clays and some silty clay loams.

Although texture is important in explaining the hydraulic properties of a soil, several studies have indicated that structure can be equally important. McKeague et al. (1982) divided 78 A and B horizons into 8 conductivity classes based on structure, porosity, texture, consistence, and density. Measurements of K-sat for the 78 horizons indicated that 45% of the horizons were placed into the correct class, and 87% were correctly placed in appropriate class, or one class away. Macroporosity and soil structure were controlled by K-sat of many soils. Horizons with textures finer than loamy fine sand showed little relation between texture and K-sat.

Bouma et al. (1977) examined water movement through several soils. Soil thin sections were prepared after applying a methylene blue tracer (dye). The tracer indicated that water moved through vughs, channels, and planar voids. The highest K-sat values occurred in soils in which macropores contribute most to the flow. Thin sections indicated that flow occurs in macropores even when soils are not completely saturated.

Kanchanasut et al. (1978) examined movement of water through soils with large voids. In saturated conditions, most movement of water occurred along channels as indicated by concentration of indicator dye. When these channels were plugged flow rates were reduced by 99%.

Southard and Buol (1988) investigated the relationships between K-sat and selected soil properties. Samples were collected from the subsoil of several middle coastal plain soils in North Carolina. A decrease in K-sat was observed with depth. A high correlation ($r^2 > 0.81$) was found between K-sat and pores > 30 u. Pore sizes were determined by water retention characteristics. No correlation between K-sat and estimated ped size or grade was observed because the soils examined were of weak structure.

Anderson and Bouma (1977a, 1977b) studied water movement through soils with silty clay loam textures and either medium subangular blocky or coarse prismatic structure. High hydrodynamic dispersion occurred in both soils, but the effect was especially evident in the soil with subangular blocky structure which had been initially drained. Hydrodynamic dispersion is the movement of water through relatively large pores around slowly permeable peds containing "stagnant" water. Conditions conducive for hydrodynamic dispersion often occur in ST-SAF as a result of intermittent loadings. Hydrodynamic dispersion is often termed by-pass or short circuit flow because wastewater moves around, instead of through, soil material with small pores. Wastewater moving around soil with small pores by-passes potential treatment. The effect of hydrodynamic dispersion suggests that structure plays an important role in water movement in a ST-SAF.

Germann et al. (1984) studied the movement of water in undisturbed columns of soil 45 to 50 cm long with a Br^- tracer. Water reached the 47 cm depth within one day in soils with minimal macropores and low applications of water. In soils with well developed macropores and water application at medium rates, water will reach the 47 cm depth within 43 minutes. Water applied at low rates to a soil with well developed macropores reached the 45 cm depth within 3 hours. Dye studies indicated that the continuity of the pores influenced the depth to which water moved. Little movement of water was observed from the macropores into the soil matrix in surface horizons. With depth however, movement of water into the soil matrix increased.

Hansen and Mansell (1986) simulated unsaturated and saturated flow from a septic tank drain to the groundwater in fine sand and sand soils. Travel times for 60 cm in unsaturated conditions were as short as 3.7 hours.

Steenhuis and Muck (1988) examined movement of Cl^- and NO_3^- following rainfall events. Soil solutions moved through both macro and micro pores. Water movement through macropores was much faster. More water moved through macropores as moisture content of the soil increased. Steenhuis and Muck (1988) suggested that when application or loading rates are above infiltration rates of micropores, movement of water will occur in macropores. Similar findings were presented by Quisenberry and Phillips (1979), McMahon and Thomas (1974), Tyler and Thomas (1981), and Cassel et al. (1974).

Clogging effects

The formation of a clogging mat, sometimes referred to as crust, is universal to all ST-SAF. The clogging mat is a layer of organic material that accumulates at the interface between the gravel envelope around drain lines, and soil receiving the STE.

Clogging usually occurs within the first several months of operation of a ST-SAF (Kristiansen, 1982a). A well developed clogging mat often forms a restrictive layer in a ST-SAF leading to reduced flow rates or hydraulic failure (ponding) of the system. Numerous researchers have investigated the genesis and associated effects of clogging on the performance of an ST-SAF. Kristiansen (1982a), Miller and Wolf (1975), and Otis (1985) have reviewed these investigations and discuss the cause and effect relationships associated with clogging of a ST-SAF.

Clogging occurs in two distinct phases. The initial phase is indicated by a rapid decrease in infiltration rates in the ST-SAF. This phase is followed by a much slower decline in infiltration rates. The second stage of clogging may reach some form of equilibrium in which changes in infiltration rates over time are at a minimum (Kristiansen, 1982a). There are several schools of thought on clogging mat genesis. The most common explanations are: clogging is the result of microbial growth; clogging is the result of physical accumulation or deposition of organic material in the STE; or a combination of both of these explanations.

Clogging has been observed in soils in which only water had been applied. Allison (1947) showed that additions of water to soil reduced the permeability. In soils sterilized however, infiltration rates remain fairly constant. Allison (1947) concluded that the by-products of microbial activity such as cells, slime, and polysaccharides, plugged the soil pores and reduced permeability.

McCalla (1950) examined the effect of microbial growth on percolation rates in soil columns. Addition of sucrose increased microbial growth and reduced percolation rates. Soils maintained at 7°C showed higher infiltration rates than those at 27 to 32°C. Additions of $HgCl_2$ increased infiltration rates by acting as a disinfectant.

Avnimelech and Nevo (1964) examined clogging in columns filled with sand and various types of organic matter. Water or N solutions were added to the columns and infiltration rates were recorded. Columns with reduced C:N ratios showed less clogging. Clogging was positively correlated with polyuronide concentration, and therefore a function of polysaccharide production. Production of polyuronides and polysaccarides occurred as organic matter was degraded during microbial activity.

Kristiansen (1981a) investigated relationships between microbes and clogging in 3 sand filters. Ratios of C:N were similar in the clogging mat and sand filter suggesting that polysaccharides were not the cause of clogging. Kristiansen (1981a) suggested that bacteria cells may be important in clogging by producing slime that would link soil particles together and thereby restrict flow. In studies using the same sand filters, Kristiansen (1981c) found that in the most clogged systems, biomass was correlated with C content (r=.999). In the least clogged filter however, biomass and C content were not correlated (r=.11).

Several studies have investigated the effect of organic content, or concentration of SS, on clogging mat formation. Kristiansen (1982b) monitored infiltration rates within sand filter columns after adding either 5 cm per day of gray water, or 5 cm per day of STE. The COD content of STE and grey water were similar, but the grey water had a much smaller SS content. The reductions in infiltration rates were much smaller for the grey water column over a 1 year period suggesting the importance of SS in explaining clogging. Siegrist (1987) investigated the effects on varying the loading rate and type of wastewater on clogging mat development in a ST-SAF. Increases in loadings reduced infiltration rates, and at a more rapid pace. The BOD levels in STE were 2 times greater than BOD concentrations in grey water. Systems in which only grey water was applied showed higher infiltration rates than systems in which STE was applied. Grey water systems did not show significant clogging for 12 to 28 months depending upon loading rates. Macro and micro morphology of clogged, but unponded systems showed a < 5 mm layer of black material covering the soil surface. Soil 15 cm below the black material was reduced (grey colors, 5Y 6/1). Black material was observed at the soil surface of ponded systems, as well as coating the soil peds. Ped coatings were < 1 mm thick and composed of amorphous organic material that was either deposition from STE or synthesized by microbes. Polysaccharide concentrations were similar for soils regardless of the stage of clogging, suggesting that polysaccharides were not responsible for clogging. Siegrist (1987) concluded that "humus" formed under anaerobic conditions filled pores and resulted in clogging.

Daniel and Bouma (1974) compared the degree of soil clogging in columns treated with STE, aerated STE, and water. Undisturbed cores 60 cm in length were used in the column studies. The highest moisture tensions (matric potentials) were recorded for the columns treated with aerated STE. These columns also showed the quickest reduction in Eh, suggesting that columns treated with aerated STE were the most clogged. Equal amounts of suspended solids occurred in STE and aerated STE, but solids in aerated STE were smaller in size. Daniel and Bouma (1974) concluded that the smaller aerated STE particles moved farther in the columns, and therefore clogged more of the continuous pores.

Laak (1970) examined the effect of aerobic pretreatment of domestic wastewater to reduce clogging in an OSWDS. Columns were filled with 30 cm of 3 different soils. Either STE or aerated STE effluent were applied to the columns. A greater degree of clogging occurred in columns treated with STE than those columns treated with aerated STE. Amounts of BOD and TSS were greater in STE than aerated STE. In addition, the sizes of the SS particles were smaller in the STE than aerated STE. Laak (1970) concluded that the higher concentrations of BOD and TSS in the STE increased clogging development, and the smaller suspended solids clogged more of the continuous pores. These results and conclusions contradict those of Daniel and Bouma (1974), and suggest that the effect of aerobic treatment on clogging is dependent upon the individual aerobic unit.

Rice (1974) examined clogging of 50 cm columns filled with sand or loamy sand in which STE was applied. Clogging occurred mainly at the surface with hydraulic conductivity of the rest of the column changing little during inundation with STE. Similar findings were reported by De Vries (1972).

Magdoff and Bouma (1975) investigated the formation of the clogging mat in a sand filled column in which STE was applied. Columns were constructed to simulate a mound system. Calculations and measurements of clogging mat resistance and infiltration rates suggested that clogging occurred at the surface. Sand filter-clogging mat morphology and tensiometric data however, suggested that clogging occurs inside pores.

Kristiansen (1982a) after reviewing the literature concluded that clogging is initiated by an accumulation of suspended solids. These suspended solids are linked together with soil particles by biologically produced material during the second stage of clogging.

Factors such as temperature, loading rates, and aerobic or anaerobic conditions show varying effects on clogging. Kristiansen (1981a) investigated the relationships between temperature, loading rates, and clogging in 3 sand filters. One filter was maintained at temperatures between 12 and 15°C and loaded with 4-6 cm/day of STE. The other 2 filters were maintained in ambient air temperatures (4 to 16°C), and loaded with STE at 4 to 6, or 12 to 18 cm/day. The clogging mat was better developed in the sand filter maintained at the higher temperatures (12-15°C) than the mats developed in filters at ambient temperatures even though the loading rate in one of the filters was greater.

Jones and Taylor (1965) examined the infiltration of STE through sand columns. A more restrictive clogging mat formed under continuous flow than under intermittent flooding and aerobic conditions. Clogging was due to physical clogging of pores from organic matter. Increasing loading rates increased ponding. Clogging occurred 3 to 10 times faster under anaerobic conditions.

Simons and Magdoff (1979a) examined the effect of loading rates, anaerobic conditions, and temperature on the degree of clogging of columns constructed to simulate mounds. Columns in which STE was applied at 17.5 cm/day were the first to clog and pond. Columns in which loadings were maintained at 1.2 cm d^{-1} did not pond during the 511 days of the experiment. Lower temperatures and anaerobic conditions increased clogging. Anaerobic conditions reduce organic matter decomposition and thereby increase clogging. Inducing aerobic conditions by resting a ST-SAF may therefore, reduce clogging.

De Vries (1972) investigated the effect of resting a system to reduce clogging. Columns constructed to simulate sand filters were monitored for O_2 and CO_2 changes at various depths following addition of STE. Concentrations of O_2 and CO_2 did not stabilize within the columns until at least 22 hours had passed, signaling that BOD components had been oxidized. These data suggest that resting periods of at least 22 hours are necessary to maintain aerobic conditions and reduce clogging. Simons and Magdoff (1979a) found that longer periods of rest were required for columns with the most clogging, before infiltration rates would increase. Otis (1985) suggested that a ST-SAF should be rested at least 6 months and include summer months for an effective reduction in clogging. Magdoff and Keeney (1975) examined clogging in columns representing mound systems. Columns were clogged within 104 days. Columns were dried for 2 months and STE was reapplied. Columns clogged again after 89 days.

Clogging mats have both beneficial and detrimental effects in an OSWDS. Hydraulic failure occurs in systems in which the clogging mat is too restrictive. In systems in which the clogging mat is poorly developed treatment efficiency of the system is sacrificed. The clogging mat increases treatment efficiency by filtering organic matter and microbes as well as controlling saturated flow. Therefore, the clogging mat is important in the purification process of OSWDS (Otis, 1985; Kristiansen, 1981c, 1982a; Laak, 1974).

The effect the clogging mat has on flow through a soil is dependent upon the capillary properties of the underlying soil. Capillary properties are governed by both soil texture and structure. Bouma (1975) reviewed the effects of a restrictive barrier such as a clogging mat on flow through soils. As barrier resistance increases, the infiltration rates of the barrier decrease, and water tensions below the barrier increase. Identical barriers produce different tension effects if soils are of different texture or structure. Increasing the hydraulic head on top of a barrier will only slightly increase flow rates and reduce tensions in the soil. Soils with low permeabilities such as poorly structured clays show only minimal effects from barriers with small resistances.

Janni et al. (1980) simulated flow rates through four different soils in which STE was applied. The objectives of the computer simulations were to determine the various effects of texture, clogging mat, and soil moisture conditions on flow rates. Clogging effected flow rates in all but the clayey soils. The simulation indicated that a well developed clogging mat could totally restrict flow in a permeable soil.

Bouma (1975) determined the conductivity of 12 established ST-SAF with textures ranging from coarse sand to clay. Restrictive clogging mats drastically affected hydraulic conductivity of a soil. In similar studies, Bouma (1971) observed that unsaturated flow occurs in most ST-SAF because of clogging, and in situ conductivity as a function of soil moisture tension (matric potential) should be used to determine the ability of soil to accept STE. The conductivity will depend on loading rates, period of application, and resistance of clogging mat. Bouma (1971; 1975) concluded that percolation tests do not adequately describe movement of STE through a clogging mat and into the soil below the mat.

ALTERNATIVE SYSTEMS

Many soils are unsuitable for conventional OSWDS because of high water tables, shallowness to bedrock, extremely slow or high percolation rates, or restrictive layers at shallow depths such as fragipans. Several alternative systems have been developed to make these sites more suitable for domestic waste disposal. The US-EPA Staff (1980) reviewed the design and operation of several of the alternative systems commonly used.

Mounds

Mound systems are by far the most studied as well as used alternative OSWDS. The design, construction, and operation have been described by Hoover et al. (1980), US-EPA (1980), Converse and Tyler (1985), Converse et al. (1978), and Bouma et al. (1975). In general, mound systems consist of a septic tank, a pumping chamber, and the elevated sand mound. The pump is used to move the STE above the ground to the elevated mound as well as assure equal distribution of STE within the drain lines of the mound. The mound is built on top of the natural soil surface. At least 60 cm of sand separates the bottom of the drain lines from the original soil surface. Drain lines are covered with sand and a protective soil cap is placed over the sandy fill to promote vegetative growth and reduce infiltration of precipitation falling onto the mound. The cross-sectional area of a mound is generally the same size as a conventional ST-SAF. Berms are often placed around mounds to redirect surface water away from these systems. Properly designed mound systems can provide excellent treatment of STE. One of their major drawbacks however, is the high cost (2 to 3 times more than conventional systems) of construction of mound systems (Carlile, 1980; Bouma et al., 1975).

Flow rates within mounds must equal or exceed effluent loading rates, or hydraulic failure of the system will occur (Mott et al., 1981). Flow rates can be determined for wastewater within a mound if several parameters are known. These parameters include: the K-sat of the mound; K-sat of the most restrictive layer of the soil, depth to restrictive layer; thickness of the most restrictive layer; slope, dimension, and shape of the mound; depth to the water table; and cross sectional area of wastewater flow (Mott et al., 1981). The effect of clogging on the system should also be considered in the loading rate design for mound systems. Simons and Magdoff (1979) constructed columns to simulate mound systems and applied STE at loading rates of 17.5 cm/day, 2.2 cm/day, and 1.2 cm/day. All columns loaded at 1.2 cm/day, and 2 out of 3 dosed at 2.2 cm/day, did not form a restrictive clogging layer within the 511 days of the study.

Surveys analyzing the operation of mounds have been conducted in the states of Virginia, Wisconsin, and Pennsylvania. Converse and Tyler (1985) surveyed the effectiveness of over 40 mound systems in Wisconsin. Mounds were operating properly on filled sites, sites with high water tables, slopes up to 21%, and on slowly permeable soils. Most systems offered quality STE treatment. Mound systems on severely limited site conditions ponded and leaked during wet weather. Some systems on steep slopes leaked at the toe portion of the mound.

Alexander (1991) presented results from a survey conducted by the Virginia Department of Health on the operation of 46 mounds. Twenty-four percent of the mounds at the time of the survey were failing (as defined by ponding or leaking at the surface). Most mounds that were failing (73%) showed evidence of erosion of the mounds protective cover. Erosion may indicate that failing mounds were not carefully constructed.

Hoover et al. (1980) surveyed the operation of mound systems in Pennsylvania. About 50% of the mound systems constructed were failing, or had failed. Only 6.5% of the home owners surveyed however, reported failing systems. Twenty-six of the 36 failing sites had water tables within 75 cm of the soil surface. Some systems were failing at the downslope side of the system. The opposite side of the mound remained unused, suggesting that equal distribution of effluent was not occurring. These failing systems had distribution boxes, but did not have pressurized distribution.

Low pressure distribution

Conventional OSWDS rely on gravity to distribute the STE onto the adsorption field. In most conventional OSWDS STE is poorly distributed, and an area of the adsorption field is overloaded with STE. Overloading induces saturated flow within the soil and leads to poor treatment of the wastewater. Equal distribution of effluent is especially important in sandy soils because of their poor treatment ability (Converse et al., 1975). Low pressure distribution (LPD) is a fairly effective method to distribute STE evenly in an adsorption field (Carlile, 1980; Simon and Reneau, 1985; Converse, 1974). These systems have been successfully used in soils with high water tables, soils with restrictive horizons, soils with rapid permeability, soils on steep slopes, and in fill material or beds (Hargett, 1985).

There are over 300 different designs of LPD systems (Carlile, 1980). US-EPA Staff (1980) described the design, construction and operation of a commonly used LPD system.

In general, a pump is used to assure that even pressure occurs in each of the distribution lines. Distribution lines are smaller in diameter than conventional drain lines and often shallowly placed to take advantage of the microbial active and aerobic soil material (Hargett, 1985). Pressure from the pump forces STE through each line so that equal amounts are released across the entire ST-SAF during each dosing. Converse et al. (1975) examined the effectiveness of five LPD systems over a two year period. Most systems did not show a complete distribution of effluent because the dosing pump was smaller than was required.

Several studies have investigated the relationship between evenly distributed STE and clogging mat development. Simon and Reneau (1985) evaluated the performance of LPD systems in soils with slow percolation rates. Tensiometric data indicated that a restrictive layer in the form of a clogging mat had not formed in the distribution trenches. These data suggested that effluent was equally distributed within the ST-SAF.

Harper et al. (1982) examined the performance of 3 filter fields constructed over fragipan soils, and dosed using LPD. Two of the filter fields showed no effect due to clogging and the systems were hydraulically working well. The other site showed evidence of ponding and therefore, clogging.

Bouma et al. (1974) examined movement of STE from distribution trenches in a silty clay loam soil. Effluent was applied with by LPD. Tensiometric data suggested that trenches did not pond during the 10 months of the study indicating that a restrictive clogging mat had not formed. Data from similar soils in which STE was applied with gravity distribution had shown clogging within 10 months.

Aerobic units

Some OSWDS use aerobic units instead of septic tanks to treat domestic wastewater initially. Aerobic units that thoroughly oxidize wastewater can provide high quality effluent. Much of the oxidation and treatment is related to high populations of microbes (US-EPA Staff, 1980). Microbes in aerobic units convert organic matter to stable compounds such as CO_2, H_2O, or NO_3^-, or new cells. About 50 to 60% of organic matter added to an aerobic unit is utilized for cell synthesis (Hutzler et al., 1978). These microbes create more sediment (sludge) than conventional septic tanks increasing the number of times sludge must be removed during the lifetime of the system. Aeration is maintained by force air or mechanical mixing. Therefore, if these units are not properly maintained, low quality effluent will be produced (Hutzler et al., 1978; US-EPA, 1980).

Several studies have reported that variability in effluent from aerobic units is much higher than variability associated with septic tanks carrying similar loads (Otis et al., 1975a; Sauer and Boyle, 1978; Bennett et al., 1975b). Variability in aerobic unit effluent quality can be related to maintenance problems, home owner neglect, or wastewater surges. Hutzler et al. (1978) examined the operation and efficiency of 36 aerobic units in Maine. Fifty-five percent of the units surveyed were operating properly with effluent meeting standards. Twenty-one percent of the units were mechanically operable but were not meeting standards. Variability related to wastewater surges occurs because at certain times of the day water use in a household is drastically increased (Bennett et al., 1975a).

The justification of additional cost of an aerobic unit over a standard septic tank has been explored by several authors and researchers. Locker et al. (1980) presented several case studies where conventional OSWDS hydraulically failed and ponded at the surface due to clogging. Replacement of these systems with aerobic units reduced clogging, and systems operated without hydraulic failure (ponding).

Mitchell et al. (1982) observed that columns treated with aerobic unit effluent showed no ponding effects for a longer period, than columns treated with STE. Mitchell et al. (1982) concluded that use of aerobic units can prolong the life of an adsorption field.

Otis et al. (1975b) and Bennett et al. (1975b) compared the efficiency of septic tanks and aerobic units on treatment of domestic wastewater. Both studies indicated that effluent from aerobic units was not of high enough quality to be surface applied.

Effert et al. (1985) examined the efficiency of sand filters to treat effluent from an aerobic unit and STE. Water quality of effluent from the sand filter was similar for both STE and aerobic effluent. Effert et al. (1985) concluded that pretreatment with an aerobic unit prior to treatment with a sand filter was not justified.

Sand filters

Sand filters are commonly used to treat STE in the primary or final stage. Sand filters are beds of sandy granular material 61 to 91 cm deep that are underlain by gravel and collecting tile. Effluent is intermittently applied to flood the entire surface of the bed. Filters can be open or buried in the ground (US-EPA Staff, 1980; Hines et al., 1978). Effluent applied to sand filters should have low suspended solids. Therefore, retention time in the septic tank should be sufficient to assure maximum removal of suspended solids. Surface waters should be diverted away from sand filters so that only wastewater moves through these systems (Hines et al., 1978). The effectiveness of sand filters in removing pollutants from STE has been thoroughly examined by Pell and Nyberg (1989a, b, and c) and Kristiansen (1981a, b, and c). Hines et al. (1978) concluded that properly designed operating sand filters can produce effluent of high quality.

Electro-osmosis

Electro-osmosis (E-O) systems have been used in some soils in which percolation rates are low. This patented process is designed to increase the infiltration rates of soils by moving water away from the ST-SAF. In general, E-O systems were designed to function as a galvanic cell (Beer et al. 1978; US-EPA Staff,1980). The anode, constructed from rock, is placed adjacent to the adsorption trench. The cathode, constructed of coke and graphite, is installed some distance from the adsorption trench. Water is theoretically thought to move with the current to the cathode, and away from the ST-SAF. These systems were developed for use in soils with percolation rates slower than 24 min cm^{-1} (60 min/in) that do not have high water tables (US-EPA Staff, 1980; Anderson, 1980).

As of 1980, over 200 E-O systems were in place and operating mostly in Wyoming, California, and Minnesota. Most were placed in soils with percolation rates > 24 min cm^{-1} (Anderson, 1980). Although the research examining E-O systems has been minimal, Anderson (1980) concluded that E-O systems are proven alternative systems for soils with slow percolation rates.

Research conducted by Beer et al. (1978) and Effert and Beer (1985) dispute Anderson's conclusions about E-O systems. Beer et al. (1978) designed a laboratory model E-O system to investigate the possibility that these systems can increase the hydraulic conductivity of a soil. The laboratory model did not show an increase in internal drainage within the system. Effert and Beer (1985) examined the effectiveness of E-O systems on increasing the infiltration rate of two ST-SAF with moderate limitations due to low percolation rates. The ST-SAF studied were used for wastewater disposal by conventional means until the system began to pond due to limited infiltration. At this point E-O systems were installed. No effect was observed in the infiltration rates after the installation of the E-O systems, and the systems continued to pond.

Evapotranspiration

Evapotranspiration (ET) systems are designed to remove excess waste water through the processes of evaporation and transpiration. Evapotranspiration systems can be closed to seepage by using a plastic or clay liner, or left open so that percolation as well as evapotranspiration occurs. Wastewater is applied subsurfacely to the sandy bed of the ET system. Water moves to the soil surface by capillary action and is either used by plants or evaporates into the atmosphere (US-EPA Staff, 1980; Hines et al., 1978). How well an ET system functions is dependent upon: 1) climate; 2) loading; 3) sand bed physical properties; and 4) cover vegetation (US-EPA Staff, 1980). Properly operating ET systems must evaporate or transpire more water than is applied as waste or collected during precipitation. In order for this to occur, wastewater should contain > 3.5 mg L^{-} dissolved oxygen for plant roots, the evapotranspiration bed must be shallow (40-60 cm) so that plant roots reach the wastewater, and the evapotranspiration bed must be of sufficient size (Bernhart, 1978).

Over 5000 ET systems were in use as of 1977 in the U.S. Most of these systems are located in the western U.S. because ET systems are generally considered an unsuitable alternative OSWDS in areas of moderate or high rainfall (Hines et al., 1978). Several case studies have appeared in the literature discussing the operation and suitability of ET systems for wastewater renovation. Rowe (1980) briefly discussed the design and operational aspects of a non-seepage ET system in South Carolina. This system functioned very erratically during the two year study. Hydraulic failure and overflow occurred during most of the study period. Rowe (1980) concluded that ET systems are of limited use in the southeastern U.S. Bernhart (1978) however, presented 3 case studies of effective use of ET systems on the East Coast, and suggested that with proper design ET systems could be used in these areas.

Gunn (1988) described the design and operation of an ET system in New Zealand. This system was rather large and served a marine research laboratory. The system was reported to work without failure. Gunn (1988) concluded that ET systems function best in temperate climates with long growing seasons. Application rates should be low, and drainage of surface and groundwater should be controlled.

Spray irrigation

Spray irrigation is commonly used to dispose of STE, but the use of spray irrigation for OSWDS has been limited. After reviewing the alternative systems used to assure that distribution of effluent is at a maximum, Carlile (1980) concluded that spray irrigation is the most effective system for applying wastewater to a surface. Spray irrigation assures that all soil is utilized for filtration or treatment of the effluent. Prior to spraying, STE must be processed through an aerobic unit or sand filter to remove odors. Flanders (1975) described a spray irrigation system used to dispose of domestic wastewater in Vermont. The system sprayed domestic wastewater on soil shallow to bedrock or fragipan. Hathaway and Mitchell (1985) described the use of a sand filter - spray irrigation system to treat STE.

TYPES OF STUDIES

The fate and transport of various components in STE have been studied in both the field and laboratory. Laboratory investigations can be grouped as either batch or column type studies. Most batch studies are performed in a closed system. A given amount of a STE compound is mixed with a fixed amount of soil and allowed to react. Reaction times vary depending on the experiment and may range from minutes to days. Environmental conditions are easily maintained in batch studies so that parameters such as redox potential (Eh) and temperature can be held constant throughout the experiment.

Batch studies are commonly used to examine the interactions between effluent constituents and soils. These interactions may lead to adsorption of the compound to the soil, or a transformation of the compound to another form. Adsorption of P or viruses to soils is often studied in batch experiments. These type studies are used to determine the amount of P or viruses adsorbed to the soil, and the rate at which the adsorption reactions occur (adsorption kinetics). Batch studies under closed conditions provided the easiest means to study transformations in soils. Because of the complex nature of N in the soil system, transformations of N are often studied in batch type experiments where Eh, organic carbon content, and temperature can be easily monitored or maintained.

Column studies are designed primarily to simulate field conditions. Under simulated field conditions many of the important factors and parameters that effect interactions in the soil can either be maintained at a constant (such as temperature or soil moisture content), or easily monitored during the study period. Columns are semi-closed systems that are theoretically finite. Therefore, movement of the STE constituents added to soil columns are much easier to monitor in comparison to similar field experiments. Adsorption and transformation investigations are also conducted in soil columns.

Column studies can be grouped into two types depending on the degree of disruption the soil used for the experiment has experienced. Disturbed columns are constructed by packing ground, and often sieved, soil into a cylinder of various lengths and diameters. Undisturbed soil columns are constructed by obtaining an intact soil core from the field and encasing the sample in some manner to provide an undisturbed soil profile. Columns can be maintained at varying moisture conditions by applying tension at the bottom of a column. The effect of saturated conditions may also be examined by maintaining a constant head above a saturated column. Soil solution samples are usually collected over a period of time from the bottom of the column (outlet). These samples are often termed column effluent. Concentrations of various compounds in the column effluent are compared to those concentrations in STE, to estimate the efficiency of a soil to renovate wastewater. The distribution, rate of movement, transformation, or degree of reduction of a certain constituent may also be monitored by installing ports along the length of the column so that soil solution samples can be collected at various depths.

Numerous studies (Bitton et al., 1978; Kanchanasut et al., 1978; Cassel et al., 1974; McMahon and Thomas, 1974; Smith et al., 1985) have shown that disturbed columns do not always perform in a similar manner as undisturbed columns. The significance of this factor may be of considerable importance and should be appreciated while reviewing research from column studies.

Columns are often disassembled into a series of sections at the conclusion of a study. Soil from these sections may be analyzed for a number of properties and characteristics. These characterization data are compared among treatments, or compared to a control, to estimate changes that may occur in the field under similar conditions. Distribution of STE constituents with depth can also be determined from the column soil samples.

Field studies examining the effect of OSWDS to the environment can be limited to the immediate area around a drain line, or expanded to cover the entire recharge area of an aquifer. In general, the larger the overall focus of the study, the more difficult it is to design and interpret the results of the study. Changes in the chemical characteristics or concentrations of soil, soil solution, or groundwater are usually monitored in field studies. These changes are examined to determine if a cause and effect relationship can be attributed to an OSWDS. Soil solutions are most often obtained from lysimeters installed within the soil. These sampling devices may be installed at various depths within a ST-SAF, or installed at various distances from the ST-SAF to monitor rates and direction of lateral flow, or movement of pollutants added in STE. If groundwater is near the surface, sampling wells are often placed at various distances from the ST-SAF to monitor changes in groundwater height as well as the concentration of STE constituents in the groundwater. In both lysimeter and well sampling schemes, sampling stations are placed in the direction of the projected groundwater or subsurface water flow.

Several field studies have used some form of tracer to determine the direction and rate of flow from a ST-SAF or similar source of ground or surface water pollution. Two examples of this type of study, in which Br^- was used as the tracer, can be found in Blume et al. (1987) and Starr and Glotfelty (1990). Tracers have also been used to model NO_3^-, bacteria, and virus movement away from a ST-SAF. Chloride is often used to monitor the flow and transformation of NO_3^- as it moves away from a ST-SAF (Steenhuis and Muck, 1988; Rea and Upchurch, 1980; Gambrell et al., 1975). Antibiotic resistant bacteria (Hagedorn et al., 1978) and bacteriophages (Schaub and Sorber, 1977) have been used to monitor the movement of bacteria and viruses, respectively, from wastewater disposal systems. In addition, bacteriophages have been introduced into the groundwater to trace movement within an aquifer (Martin and Thomas, 1974). Keswick et al. (1982) reviewed the use of microbes for groundwater tracers.

Soil samples from the ST-SAF, and soils adjacent to the ST-SAF are often collected to determine the effect of addition of STE to soils. Data from these samples can be compared with data obtained from samples collected prior to the introduction of the STE to the soils, or samples collected from the same general area as the ST-SAF, but were not affected by additions of STE (control samples).

POLLUTANTS

ORGANIC COMPONENTS

Domestic wastewaters contain various organic components that are usually expressed as biochemical oxygen demand (BOD, determined after a 5 day incubation period), chemical oxygen demand (COD, determined by dichromate method), suspended solids (SS), and total suspended solids (TSS) (Bouwer and Chaney, 1974; Bennett et al., 1975a; US-EPA Staff, 1980). Total organic carbon (TOC, determined by the difference between total and inorganic carbon) is also commonly used to express a portion of the organic components in wastewater (Bouwer and Chaney, 1974; Bouwer et al., 1974; Bennett et al., 1975a). Ranges in concentration of COD (680-730 mg L^{-1}), BOD (200-290 mg L^{-1}), SS (200-290 mg L^{-1}), and TSS (350-530 mg L^{-1}) in domestic wastewater were reported by US-EPA Staff (1980). The source of most of these organics is the garbage disposal and toilet (Bennett et al., 1975). Reductions in the concentration of these organic can occur in either the septic tank or the soil adsorption system.

Levels of organics in septic tanks can be reduced by sedimentation, filtration, or biodegradation. Removals of > 60% of BOD, and > 70% of the SS have been reported for standard septic tanks (Bouma, 1979). Hansel and Machmeier (1980) pooled data from 7 studies and reported reductions of 45 to 55% in BOD concentrations, and about 85% reduction in TSS in septic tanks. The degree to which the organic components are reduced in a septic tank can vary depending upon the type of septic tank, retention time, and the environment within the tank. Stensel et al. (1973) applied a kinetic approach to determine necessary retention times in septic tanks for minimal COD concentrations in STE. Batch studies were conducted at 10, 20 and 30°C. For temperatures of 20 and 30°C a retention time of 4 days is necessary for maximum COD retention in the septic tank. Between 6 and 8 days are necessary for an equivalent reduction in COD levels for temperatures of 10°C.

Brandes (1977) examined the effect of adding alum to a septic tank. Addition of alum decreased the BOD content an additional 42%, but the reduction in TOC was only an additional 7%.

Otis et al. (1975b) compared the effectiveness of standard septic tanks, 3 compartment septic tanks, and aeration units on removal of BOD components and total suspended solids (TSS). The BOD of the effluent from the aeration unit was significantly lower than that of the septic tanks, but the TSS were about the same. Variation in both TSS and BOD levels were considerably higher in aeration unit than septic tanks. Three compartment septic tanks showed no clear advantage over conventional septic tanks.

Organic material applied as STE to a soil is generally filtered or biodegraded by the microflora. Factors which affect the fate of organic material in a ST-SAF are the type of organic material, population and type of microbes, surface area available, aeration status, loading rate of STE, and temperature (Kristiansen, 1982b). Organic particles larger than 1 u will generally be filtered by coarse soils (Zaghloul et al., 1988). Organic material that is < 1 u may therefore move through coarse textured soil if it is not degraded by soil microbes. Kristiansen (1981c, 1982b) investigated the microflora of a sand filter trench. Addition of STE to a sand-filter trench increased the number of bacteria from 10^6 to 10^8 g^{-1} of sand. The largest biomass occurred at the interface between the aerobic and anaerobic zones (right below the clogging mat). This distribution occurs because output from aerobic metabolism is much higher than anaerobic. In addition, most bacteria in the anaerobic clogging layer are remnants of the STE, while in the aerobic zone bacteria are the type which actively degrade organic matter. Kristiansen (1982b) concluded that in most aerobic systems the potential for pollution related to organic material from an OSWDS is minimal.

Several studies have examined the distribution and removal of organic within soil columns or ST-SAF. Viraraghavan and Warnock (1975, 1976) examined reductions in organic components applied to 3 ST-SAF placed in sandy clay and clay soils. Reductions in TSS, BOD, COD, and soluble organic carbon ranged from 75 to 90%. Reductions in these organic components were less after rains or during the winter when water tables were elevated.

Tare and Bokil (1982) examined the reduction in COD levels after application of STE to columns. Columns were filled with various mixtures of sand, silt, and clay. Most COD components were removed in upper 15 cm of columns. In columns continuously flooded, reductions in COD levels increased for first 16 days then showed reduced attenuation of COD. Finer soils removed more COD components, but these differences were minimal during periods of continuous flooding.

Thomas and Bendixen (1969) examined the degradation of BOD and COD components of STE in columns filled with silt loam or sand. Similar degrees of degradation of organics were observed regardless of temperature or duration of dosing. From 72 to 83% of organic carbon in STE was degraded.

Much of the research on attenuation of organic materials applied to OSWDS is focused on the fate of these organics in alternative systems. Several studies have investigated the fate of organic components in simulated mounds. Magdoff and Keeney (1975) examined the fate of COD components applied to columns representing mound systems. Greater than 98% of COD introduced into columns was attenuated.

Magdoff et al. (1974b) examined the efficiency of columns constructed to represent mound system for removal of COD components. In columns in which a clogging mat had not developed, most of COD components were removed. After the clogging mat had formed, and anaerobic conditions were maintained above the clogging mat, about 60% of COD was removed. Magdoff et al. (1974b) concluded that mounds provided adequate treatment for COD components.

Willman et al. (1981) examined various mixtures of sand and clay for treatment of STE. Mixtures ranged from 0-12% clay and were packed in columns to simulate a mound system. Greater than 70% reductions in COD levels were observed during the study regardless of the mixture. Columns with only sandstone sand (0% clay) showed the lowest pH and lowest reduction in COD levels. Columns filled with sandstone sand and 12% clay had the highest pH and showed the highest reduction in COD levels.

Various types of filters are used as primary or final treatment of domestic wastewater. The most common type of filter is the sand filter. Pell and Nyberg (1989a) examined the fate of BOD and COD components in a newly started sand-filter system. An equilibrium was reach in respect to COD and BOD levels in sand-filter effluent within 7 days. After equilibrium less than 92% of BOD and COD were removed. In columns constructed to simulate sand filters, COD levels varied. Variations were related to periods of saturated flow which increased COD in column effluents. Pell and Nyberg (1989a) concluded that less COD is removed under anaerobic than aerobic conditions.

Effert et al. (1985) examined the efficiency of sand filters, upflow gravel filters and subsurface and gravel beds in reducing levels of BOD and COD in STE and aerated STE. Concentrations of BOD were low for sand filters and relatively high for gravels. COD contents varied with STE, but were consistently low in aerated STE.

Kennedy (1982) examined the efficiency of an anaerobic filter to treat STE. Anaerobic filters removed > 80% of the BOD components about 50% of the time. A similar degree of BOD removal was only observed in septic tanks 10% of the time. Suspended solids in the STE were 20-250 mg L^{-1}, but in anaerobic filters SS ranged between 9 to 70 mg L^{-1}.

Rock et al. (1984) examined the fate of COD and BOD components in peat filters. The peat filter reduced BOD levels by > 95% even at high loading rates. In most cases COD levels were reduced by > 80%, but for the first 90 to 100 days decomposition of the peat increased COD loadings.

Rock et al. (1982) examined the efficiency of peat beds in removing COD and TSS levels. Levels of COD were reduced by 83% and 90% of the TSS were removed.

Bouwer et al. (1974) examined the distribution of COD, BOD, and TOC after application of STE to the Flushing Meadows RI System. Nearly 100% of the BOD was removed within the upper 9 m of the RI system. Concentrations of COD and TOC were reduced 66 and 75%, respectively, in the upper 9 m. Most of the reduction of the organic components occurred within the upper meter.

Tilton and Kadler (1979) examined COD levels in STE and a freshwater wetland used for treatment. Levels of COD in wetland were higher than STE, and therefore COD would not be a problem in such systems.

NITROGEN

Nitrogen is generally considered the most important chemical contaminant from OSWDS. The processes governing the transformation and movement of N within and away from an OSWDS depend on various soil and environmental factors. Reviews regarding the fate and transport of N applied as wastewater can be found in Bouwer and Chaney (1974) and Lance (1972). Reviews specifically related to N applied in OSWDS are found in Cogger (1988), Reneau et al. (1989), Sikora and Corey (1976), Bicki et al. (1984), Laak (1982) and Miller and Wolf (1975).

Most properly sited and operating conventional ST-SAF convert NH_4^+ and organic-N to NO_3^-. Nitrate is very mobile in the soil system and can easily move into the ground and surface waters initiating both health and ecological problems. Health problems occur when NO_3^- laden water is consumed by infants. When this occurs, NO_3^- is reduced in the body to $NO2^-$, which can oxidize Fe in the hemoglobin. If this occurs, hemoglobin is no longer able to carry O_2 (Baum, 1982). This condition, termed methomoglobanemia, is particularly toxic to human infants and ruminants (Koren, 1980). A standard of < 10 mg L^{-1} of NO_3^- has been imposed on all drinking water supplies to control methomoglobanemia (Alexander, 1977). Nitrate in surface waters can stimulate algal blooms leading to eutrophication.

In the ST a portion of the N is removed through sedimentation and microbial assimilation. Laak (1982) estimated that about 10% was removed, and Pell and Nyberg (1989) found an average loss of 17% of the N due to sedimentation and assimilation. Most N in the STE occurs as NH_4^+ (75-85%) and organic-N (15-25%) (Walker et al., 1973; Lance 1972, 1975; Otis et al., 1975; Kristiansen, 1981). Mean concentrations of total N in STE range from 29-61 mg L^{-1} (Table 1). Many mechanisms and processes effect the form of N and relative mobility in the soil system including: 1) immobilization; 2) mineralization; 3) volatilization; 4) nitrification; 5) denitrification; 6) chemical decomposition of NO_2^-; 7) cation exchange; and 8) plant uptake. The degree to which each process affects N in the ST-SAF is dependent upon the soil pH, moisture content, redox potential, O_2 content, temperature, CEC, organic carbon form and content, and microbial population and diversity.

Mineralization and immobilization often occur simultaneously because the same microbes are responsible for both processes. Immobilization occurs as microbes utilized organic matter as a C and energy source. During this process available N is retained in the microbial cell for various synthesis reactions (immobilization). The amount of N immobilized is fixed within a system because as microbes die N is released. Lance (1972) reported that the amount N immobilized after addition of wastewater is probably less than 5 to 10%.

If the C:N ratio in the soil is <22:1, excess N is available to the microbe and is released in an inorganic form, generally NH_4^+ (Black, 1968; Lynch, 1979; Keeney, 1981). Release of N from the organic form is termed mineralization. Ratios of C:N in STE are generally <10:1 so in the ST-SAF mineralization occurs (US-EPA STAFF, 1980). Rates of mineralization are affected by several soil factors. A neutral pH encourages mineralization, while acid conditions and low temperatures depress the rates (Alexander, 1977). Anaerobic and aerobic organisms mineralize organic N so that mineralization is significant in both aerated and submerged soils. Moisture levels below wilting point (-1.5 MPa) however, restrict mineralization (Alexander, 1977).

Nitrogen in the form of NH_4^+ can be subjected to several transformations and removal mechanisms including leaching, cation exchange, plant uptake, fixation, volatilization, and nitrification. Plants uptake NH_4^+ readily. Loss of N due to plant uptake however, is generally minimal. Most conventional OSWDS dispose STE subsurfacely, and therefore below the zone of plant uptake.

If factors are not conducive for nitrification or volatilization, and exchange sites become saturated with respect to NH_4^+, leaching of NH_4^+ can occur. Brown et al. (1984) examined N movement after applying STE to undisturbed monoliths of three soils over 2 years. Soil monoliths were 1.8 m long and had a surface area of 3.1 m^2. Textures for the three soils were sandy loam; sandy clay, sandy clay loam, clay, or clay loam; and clay, with loading rates of 82, 33, and 16 L m^{-2} per day respectively. Effluent from the sandy loam soil showed only background levels of NH_4^+ for the first 18 months of monitoring. Thereafter a dramatic rise was observed which continued throughout the study. The rise in NH_4^+ was thought to occur after the exchange sites within the soil had been filled to capacity. Similar trends were not observed in the other two soils. Higher rates of movement of NH_4^+ in the sandy loam soil were related to the higher loading rates, poor aeration due to loading rates reducing nitrification, and lower CEC.

Cho (1971), using 1st order reaction rates for N transformations, described the movement of N in a soil. Results of these studies suggested that as flow rates increased the maximum NH_4^+ concentration moved deeper in the soil column.

The loss of N due to volatilization is generally of minimal importance in OSWDS. Volatilization of NH_4^+ follows the following reaction equation:

$$NH_4^+ + OH^- \rightarrow NH_3 + H_2O.$$

This reaction is only significant at elevated pH values because of the high equilibrium pH (9.5). At a pH of 5, 6, or 9, ratios of NH3:NH_4^+ are 0.0036, 0.36, and 36, respectively (Nelson, 1982). The pH of wastewater is generally between 7.5 and 8, therefore <10% of the N will occur in the NH3 form (Laak, 1982). Once in solution, the rate of volatilization to the gaseous form is governed by the amount of air filled pore space and the distance to the soil surface (Nelson, 1982). If the air space is minimal, the amount of NH_3 in the gaseous phase in equilibrium with the concentration in the aqueous phase is minor. For the exchange of NH3 from solution to the gaseous form to continue, the NH_3 gas must diffuse to the soil surface and be released (Freney et al., 1981). In subsurface ST-SAF, loss of N by volatilization would be minimal because of the considerable distance to the soil surface (>45 cm), and the minimal air space in the moist to saturated ST-SAF soils. Movement of NH_3 to the surface is also dependent on pH. If the pH drops, or a lower pH is encountered during diffusion, NH_3 may revert to the NH_4^+ form in accordance with the volatilization reaction.

Nitrification

In most aerobic soils and OSWDS, NH_4^+ is readily oxidized. The biologically controlled oxidation of NH_4^+ to NO_2^- and/or NO_3^- is termed nitrification. The dominant microbes involved in nitrification are the obligate chemolithotrophic bacteria, *Nitrosomonas* and *Nitrobacter*. These bacteria utilize inorganic N for their energy needs. Organic material can be used for cellular synthesis reactions, but CO_2 is often used as the sole C source (Hamilton, 1979). Heterotrophic nitrification by some bacteria, actinomycetes and fungi may also occur, but it is not considered to be a significant contributor to the nitrification process (Campbell and Lees, 1967).

The following reaction equations describe nitrification (Gilmour et al., 1977):

$$NH_4^+ + 1.5\ O_2 \rightarrow NO_2^- + 2H^+ + H_2O$$

$$NO_2^- + 1.5\ O_2 \rightarrow NO_3^-$$

Nitrosomonas is associated with the oxidation of NH_4^+ to NO_2^-, and *Nitrobacter* completes the oxidation of NO_2^- to NO_3^-. Rates of nitrification are dependent upon available NH_4^+ or NO_2^-, pH, temperature, and contents of O_2 and soil moisture. Mishra et al. (1974a) reported nitrification followed 1st order kinetics. If concentrations of NH_4^+ were high however, zero order kinetics were observed. In natural soil systems, the breakdown of organic matter and subsequent release of NH_4^+ is generally considered the rate controlling step for nitrification (Black, 1968). In a ST-SAF, N enters the soil system primarily as NH_4^+ (75-85%), so that the rates of mineralization have little bearing on rates of nitrification. The optimum pH for nitrification in soils is from 6.6 to 8.0 (Alexander, 1977). At pH values above 8.5, nitrification may be inhibited due to NH_3 toxicity to *Nitrobacter* (Campbell and Lees, 1967; Schmidt, 1982). Rates of nitrification rapidly decline below a pH of 6 (Dancer, et al., 1973; Alexander, 1977; Schmidt, 1982). The oxidation of NH_4^+ to NO_2^- results in an increase in acidity within the system. Ardakani et al. (1974) reported a decrease in pH from 7.4 to 5.0 as the result of nitrification. Similar decreases in pH have been observed by

Stewart et al. (1974), Stewart and Reneau (1988), and Whelan (1988). Decreases in pH may have only minimal effects on nitrification in some systems because several strains of nitrifiers are adapted to acid soil conditions (pH values as low as 4.0). In addition, if a C source is available, heterotrophic nitrifiers that thrive in acid conditions, such as fungi, may contribute to nitrification.

The optimal temperature for nitrification is between 30° C and 35° C. Rates of nitrification will decrease above and below this range in temperatures (Black, 1968).

The most important factor controlling nitrification rates is the availability of O_2 to nitrifiers. Nitrifiers are obligate aerobes that use oxygen as a terminal electron acceptor. Therefore these microbes are most efficient in aerobic, well-drained soils (Martin and Focht, 1977). Mishra et al. (1974a) observed that nitrification was totally inhibited if the O_2 content fell below 0.5%. Similar studies have also demonstrated the dependence of nitrification on aerobic conditions (Pilot and Patrick, 1972; Andreoli et al., 1979; Gilmour, 1984).

Field and column nitrification studies

Both field and column studies have been used extensively to examine the movement and transformation of N. Ardakani et al. (1973) examined nitrification in columns filled with a mixture of sand and soil. Nitrite was added and the concentration and distribution of NO_3^-, NO_2^-, and nitrifiers was monitored. As the number of nitrifiers increased, the amount of nitrification increased. Most NO_2^- was converted to NO_3^- within the upper 10 cm of the column which was the location of the maximum number of nitrifying microbes. Decreases in population density of nitrifying organisms with depth were related to the depletion of O_2 and NO_2^- deeper in the column.

Brown et al. (1984) examined N movement after applying STE to undisturbed monoliths of three soils over 2 years. Textures for the three soils were sandy loam; sandy clay, sandy clay loam, clay, or clay loam; and clay. Initially effluent from the sandy loam soil showed high concentrations of NO_3^- (> 10 mg L^{-1}), however after 5 months concentrations decreased rapidly, and remained below 5 mgL^{-1}. Similar trends were observed in the other soils although the NO_3^- concentrations were less. Lower levels observed after 5 months were related to the saturated conditions within the monoliths which inhibited nitrification. Only 2.2% of the N applied to the sandy loam soil was present in the column effluent. Less N was found in the effluent of columns filled with finer textured soils. Nitrogen concentrations in column effluents were expected to increase as the exchange capacity of the soils reached saturation with respect to NH_4^+. Higher concentrations of NO_3^- were observed with distance from the point of application in the sandy loam soil. This increase in NO_3^- suggested nitrification was occurring away from the saturated conditions around the area of application.

Ardakani et al. (1974) examined the movement and transformation of N in a 40 m^2 plot after application of NO_2^- or NH_4^+. Soil solution samples were collected in ceramic cup lysimeters. Following the application of NO_2^-, concentrations of NO_2^- decreased until reaching a steady state at 26 days. Constant levels of NO_2^- implied the population of *Nitrobacter* in the system had reached a steady state. Nitrite was restricted to the upper 6 cm of the soil. About 35 days were required before consistent levels of NO_3^- were recorded following addition of NH_4^+. These results implied that about 35 days were required for *Nitrisomonas* to reach an equilibrium in the system. By the end of the experiment *Nitrosomonas* were present at all depths, but *Nitrobacter* was primarily at the surface.

Walker et al. (1973a) examined transformations and distributions of N within a ST-SAF. Most of the soil N occurred in an organic form within the clogging mat at the soil-gravel interface. Older ST-SAF had higher concentrations of organic-N in the clogging mat, than younger ST-SAF. The NH_4^+-N was highest just below the clogging mat and rapidly decreased with depth. A concurrent increase with depth of NO_3^- indicated that NH_4^+ was rapidly being transformed to NO_3^-. Most nitrification occurred within the first 6 cm below the clogging mat and occurred within a couple of hours.

Whelan and Barrow (1984) examined N concentrations in soil and soil solutions below 7 soakwells and leach drains in Australia. Soils were composed of greater than 90% sand. In systems where the water table was deep, a large increase in NO_3^- was reported just below the clogging mat. In some cases a subsequent decrease in NH_4^+ was observed, however in other systems decreases did not occur until deeper in the profile. In systems in which the water table was within a m of the bottom of the soakwell or leach drain, distribution of N depended upon the age of the system. In systems younger than 4 years, a clogging mat had not formed at the far end of the drain and nitrification was occurring prior to effluent discharge. In the older systems, with completely saturated leach drains, nitrification did not occur. In these coarse sands, Whelan and Barrow (1984) concluded that all of the N added as STE was moving into the groundwater.

Whelan (1988) examined N concentrations in soil and soil solutions below a soakwell in calcareous sands. Most NH_4^+ in the STE was converted to NO_3^- within 50 cm of the soakwell. Levels of NO_3^- at the deepest sampling point 8 m below the soakwell were >50 mg L^{-1}. Loading rates per m^2 for soak wells are very high due to their design. Movement of NO_3^- from the soakwell was almost entirely vertical until a cemented calcareous layer was reached. The cemented layer initiated lateral NO_3^- movement.

Cogger and Carlile (1984) examined N movement away from 15 ST-SAF in the coastal plain of North Carolina. Wells were placed at distances of 1.5 and 7.5 m from the ST-SAF. Soils with higher water tables showed higher levels of NH_4^+ and lower levels of NO_3^-. Levels of NO_3^- increased in the fall as rising water tables and increasing precipitation displaced NO_3^- remaining in the soil from the dryer summer months. Lower concentrations of N were observed in the winter months because of higher amounts of dilution. Concentrations with distance from the ST-SAF were greatest in the summer months. These results most likely occurred because the water table gradients were highest at this time of year due to the water table mound below the ST-SAF. Average redox potentials showed NO_3^- was the dominant N species at potentials of 477 mV, while NH_4^+ was dominant at 260 mV. These differences were significant at the .01 level.

Cogger et al. (1988) examined N movement in a Quartzipsamment in North Carolina. Two sites were examined in which the height of the water table differed by 30 cm. Wells were installed .3 to 3 m from each ST-SAF trench. Higher concentrations of NH_4^+ were observed in the water samples taken from wells adjacent to the ST-SAF with the high water (Eh = 280 mV), with higher concentrations of NO_3^- in wells adjacent to ST-SAF with the lower water tables (Eh = 510 mV). In the soil with high water tables NO_3^- was only observed after heavy rainfall and a previous dry period. Concentrations of NO_3^- 3 m from the ST-SAF were above 10 mg L^{-1}.

Lateral movement of NO_3^- has been documented in numerous field studies. Hubbard and Sheridan (1982) examined N concentrations in surface run-off and subsurface flow over a ten year period in a small (0.34 ha) watershed in Georgia. Soils within the watershed had loamy sand surfaces and plinthic subsurface horizons. Slopes averaged 2%. Subsurface flow accounted for 99% of the N losses after fertilization, with 74% of the losses occurring between December and Máy. About 16% of the N added was lost from the watershed. Concentrations of NO_3^- in the subsurface flow averaged 8.75 mg/L.

Steenhuis and Muck (1988) examined the movement of NO_3^- after application of KNO_3^- to an experimental plot with an 8% slope. Water samples were collected 36 and 69 m from the point of application. Under saturated conditions, up to 60% of the fertilizer added was accounted for in the water samples. Flow rates in saturated conditions were governed by macropore flow.

Rea and Upchurch (1980) examined N movement as a result of application of STE to Blanton and Leon fine sands with < 3% silt and clay. Water tables were within a meter of the soil surface. Wells were established at five points along a 30 m toposequence. Ratios of NO_3^-:Cl- showed similar trends in both the lateral and horizontal direction suggesting dilution was the only mechanism for reduction in NO_3^-

levels. Concentrations of NO_3^- were as high as 50, 30, and 10 mg L^{-1} in water samples taken from wells at distances of 6, 15, and 25 m away from the ST-SAF respectively.

Peavey and Groves (1978) examined N movement away from a ST-SAF in Montana. The ST-SAF was build in alluvial sediments composed of silt loam, over a gravelly loam, over sands, gravels, and cobbles. Water tables were within 1.2 meters of the soil surface. Average concentrations of two wells located 9 m from the ST-SAF were 10 and 75 mg L^{-1} suggesting nearly complete nitrification of the STE and dilution as the only treatment for reduction of N.

Reneau (1977) examined changes in N forms and levels after application of STE to a 15 year old ST-SAF in a Virginia. Plinthic subsurface horizons occurred within the soils of the study area. The plinthic horizon inhibited vertical flow resulting in a fluctuating high water table. Water samples were taken from wells spaced from 15 cm to 12 m from the ST-SAF over a 4 year period. Concentrations of NH_4^+ decreased with increasing distance from the ST-SAF. Higher levels of NH_4^+ near the ST-SAF were due to the high concentration of NH_4^+ in the STE, the anaerobic conditions, and the high O_2 demand. Decreases in NH_4^+ with distance were related to nitrification and cation exchange or fixation. Levels of NO_3^- taken from the plinthic horizon increased for the first 3 m from the ST-SAF. Concentration of NO_3^- in soil solutions within the plinthic horizon showed minimal change with distance from the ST-SAF. Redox potential within the plinthic horizon increased with distance from the ST-SAF, but was always above 340 mV.

Reneau (1979) examined lateral movement of N from 3 ST-SAF on the coastal plain of Virginia. Soils were either fine -loamy or coarse loamy Ochraquults. Wells were placed for sampling along the groundwater flow pattern to an agricultural drainage tile located 12 to 19 m from the three ST-SAF. High water tables were observed at various times during the study period. The amounts of NO_3^- in water samples were shown to increase for the first 5 meters from the ST-SAF. The increases were the result of nitrification as aerobic conditions increased with distance from the

ST-SAF. Highest levels of NO_3^- were observed during rising water tables as NO_3^- formed during dryer periods moves into the rising groundwater. Although NO_3^- was observed entering the drainage tile from the ST-SAF, greater NO_3^- pollution of the groundwater was occurring from cultural practices such as lawn and garden care as evidenced by the elevated levels of NO_3^- in the control wells.

Stewart and Reneau (1988) examined N movement in soils treated with STE. These soils had high water tables and STE was applied at < 30 cm below the soil surface using LPD. Soils were poorly drained fine loamy, siliceous, Typic Ochraquults. Wells were placed at distances of 2.8 and 8.4 m from the ST-SAF to monitor N concentrations. Highest NO_3^- concentrations were observed during rising water tables after extend an extended dry period. Movement of NO_3^- was found to occur in the greatest amount and to the maximum distance during high water tables as a result of saturated flow and an increase in the water table gradient.

Walker et al. (1973b) examined N forms and concentrations with distance from 4 ST-SAF in Wisconsin. The ST-SAF were placed in interlayered glacial deposits of sand, silt and clay. Water tables at one site were at or above the drain tile. Samples were collected 6 times over a 15 month period. There appeared to be no seasonal effects. In unsaturated systems, most of the N in the groundwater occurred as NO_3^-. Concentrations of NO_3^- in groundwater immediately below the ST-SAF were as high as 40 mg L^{-1}, and were reduced to < 10 mg L^{-1} within 70 m down gradient of the ST-SAF. In these systems, reduction in NO_3^- levels could only be attributed to dilution. Up-gradient of the ST-SAF NO_3^- was at background levels. In saturated systems, most N occurred as NH_4^+, and decreased rapidly with distance. The decreases were probably related to cation exchange.

Wilson et al. (1982) examined NO_3^- concentrations in a tile drain used to lower the water table for ST-SAF usage. Drains were placed at depths of 1.2 or 1.8 m, at distances of 3 or 6 m from the ST-SAF, respectively. Levels of NO_3^- were always less than 10 mg L^{-1} with mean values ranging from 0.6 to 4.7 mg L^{-1}. Water quality was similar in drains set at 3 and 6 m separation distances. The control well had the highest mean NO_3^- concentration due to fertilizer practices in the adjacent area.

Denitrification

Denitrification is the most important process for removing N applied to a ST-SAF. In this biological process, N oxides are reduced to a gaseous form by facultative anaerobic bacteria. These bacteria use N oxides as terminal electron acceptors in the absence of O_2 (Black, 1968; Alexander, 1977; Firestone, 1982). Denitrification will only occur if denitrifying bacteria are present; electron donors such as organic C, H_2, or reduced sulfur are available; anaerobic conditions exist; and nitrogen oxides such as NO_3^-, NO_2^-, nitrogen oxide (NO), or nitrous oxide (N_2O) are available to serve as electron acceptors.

Rates of denitrification are dependent upon the concentrations of NO_3^-, soluble C, and O_2, as well as the pH, temperature, and moisture content of the soil (Bremner and Shaw, 1958; Myers and McGarity, 1972; Burford and Bremner, 1975; Firestone et al., 1979; Firestone et al., 1980; Koskinen and Keeney, 1982). The order in which denitrification kinetics follow is under some dispute. Several researchers have suggested that denitrification follows zero order kinetics (Bremner and Shaw, 1956; Doner et al. 1974;). Other researchers however, have suggested that 1st order kinetics are more appropriate (Mishra et al., 1974a; Reddy et al., 1980; Sikora and Keeney, 1976). In batch studies, Reddy and Patrick (1978) examined 15 soils with various physical and chemical characteristics. Nitrate and rice straw were added to each soil. One set of soils was saturated the other flooded. Saturated soils followed 0 order kinetics. Flooded soils however, followed 1st order kinetics. Reddy and Patrick (1978) concluded that NO_3^- must first diffuse into the soil before finally being reduced. Similar results were reported by Stanford et al. (1975a).

Bowman and Focht (1974) examined denitrification in batch studies using a fine sand soil. Glucose was added as a C source. Denitrification rates followed Michaelis-Menten kinetics. Rates at low NO_3^- concentrations followed first order kinetics. At higher concentrations of NO_3^- denitrification occurred according to 0 order kinetics. Similar results were observed by Stanford et al. (1975a). Doner et al. (1974) and Sikora and Keeney (1975) used column studies to examine denitrification kinetics. Doner et al. (1974) found denitrification followed 0 order kinetics near the surface of the column, but followed higher order kinetics deeper in the column. Results presented by Sikora and Keeney (1975) suggested first order kinetics near the surface, and higher order kinetics with depth.

Denitrification occurs under anaerobic conditions when O_2 is restricted from microbes (Bremner and Shaw 1956). Denitrification may also occur in well aerated soils if anaerobic microsites are present. Several studies have shown that denitrification will occur in the presence of O_2. Mishra (1974b) reported similar rates of nitrate reduction for samples maintained at 0.5 and 5.0% O_2, but rate coefficients were 10 times less at 20% O_2 content. Engler et al. (1976) examined denitrification in 4 soils and floodwaters. As much as 16 ppm O_2 did not inhibit denitrification because of the rapid depletion of O_2 in the systems.

Numerous studies have shown relationships between moisture content and denitrification rates. Freney et al. (1979) observed a marked increase in rates of N_2O production as water was added to the soil (up to 62% saturation). Similarly, Bremner and Shaw (1956) noted that minimal NO_3^- reduction occurred when the moisture content was < 60% of the water holding capacity of the soil.

Pilot and Patrick (1972) examined NO_3^- loss from three soils of very fine sandy loam, loamy sand, and silty clay loam textures. Experiments were conducted in columns 7 cm long and maintained at various moisture tensions. At the low moisture tensions high rates of NO_3^- reduction were recorded. Critical tensions to induce denitrification ranged from 20 to 40 cm. Greater than 11 to 14% air filled porosity restricted denitrification. The finer the soil the higher the air filled porosity necessary to prevent nitrate reduction.

Mahendrappa and Smith (1967) examined the effect of moisture content on denitrification. Four soils, two acid and 2 basic, were examined in batch studies. Data showed that each soil exhibited a moisture level in which denitrification was optimized. The optimum moisture content was related to an equal distribution of N products and microbes, as well as dilution of toxic products produced.

Redox potential (Eh) is often used as an estimate of the O_2 status of a soil. The upper limit for the reduction of NO_3^- to NO_2^- is about 420 mV at pH 7 and 25° C (Focht, 1978). The Eh for the reduction of N_2O to N_2 is approximately 250 mv at pH 7 and 25° C. Engler et al. (1976) examined denitrification in 4 soils and floodwaters. Redox measurements over time showed a rapid decline as O_2 concentrations approached 0 (15 minutes to 4 hours). After which Eh leveled off, suggesting the system was being controlled by other elemental forms such as NO_3^- and Mn. The Eh at which this occurs depended upon the soil, and ranged from 100 to 375 mV for the 4 soils examined.

Meek et al. (1969) examined relationships between denitrification and Eh. In batch studies NO_3^- became unstable at Eh values below 300 mV leading to denitrification. Addition of organic C (barley straw) increased the amount of denitrification without a further lowering of the Eh. Lowering of the Eh was necessary in samples without additional C for an equivalent amount of denitrification to occur.

Rates of denitrification increase with increasing temperature (Stanford et al., 1975b, Bailey, 1976; Mishra, 1974b; Reddy et al., 1980; Sikora and Keeney, 1976). Sikora and Keeney (1976) examined effects of temperature on denitrification rates in 64 cm columns packed with dolomite limestone chips. Methanol and KNO_3^- were added to aerated STE. Reduction of NO_3^- followed first order kinetics, with the 5° C temperature showing the highest correlation (r=.838), and 20° C the lowest (r=.412). Both were significant at 1% levels.

Stanford et al. (1975b) examined the effect of temperature on rates of denitrification in nine soils. Rates between 35 and 45° C were similar. A steady increase in rates occurred between 15 and 35° C, with each 10° rise a 2 fold increase was observed. Below 10° C a rapid decrease in rates were recorded.

The optimal pH for denitrification varies depending upon the soil. Russell (1973), and Alexander (1977) reported that the highest rates of denitrification occurred in soils with pH values ranging from 8-8.6, but in some soils the reaction may be rapid at a pH of 4.7. Bremner and Shaw (1956) examined relationships between denitrification and soil pH. Soils with pH values of 4.1 and 3.6 showed minimal amounts of denitrification. Soils with pH values of 7.5 and 8.2 however, showed similar rates of denitrification and the highest amount of denitrification. Stanford et al. (1975a) examined the relationship between soil pH and rates of denitrification. Thirty soils were examined with various physical and chemical attributes. Soil pH ranged from 5.3 to 8.1. Rates of denitrification were not affected by soil pH. Fillery (1983) suggested that at high pH conditions, levels of soluble C increase. Therefore, increases in rates of denitrification may be a response to the additional C released at the higher pH.

Denitrification will only occur if denitrifying bacteria, C, and NO_3^- or NO_2^- are present. Which of these components is the limiting factor in rates of denitrification is dependent upon several factors. Volz et al. (1975) examined NO_3^- reduction and changes in associated microbial populations after ponding a soil. Nitrate reducers were always in greater numbers than denitrifying bacteria. Most NO_3^- reduction occurred in the upper 16 cm of soil. Denitrifying organisms decreased with depth and time suggesting that organic matter and not NO_3^- was limiting in microbial activity and growth.

Stensel et al. (1973) used a kinetic approach to determine necessary retention times to produce effluent with minimal NO_3^- following introduction of wastewater. Results indicated that NO_3^- was not limiting to biological growth until low levels were reached, and organic matter is the limiting factor. Removal of NO_3^- was dependent upon concentrations of methanol, dissolved oxygen in effluent, and NO_3^-, as well as retention time.

Patrick and Wyatt (1964) examined the loss of N from soils due to alternating submergence and drying. This study showed that after the first 3 submergence-drying cycles, changes in N forms did not occur. Patrick and Wyatt (1964) concluded that after 3 cycles all reactive organic matter had been used by microbes, and further NO_3^- reduction would not occur.

Ardakani et al. (1975) examined denitrification in soil columns under continuous flow. Columns were filled with 125 cm of sandy loam soil material and $Ca(NO_3^-)_2$ added as an N source. Some columns received additions of glucose. Most bacteria were found near the soil surface where most of the denitrification occurred. Glucose was not limiting in this area of the column and moved lower into the column. Columns without glucose added showed less denitrifying organisms as well as less denitrification.

Bailey (1976) examined the effects of vegetation on NO_3^- reduction. The addition of the corn plants increased the rate and amount of denitrification by providing an additional C source for the microbes.

Reddy and Patrick (1978) examined denitrification in 15 soils with various physical and chemical characteristics. Soils with an additional C source showed complete denitrification within 24 hours, but those without C additions required 10 days before total denitrification occurred.

Engler and Patrick (1974) examined denitrification in batch and column studies. Addition of rice straw increased denitrification rates by increasing microbial activity and thereby decreasing aerobic conditions.

Sources and forms of C vary widely in soil and wastewater. Some C forms are utilized much more efficiently than others by denitrifying bacteria. Stanford et al. (1975a) examined the influence of total and extractable C on rates of denitrification. Thirty soils were examined with various physical and chemical attributes. Denitrification rates were better correlated with soluble C (r2=.82) than for TOC (r2=.69).

Sikora and Keeney (1975) examined several C sources to determine which was the most efficient for denitrification. In both column and batch studies, methanol provided a much better C source than paper mill sludge, peat, or leaf litter. Additions of methanol increased denitrification rates so that in hours complete denitrification occurred.

Burford and Bremner (1975) examined the relationships between different C components and the amount of denitrification for 17 surface soils in batch experiments. Denitrification capacities were significantly correlated with TOC (r=.77), water-soluble C, and readily decomposable C (both forms r=.99). On the average, the amount of TOC, decomposable C, and water-soluble C were 193, 1.36, and .72 times the amount of N needed before N_2 or N_2O was observed. Addition of glucose to samples which exhibited only 4 to 17% denitrification increased the amount to 96-99%. Bremner and Shaw (1956) reported similar results.

Ward (1985) examined rates of denitrification and CO_2 evolution in soils obtained from the surface and subsurface of a ST-SAF. Additions of glucose and glutamic acid degraded 5 to 7, and 2 to 3 times faster for surface than subsurface soils respectively. Sterile soils showed no CO_2 evolution. Denitrification was linear with time for surface soils. Subsurface soils showed an initial lag time of 20 hours before denitrification occurred. Ward (1985) concluded that the lag time was the result of slow enzyme induction, or the microbial population needed time to increase before denitrification occurred.

Volz and Starr (1977) examined NO_3^- reduction and changes associated microbial populations during continuous leaching experiments. Columns were packed with fine sandy loam soil material and maintained in anaerobic conditions. Columns were leached with NO_3^- and glucose solutions for 96 hours. In the first 18 hours NO_3^- disappeared but concurrent increases in NO_2^- were not observed suggesting that denitrification of NO_3^- was occurring without NO_3^-reduction. Between 18 and 60 hours NO_2^- concentrations increased and then decreased. With time denitrifiers became a larger portion of the microbial population. Most C usage was associated with NO_3^- reduction and not denitrification.

Field and column denitrification studies

Relationships between water table depth and denitrification have been examined in numerous field studies. Gambrell et al. (1975a) observed higher amounts of NO_3^- in run-off from moderately well drained coastal plain soils than similar poorly drained soils. They suggested that denitrification was occurring in the poorly drained soils.

Gambrell et al. (1975b) examined the effect of drainage on levels of denitrification in coastal plain soils of North Carolina. Water tables at one site were > 3.5 m below the soil surface. At the other site, water tables varied from 0.3 to 0.6 m below the soil surface. Gambrell et al. (1975b) suggested that denitrification would occur if the Eh values were below 375 mV, and NO_3^- to soluble C ratio is greater than 1.3. In the moderately well drained site, Eh values ranged from 475-675 mV, soluble C was below the required levels (soil organic content was 2%), and minimal denitrification occurred as indicated by similar ratios of NO_3^- to Cl-. In the poorly drained soils, soluble C levels were always greater than required for denitrification (organic matter was 27%). Above the water table however, Eh values were > 375 mV, but low levels of NO_3^- were observed suggesting denitrification occurring in anaerobic microsites. Below the water table Eh values were below 325 mV and little NO_3^- was observed suggesting nearly complete denitrification. Rates of denitrification in the winter were below those observed in the summer, suggesting that saturated conditions do not always result in complete denitrification. Gambrell et al. (1975b) suggested that O_2 trapped in soil during periods of lower water tables, or elevated concentrations of O_2 in rainwater may have contributed to the lower levels of denitrification in the winter. Effects related to lower temperatures were not mentioned.

Stewart and Reneau (1988) examined N movement in soils treated with STE. These soils had high water tables and STE was applied at < 30 cm below the soil surface using LPD. Soils were poorly drained fine loamy, siliceous, Typic Ochraquults.

Wells were placed at distances of 2.8 and 8.4 m from the ST-SAF to monitor NO_3^- concentrations. Ratios of NO_3^-:Cl^- declined with distance from the ST-SAF indicating that denitrification was occurring. Based on NO_3^-:Cl^- ratios, > 90% of the NO_3^- recorded under the ST-SAF could not be accounted for at 8.4 m. These results suggest that denitrification is substantial in these poorly-drained soil systems and minimal N pollution to the groundwater occurs at distances > 9 m.

Reneau (1977) examined changes in N forms and levels after application of STE to a 15 year old ST-SAF in Virginia. Soils in the study area had formed from coastal plain sediments. A plinthic subsurface horizon in these soils was inhibiting vertical flow resulting in a fluctuating high water table. Levels of NO_3^- taken from the plinthic horizon increased for the first 3 m from the ST-SAF, and then decreased with distance. The increases in NO_3^- content were the result of nitrification as NH_4^+ reached a more aerobic environment. The subsequent decreases were related to denitrification as the NO_3^- approached the water table or other anaerobic zones. Redox potentials in the zones of denitrification were below 200 mV.

Reneau (1979) examined lateral movement of N from 3 ST-SAF on the coastal plain of Virginia. Soils were either fine -loamy or coarse loamy Ochraquults. High water tables were observed at various times during the study period. The amount of NO_3^- in water samples was shown to increase for the first 5 meters from the ST-SAF, then decrease. The increases were the result of nitrification as aerobic conditions increased with distance from the ST-SAF. Reduction in NO_3^- after the first 5 m, was attributed to denitrification.

Meek et al. (1969) examined relationships between denitrification and Eh. Levels of NO_3^- declined as the water table was approached. These changes were related to denitrification. Horizons which showed evidence of denitrification had Eh values below 300 mV.

Meek et al. (1970) examined N transformations in columns filled with 3 m of soil ranging in texture from silt loam, silty clay, and clay. Ground Bermuda grass was incorporated into the upper 10 cm to provide a C source. The water table was fixed at 175 cm. Nitrogen was applied as fertilizer. Denitrification was rapid at or near the water table. Drying and subsequent irrigation increased the soluble C content and may have stimulated denitrification. Denitrification was associated with decreases in redox potential and O_2 in the soil solution and atmosphere.

Weil et al. (1990) examined NO_3^- concentrations within fields in which poultry manure was applied and the adjacent forested land. Four wells were established in a transect from the manured fields into forested land. Wells were oriented with the groundwater flow path and installed to a depth below the water table. Field soils were well drained Sassafras soils. High concentrations of NO_3^- (mean concentrations of 43 mg L^{-1}) were found under the manured fields. Only one well showed evidence of denitrification This well had a buried horizon within the profile which probably provided a C source. Wells located in the forested areas showed low NO_3^- concentrations (< 1 mg L^{-1}). Weil et al. (1990) concluded that denitrification was occurring under the forested areas.

Gilbert et al. (1979) examined the effect of infiltration rates and addition of a C source on denitrification. Columns were filled with 250 cm of loamy sand (Flushing Meadows) and flooded with secondary sewage effluent (SSE) for 9 days followed by 5 days of drying. Increasing infiltration rates decreased the amount of NO_3^- lost. Addition of glucose only effected the amount of denitrification at higher infiltration rates. Denitrifying bacterial populations increased with the addition of C. Higher microbial populations maintained anaerobic conditions during drying providing additional time for denitrification. At 2 cm depths Eh was 418 mV if glucose was not added. If glucose was added continuously Eh ranged from 45 to 240 mV.

Lance et al.(1976) examined denitrification in columns filled with loamy sand. Data showed an exponential increase in N removal as infiltration rates decreased (r^2 = .95). Removal was dependent upon infiltration rates and not loading rates. Lance et al. (1976) suggested that at high infiltration rates recirculation may be needed for adequate N removal.

Lance and Whisler (1972) examined the fate of N in columns flooded on several cycles with SSE. Columns flooded for 2 days and dried for 5 did not remove N. Flooding soils for 9 and 16 days followed by 5 days of drying increased NO_3^- removal through denitrification by 28 and 33% respectively. Continuous flooding removed most N in the first 20 days but afterwards no N was removed. The exchange capacity of a soil was important in adsorbing NH_4^+ so that it could be nitrified and finally denitrified.

Although NO_2^- is most often lost through denitrification, NO_2^- can also undergo an abiotic decomposition process. Nitrite decomposition occurs as a result of the reaction of NO_2^- with organic matter to form N_2, nitrogen dioxide (NO_2), or N_2O (Alexander, 1977). This process is only important if a significant amount of NO_2^- accumulates. During nitrification NO_2^- is converted to NO_3^- at a much faster rate than NH_4^+ to NO_2^-, so little NO_2^- accumulates (Chalk and Smith, 1983). At a high pH (above 9.0) however, Nitrobacter is inhibited and NO_2^- will accumulate. Nitrite can also accumulate during denitrification. If NO_3^- concentrations are high, the NO_2^- produced may be fairly significant because microbes prefer to use NO_3^- as the electron acceptor.

Decomposition of NO_2^- is probably of minimal importance in conventional ST-SAF, because complete nitrification occurs within a short distance of the system. If denitrification occurs within the ST-SAF however, NO_2^- decomposition may effectively release N from the system as a gas.

Nitrogen in groundwater

Nitrate contamination of ground waters from OSWDS has been reported by Quan et al. (1974), Miller (1975), Valiela and Costa (1988), Mancl and Beer (1982), and Bauman and Schafer (1985). Quan et al. (1974) reported that from 30280 to 37850 m3 per day of effluent is applied via OSWDS to a 78 km^2 area in East Portland, Oregon. Levels of NO_3^- in shallow ground water within the East Portland area ranged from 5 to 12 mg L^{-1}. Much lower levels of NO_3^- (<1 mg L^{-1}) were recorded in the deeper aquifers and in the shallow ground waters up gradient of East Portland.

Miller (1975) examined NO_3^- levels in wells within Delaware. Wells were sampled from two separate areas of the state. One area consisted of poorly drained Elkton, Woodstown, or Fallingston soils. Well-drained Sassafras soils were the predominant soils within the other area. Levels of NO_3^- ranged from 0.1 to 11.3 mg L^{-1} for the 37 wells located in the poorly drained area. Average concentrations of NO_3^- for 21 of the wells were 2.4, 1.7, and 1.5 mg L^{-1} for August, December, and May respectively. These data suggest that at times of higher water tables the levels of NO_3^- are less. Levels of NO_3^- in 20 wells located in the well-drained area ranged from 5 to 30 mg L^{-1}. Average NO_3^- levels ranged from 12.2 to 14.4 mg L^{-1}, with a slight decrease from September to May. Miller (1975) concluded that groundwater pollution, in the form of NO_3^- from OSWDS, was occurring in Delaware when depths to water tables were greater than 1.5 m from the soil surface.

Mancl and Beer (1982) examined N concentrations in 68 wells within a 58 acre area of Iowa. Homes within the study area used ST-SAF for wastewater treatment. Soils were sandy loam or loamy sand terrace deposits. In 53% of the wells, concentrations of NO_3^- were > 10 mg L^{-1} with a maximum concentration recorded of 25.6 mg L^{-1}. Levels of NO_3^- in the nearby lake, with a groundwater linkage to the well water, showed low levels of NO_3^- (0.1 to 1.39 mg L^{-1}).

Valiela and Costa (1988) examined N concentrations in both the groundwater adjacent to, and salt and brackish water in, Buttermilk Bay, Massachusetts. Concentrations of N in the bay were highest in the near shore brackish water where inputs of freshwater entered as groundwater. Valiela and Costa estimated that 43% of the N entering the bay in groundwater was attributable to ST-SAF. An estimated 68% of the N added to the watershed as STE, fertilizer, and miscellaneous other sources was attenuated before reaching the groundwater. Most of the N attenuated was surface applied. The N introduced subsurfacely was likely to reach the groundwater.

A positive correlation between density of OSWDS and declining groundwater quality was presented in numerous papers (Bicki et al., 1984; Morrill and Toler, 1973;

Ford et al., 1980; Geraghty and Miller, 1978; Bauman and Schafer, 1985). Bicki et al. (1984) reviewed much of the literature pertaining to the effect of OSWDS on ground and surface water pollution. Morril and Toler (1973) examined the concentrations of soluble salts and dissolved solids in streams draining 17 small watersheds in Massachusetts. Density of OSWDS ranged from 0 to 350 OSWDS km^{-2}. Concentrations of dissolved solids increased 10 to 15 mg L^{-1} for each additional 39 OSWDS km^{-2}.

Ford et al. (1980) examined N concentrations in 164 wells in Colorado. Higher concentrations of N were found in wells in which housing density was the greatest. Concentration of $NO_3^- > 10$ mg L^{-1} were associated with housing densities of > 2.5 OSWDS km^{-2}. Ford et al. (1980) concluded that the separation distance between ST-SAF and groundwater wells should be at least 61 m to insure that densities of OSWDS are maintained at a safe level.

Bauman and Schafer (1985) introduced a conceptual model to estimate the amount of N introduced into the groundwater from OSWDS. In order to evaluate the potential for N pollution to an aquifer, Bauman and Schafer (1985) suggested that four parameters should be estimated: 1) diluting capacity of the aquifer; 2) N loading of the aquifer; 3) potential for denitrification; and 4) importance of aquifer for drinking water. Aquifers leading to wetlands or lakes should be treated differently than those used primarily for drinking water. Several factors should considered in determining the potential for N ground water pollution. These include the depth to the water table, conductivity of the aquifer, flow and gradient of the groundwater, aquifer size, NO_3^- background, geology, and potential of denitrification.

Fate of N in alternative systems

In most conventional OSWDS NH_4^+ is rapidly converted to NO_3^-. Nitrate is very mobile in the soil system and easily moves into the groundwater. Alternative systems have shown mixed success in providing adequate treatment of NO_3^-. Cogger and Carlile (1984) examined N concentrations from 15 conventional and alternative OSWDS in the coastal plain of North Carolina. Alternative systems were mounds, LPD systems, or replacement of fine textured soils with coarser material (soil replacement). Results suggested that the LPD systems worked better than conventional systems during saturated conditions because flow rates were reduced allowing for more time for N transformations. Soil replacement of finer textured soils with coarser material increased the flow rates resulting in less ponding, but did not reduce N concentrations. Mounds malfunctioned due to design and were not evaluated.

Stewart and Reneau (1984) examined the effectiveness of a shallowly placed LPD system in removing N from a Typic Orchraquult. Results suggested that LPD allowed for initial unsaturated conditions because of an equal distribution of effluent. Unsaturated conditions provided an aerobic environment for nitrification to occur. Subsequent denitrification occurred at the water table interface. Carbon may have been provided by soil organic matter or from the grass growing over the shallow placed OSWDS.

Willman et al. (1981) examined various sand and clay mixtures for renovation of STE. These mixtures were added to soil columns to represent a mound system. Regardless of the mixture or sand type, minimal treatment for N was observed in the columns.

Magdoff and Keeney (1975) examined the fate of N applied as STE to columns representing mound systems. Removal of N was 32% during the first 104 days, but was reduced to only 7% in the last 88 days of the experiment. Most removal of N occurred in the silt loam soil below the sand mound. Magdoff and Keeney (1975) concluded that denitrification was occurring in the silt loam soil. During the first 104 days of the experiment, most of the soluble C in the silt loam soil was used by microbes. Therefore, denitrification was limited by insufficient C for the remainder of the study.

Stewart et al. (1979) examined the efficiency of mounds in removing N from STE. Columns were constructed to simulate a mound system. Below each simulated mound was either gravel or an organic matter-sand mixture. Organic matter was obtained from a histic epipedon. Both the gravel and organic matter mixtures were maintained under saturated conditions. Most (> 87%) N added in the STE was nitrified in the simulated aerobic mounds. Denitrification occurred within the organic matter-sand mixture during the first 42 days of the study resulting in a 93% reduction in NO_3^- concentrations. Losses of NO_3^- fell below 28% during the remainder of the study. This decrease in denitrification was attributed to the minimal amounts of C remaining in the organic matter-sand mixture after the first 42 days of operation. Denitrification did not occur in the gravel beds.

Most alternative systems rely on denitrification to remove N. Eastburn and Ritter (1985) reviewed denitrification in conventional and alternative OSWDS. Eastburn and Ritter (1985) concluded that most OSWDS do not provide enough reactive organic C for complete denitrification to occur. Alternative C sources such as methanol (Lamb et al., 1988) or grey water (Laak, 1981a) have been investigated for denitrification. Laak (1982a, 1982b, 1991) described an alternative system for denitrification called the RUCK system. In the system, black water is separated from grey water. Black water is treated in an aerobic sand filter to induce nitrification. Grey water is treated in a septic tank. The STE and sand filter effluents are mixed in an anaerobic filter tank filled with rocks. Grey water serves as the C source. Laak (1982a) showed that grey water C will degrade as fast as methanol. A reduction of 70% NO_3^- was observed when using grey water as the C source, An 83% reduction in NO_3^- was observed using methanol. Data suggested that $TOC:NO_3^-$ ratios of > 0.7 were necessary for efficient denitrification. Retention times should be > 3 days. Laak (1991) reported continued denitrification in RUCK systems, with the average reduction in N between 60 and 70%. Systems which used liquid soap as a C source, as much as 85% reduction in NO_3^- was reported.

Lamb et al. (1988) examined the efficiency of a recirculating sand filter and a RUCK system for denitrification. Sand filters for the initial pretreatment of STE provided, on average, 66 and 70% nitrification. Temperature and degree of nitrification was highly correlated ($P > 0.01$). At temperatures below 10° C nitrification was decreased to 24 to 44%. The anaerobic rock tank with grey water as the C source (RUCK system showed a 50% reduction in NO_3^- as the result of denitrification. The use of STE as a C source showed only a 25% reduction. Addition of methanol to the rock tank increased the denitrification efficiency to 97%, suggesting that methanol was a better C source than grey water for denitrification.

Sand, peat, or sand-peat filters are commonly used as an alternative system, or as a pretreatment for OSWDS. Sandy et al. (1988) examined the use of a recirculating sand filter for removal of N from STE. In this system effluent from the filter is fed back into the ST where a C source and anaerobic conditions exist. Removal of N ranged from 80 to 90%.

Kristiansen (1981) examined the fate of N in sand filter trenches. One set of sand filters were heated throughout the year to maintain a 12 to 15° C temperature. The other set were maintained at ambient temperatures. On average 50% of the N was oxidized in the nonheated units while 65% of the N was oxidized in the heated units. In the winter months the heated units showed about a 50% increase in oxidation of N over the nonheated units. Concentrations of NO_3^- in effluent were related to temperature, flow rates, and redox conditions induced by soil water content. Greater rates of nitrification occurred at higher temperatures because of higher biological activity. Higher temperatures also increased the clogging mat and thereby reduced flow rates and moisture contents below the clogging mat. Lower moisture contents are associated with better aeration, and greater nitrification. Kristiansen (1981) concluded that clogging mats may be important to nitrification in OSWDS. Total N contents were not reduced in the sand filters. Kristiansen (1981) suggested that to reduce NO_3^- contents in sand filter effluent, the system should be designed with LPD, intermittent loadings, and filter material with a larger CEC than sand. In this system, NH_4^+ could be held on exchange sites, nitrified during resting periods, and then denitrified during the next loading. Recycling the effluent back into the ST for denitrification was also suggested.

Rock et al. (1982) examined the efficiency of a peat bed for removal of N from STE. The filter bed was under aerobic conditions with nearly complete nitrification occurring. The mineralization of the peat and leaching of NO_3^- increased the NO_3^- content of effluent leaving the filter bed to 275% of that applied.

Rock et al (1984) constructed columns to represent a peat filter system. Less than 10% of N applied in STE was removed by the peat filter. Inducing anaerobic conditions within the columns promoted denitrification. Losses of N in anaerobic columns were as much as 62%.

Brooks et al. (1984) applied STE to several peat filters and examined the concentrations of NO_3^- in the effluent. An 83% reduction in NO_3^- was observed. This degree of reduction was much higher than that reported for the associated column study (Rock et al., 1984). The differences between the degree of N removal in the two studies were thought to be related to dilution within the filter. To determine the effects of dilution, flow rates of STE influent into the system were monitored, so that a mass balance of N could be determined. Over 50% of the reduction in N was attributed to processes other than dilution. Brooks et al. (1984) suggested that denitrification may be occurring at anaerobic microsites.

Tilstra et al. (1972) examined removal of N applied as SSE by peat in a saturated soil in Minnesota. A laboratory column filled with peat was established to determine if saturated conditions effected N removal. Naturally high concentrations of N occurred in the peat. Therefore initial application of SSE induced N leaching and higher concentrations of N in leachate than applied. Concentrations of N however decreased with time suggesting denitrification, NH_4^+ fixation, volatilization, or microbial assimilation occurring. More nitrification was observed in the lab column due to the aerobic conditions.

Spray irrigation is an alternative system takes advantage of the active microbial populations in surface horizons, as well as plant uptake of N or P. Nichols and Boelter (1982) examined N removal from secondary effluent from a campground for 8 years. Effluent was sprayed on a peat-sand filter bed. In the first year as bluegrass vegetation was established, only 46% of N applied removed by the peat filter. In the next two years greater than 90% of N applied was removed. Most of this was removed by the bluegrass in the last two years of the study. Although bluegrass uptake of N was > 100%, as much as 50% of the N applied was observed in the peat filter bed effluent. This N was the result of peat mineralization.

Hook and Kardos (1978) examined nitrate leaching after spray irrigation of SSE on woodland sites. One site, irrigated for 9 years, occurred in a hardwood forest. Soil textures ranged from sandy loam at the surface, to sandy clay loam or clay loam in the subsurface horizons. Soils with coarse sand textures also occurred within the same plot. The other site had been irrigated for 11 years. This site was an old field currently maintaining white spruce trees. Textures ranged from silt loam at the surface, to silty clay loam to clay in subsurface horizons. On average 93% of the N applied to the hardwood plots leached to the 120 cm sampling depth. Although only 20 to 50% of the N applied occurred in the NO_3^- form, all of the N collected was NO_3^-. Loading rates were examined on new plots of the hardwood site. In the first year, June - Oct., < 30% of N applied was leached. Between Nov. and May, 40 to 62% of N applied was leached to 120 cm. The following June - Oct., more N was leached than applied. Hook and Kardos (1978) concluded that in the first summer N was adsorbed on the exchange sites, taken up by plants or denitrified. In the winter plants were less active and more N was leached. In the following summer, stored N was nitrified and leached. Varying the loading rates showed that in plots with lower loading rates, less N was leached. In the white spruce plots (soil textures were finer) only 36% of the N added was leached, with concentrations rarely exceeding 10 mg L^{-1}.

Osborne (1975) applied SSE to a sand filter bed using spray irrigation. In the summer months, 66 to 70% of the N applied was removed by plants. In winter months however, treatment of SSE for N was poor with little reduction occurring.

Bouwer et al. (1976) examined the fate of N applied as SSE to an RI system (Flushing Meadows). The fate of N was controlled by the time between consecutive drying and flooding. When the system was flooded for two days, and followed by two days of drying, complete nitrification occurred and NO_3^- was observed in the system effluent. Longer periods of flooding and subsequent drying however, promoted denitrification. Bouwer et al. (1976) proposed the following sequence for N removal in the system. In first few days aerobic conditions prevail and NH_4^+ is converted to NO_3^-. During the next period, due to the O_2 demand, soil becomes anaerobic and remaining NH_4^+ is adsorbed on exchange sites. During the drying period NH_4^+ is converted to NO_3^- which is reduced at anaerobic microsites or when reducing conditions are established following flooding. Peaks of NO_3^- were observed at outlets documenting the proposed sequence.

Yamaguchi et al. (1990) examined denitrification in columns simulating RI systems. Columns were filled with 98% sand or decomposed granite. Synthetic wastewater was applied to columns under saturated conditions. Concentrations of NO_3^- varied between 20 and 60 mg L^{-1} and contained glucose supplements. At 30° C all loading rates showed complete denitrification. Only the lowest loading rates showed complete denitrification at 10° C.

Bauman and Schafer (1985) suggested that one septic system is of little importance, but thousands within a watershed can lead to significant pollution. Bauman and Schafer (1985) modeled ground water pollution due to N introduced from OSWDS. The results of these studies suggested that increasing lot sizes for homes in ground water recharge areas of susceptible aquifers, may be one of the most immediate alternatives to reducing the degree of N pollution to these aquifers.

Summary and conclusions

Nitrogen is the most widely studied ground and surface water contaminant attributed to OSWDS. Most N released from a ST-SAF is in the NO_3^- form which is very mobile in the soil system and can easily move into the ground and surface waters. Nitrogen contaminated ground water is the primary cause of methomoglobanemia, which is toxic to infants. Nitrate in surface waters can promote eutrophication.

Concentrations of total N in STE range from 40-80 mg L^{-1}. Most of this N is NH_4^+ (75-85%) or organic-N (15-25%). Several mechanisms and processes effect the fate and transport of N following application of STE. The complexity and degree in which each process affects N in the ST-SAF is dependent upon various soil and environmental factors. In general, immobilization, volatilization, chemical decomposition of NO_2^-, and plant uptake are of minimal importance in conventional OSWDS. Mineralization, nitrification, denitrification, and cation exchange are the most important processes contributing to the form and mobility of N in the system.

Mineralization is important in releasing organically bound N. This process is primarily restricted to the clogging mat and upper few centimeters below the gravel-soil interface of the ST-SAF, where most of the organic matter accumulates. Most N released during mineralization occur in the NH_4^+ form. The fate of this N, and the NH4+ in the STE, are primarily determined by cation exchange and nitrification.

Cation exchange is important in holding NH_4^+ on the exchange sites until nitrification occurs. Leaching of NH_4^+ can occur if the exchange sites become saturated with respect to NH_4^+, and conditions are not conducive for nitrification. Leaching of NH_4^+ into the groundwater occurs under saturated conditions, following application of STE at high loading rates, or in soils with a low CEC. High loading rates and saturated flow conditions increase flow rates which enhance movement of NH_4^+ away from a ST-SAF and into groundwater.

In most properly sited and operating OSWDS, NH_4^+ is rapidly oxidized to NO_3^-. Rates of nitrification are dependent upon available NH_4^+ or NO_2^-, pH, temperature, O_2 and soil moisture contents. The most important factor is the amount of available O_2, which the nitrifying microbes use as an electron acceptor. In aerobic soils most nitrification occurs within 10 cm of the ST-SAF clogging mat. Saturated soils show very little nitrification directly beneath the drain line. In these soils, NH_4^+ may move away from the ST-SAF in the lateral direction and into more aerobic conditions where nitrification can occur.

In most aerobic soils NO_3^- moves quickly and without reduction away from the ST-SAF and into the groundwater. In such cases dilution is the only form of N treatment between the ST-SAF and the groundwater. High concentrations of NO_3^- have been reported at considerable distance from OSWDS. Whelan (1988) reported concentrations of $NO_3^- > 50$ mg L^{-1} as far as 8 m from a ST-SAF. Walker et al. (1973b) reported 10 mg L^{-1} of NO_3^- as far as 70 m from an OSWDS. The rate at which NO_3^- moves away from the ST-SAF, and into the groundwater is primarily on flow rate. Greater flow rates increase movement of NO_3^-. Soils with coarse textures, strong structure, or under saturated conditions, are the most susceptible to rapid transport of NO_3^-.

Denitrification is the most important process for removing N applied as STE. In this biological process, NO_3^- is reduced to a gaseous N form which is released into the atmosphere. Rates of denitrification are dependent upon concentrations of NO_3^-, soluble C, O_2, as well as pH, temperature, and soil moisture content. The most important factors in denitrification are NO_3^- and soluble C levels, and anaerobic conditions. The Eh at which denitrification will begin depends upon the soil, but for most cases redox potentials between 200 and 300 mV are necessary. C sources used for denitrification varies between soils and systems. Highest rates of denitrification occur however, when a soluble carbon source is available.

Several field and column studies showed a positive correlation between soils with high water tables and degree of denitrification. Soils with high water tables have an anaerobic zone near the soil surface. In addition, these poorly drained soils have higher amounts of organic C at greater depths than better drained soils. If enough of an aerobic zone occurs in these soils above the saturated, anaerobic soil, for nitrification to proceed, subsequent denitrification can be substantial. Miller (1975) reported that levels of NO_3^- were much lower in wells located in poorly drained areas of Delaware, than the levels of NO_3^- in wells of well drained areas. These data suggest that substantial denitrification occurs in poorly drained soils.

Nitrate contamination of groundwater from OSWDS is more than a potential problem. Quan et al. (1974), Miller (1975), and Mancl and Beer (1982) all reported elevated levels of NO_3^- in groundwater attributed to OSWDS. Levels of NO_3^- in the contaminated groundwater in each of these studies were above 10 mg L^{-1}. Miller (1975) concluded that wells located in areas of Delaware in which depths to the water table were > 1.5 m were being polluted by N from OSWDS. Valiela and Costa (1988) estimated that 43% of the N entering a bay in Massachusetts could be attributed to OSWDS.

Alternative systems have shown mixed success in providing adequate treatment of N. The amount of success a particular alternative has is often dependent upon the individual system. Several studies indicated that nitrification occurred within mounds and subsequent denitrification occurred at the mound-soil interface. The amount of denitrification however, decreased with time as the C within the old soil surface became exhausted. These studies suggest that if mound systems are to provide adequate treatment for N, an additional C source such as methanol will be necessary.

Alternatives such as recirculating systems and denitrification units (RUCK system) may provide the best methods to reduce N from OSWDS. These systems have been reported to provide from 50 to 85% reduction in N concentrations. Addition of a soluble C source has been shown to increase the effectiveness if these units.

The efficiency of spray irrigation systems varies depending upon the vegetation, time of year, soil texture or filter media. In the winter when microbes are not as active and vegetation is dormant, little treatment of N occurs within a spray irrigation system. Grasses provided the best removal of N as long as they are removed from the site after mowing.

Other strategies which provide both aerobic and anaerobic conditions, such as periods of resting followed by increased loadings, or low loading rates applied to poorly drained soils using LPD, may also be effective alternatives for removing N from certain soils.

Restricting the use of OSWDS in high density areas, or increasing the lot size of homes in recharge areas of aquifers susceptible to N groundwater pollution, is an immediate step that can be taken to reduce pollution from OSWDS.

PHOSPHORUS

Of the elemental components in STE, the attenuation and movement of P within and away from ST-SAF is second only to N in the degree to which it has been studied. Reviews of the fate and transport of P applied to soils in wastewater include those of Bouwer and Chaney (1974), Miller and Wolf (1975), and Enfield and Bledsoe (1975). Reviews specifically related to P applied in wastewater from OSWDS can be found in Sikora and Corey (1976), Bicki et al. (1984), Cogger (1988), and Reneau et al. (1989).

Primary sources of P in domestic wastewater are P-based detergents and human excreta (Sikora and Corey, 1976). If P-based detergents are used, as much as 46% of the P in wastewater can be contributed by these detergents (Alhajjar et al., 1989). In the septic tank, P can be reduced as much as 48% from biological uptake and sedimentation before being applied as STE (Pell and Nyberg, 1989). Concentrations of P in STE range from 11 to 31 mg L^{-1} (Table 1). Most P in STE (85%) is in the orthophosphate form, with the remaining (15%) either as polyphosphates or organic P (Magdoff et al. (1974). Polyphosphates are rapidly converted to orthophosphates in the soil (Hashimoto and Lehr, 1973; Brandes, 1977).

Phosphorus is one of the most immobile elements in the soil system. Inhibition of P movement in the soil can be explained by plant uptake, adsorption to the soil, precipitation as a secondary mineral, or biological immobilization. Adsorption and precipitation are the two most important mechanisms of P removal in conventional OSWDS. Mechanisms which govern these processes have been widely studied with only mixed success. The apparent difficulty lies in determining which of the two processes control the fate of P following addition to the soil. Precipitation refers to a process in which phosphate forms with another compound to form a difficultly soluble, secondary mineral. Adsorption is a special case of precipitation in which the compound that P is adsorbed to remains in the original phase but reacts with P by a residual force (Hsu, 1965).

Phosphorus adsorption to soils is often examined in batch studies (closed systems) using isotherms. Sorption isotherms are curves which illustrate the amount of a given component sorbed at an interface versus the equilibrium concentration of that component within the system (Bache and Williams, 1971). Although based on gas kinetics and developed to describe adsorption of gases to a solid, the Langmuir equation is often used to describe the isotherms of sorption of ions in solution to solids. Olsen and Watanabe (1957) were the first to apply the Langmuir equation to describe the sorption of P to soil. The Langmuir equation offers several advantages over other isotherms. The primary advantages are that P adsorption can be described at low concentrations, and a means is provided for determining maximum adsorption and binding energy of P to a substrate (Olsen and Watanabe, 1957). A linear form of the Langmuir equation is:

$$\frac{C}{\frac{x}{m}} = \frac{1}{Vmb} + \frac{C}{Vm}$$

where:

C = equilibrium concentration of P in solution;
x/m = amount of P adsorbed per unit weight of absorbent;

b = a constant related to the binding energy of the adsorbent to adsorbate;
Vm = adsorption maximum.

Plotting C/x/m vs C, Vm and b can be determined.

Many researchers (Olsen and Watanabe, 1957; Seyers et al., 1973; Griffen and Jurinak, 1973; Taylor and Ellis, 1978; Munns and Fox, 1976; Sawhney, 1977) have reported P-sorption isotherms with two separate slopes. Olsen and Watanabe (1957) attributed the differences to differing reaction rates in adsorption vs precipitation. At low concentrations P was quickly adsorbed, however as coverage increased kinetically slower reactions such as precipitation occur.

Syers et al. (1973) examined P adsorption to three soils varying in clay and DCB-extractable Fe contents. A rearranged version of the Langmuir isotherm showed that P was being held with at least two different binding energies suggesting that P may be held tighter and in greater amounts than predicted by isotherm data.

Griffin and Jurinak (1973) examined P sorption to calcite. Langmuir plots showed two slopes which were attributed to either multilayer coverage, two different adsorption sites, or P precipitation. Data from BET isotherms indicated multilayer coverage. In addition, amounts of adsorption versus surface area indicated that only 5% of the surface area was covered. These data led Griffin and Jurinak (1973) to conclude that P was adsorbed at specific sites and was clustering at these sites. With time nucleation of the clusters occurred and Ca-P precipitates formed on the calcite. Calcite-P kinetics, as well as measurements of the isosteric differential heat during P adsorption, supported these conclusions. Taylor and Ellis (1978) examined P adsorption using Cl^- saturated anion exchange resins. In these studies, as P adsorption increased, the pH decreased, the Cl^- to P ratio decreased from 2:1 to 1:1, and a change in the isosteric differential heat of adsorption occurred. These changes indicated that P was being adsorbed at two sites. At low P concentrations and Cl^- ratios of 2:1, P was adsorbed in a bidentate form. As coverage increased, P was held in a monodentate fashion.

Sawhney (1977) and Munns and Fox (1976) reported P sorption to occur in two phases. The initial reaction was fast. The second was much slower, and followed first order kinetics. Sawhney (1977) showed that by increasing reaction time from 40 to 916 hours, P adsorption would double. Sawhney (1977) concluded that batch studies over short periods may not indicate the total amount of P a soil can adsorb. Munns and Fox (1976) showed that the time needed for P to reach equilibrium was considerable and depended on soil type. Equilibrium was reached for an Andept in about 50 days. Oxisols, however, required between 100 and 200 days to reach equilibrium.

Although standard isotherm plots are most often used to study P sorption, several researchers contend that P attenuation can be best described by solid-solution reactions (Lindsay and Moreno, 1960; Lindsay and Vlek, 1977; Blanchar and Stearman, 1984; 1985). To examine solid-solution reactions, P concentrations at equilibrium are related to the dissociation constants of the Al, Fe, or Ca compounds thought to control P activity. These values can be plotted on a solubility diagram to show theoretically which minerals control P in the soil system at a given pH and P concentration. Blanchar and Stearman (1985) reacted potassium dihydrogen phosphate or monocalcium phosphate with various Missouri soils. These researchers found that P sorption could be well correlated with regular solid-solution systems. Lindsay and Moreno (1960) and Lindsay and Vlek (1977) found similar results.

Regardless of whether adsorption or precipitation governs the P reactions in the soil, most researchers agree that P reacts primarily with Ca, Fe, and Al soil compounds (Miller and Wolf, 1975; Enfield and Bledsoe, 1975, Sikora and Corey, 1976). Yuan et al. (1960) examined P partitionment in 3 coastal plain soils of Florida. Over 80% of the P was associated with either Fe or Al. Using the Langmuir equation to examine sorption of P to surface horizon of a sandy siliceous Haplaquod, Van der Zee et al. (1988) found that the adsorption maximum (Vm) was proportional to the sum of the oxalate-extractable Fe and Al. In a similar study, Van der Zee and van Riemsdijk (1988) reported that oxalate extractable P is a good estimator of total P, and P is primarily associated with oxalate extractable Fe and Al. Beek et al. (1977) partitioned the P components of a soil in which domestic sewage was applied for 30 to 50 years. Most of the P was attenuated in the upper 50 cm. From 15 to 20% of the P occurred in the organic fraction, with the remaining P in combination with Fe or Al. Most P appeared in the Al form.

Stuanes (1984) added various concentrations of P in solution to 12 soils with clay contents of < 10%. Domestic wastewater was also added to 3 of the soils. In soil-solution experiments, soils governed the pH (pH ranged from 5.0 to 8.6). Samples in which wastewater was added, pH was controlled by wastewater. Wastewater sorption was less than solution adsorption for same soils, suggesting that precipitation was not of much importance (Stuanes, 1984). Step-wise regression techniques indicated that only pH and DCB extractable Al could be correlated significantly with P adsorption.

Whelan (1986) examined P concentration and mineralogy in slime layers below 4 ST-SAF using both SEM and microprobe analysis. Slime layers were higher in P, Al, and Fe than controls. Calcium was higher in slime layers of acidic soils. Discrete P minerals were observed in the slime layer of a 16 and 17 year old ST-SAF. Phosphate minerals forms were either Ca-P or Fe-P. Ratios of Ca:P were similar to octacalcium phosphate, Fe:P ratios were not similar to strengite. Solubility diagrams showed P at equilibrium under saturated with respect to octacalcium phosphate. Whelan (1986) concluded precipitation reactions were too complex in these systems to be described by solubility diagrams. No Al-P minerals were observed.

Van Riemsdijk et al. (1977) added P to either $Al(OH)_3$ or Al_2O_3. The subsequent reaction occurred in 2 steps. An initial fast reaction followed by a much slower secondary reaction which was assumed to be precipitation. The first reaction occurred in about 1 day for $Al(OH)_3$ and 4 days for $AlCl_3$. X-ray diffraction indicated an $AlPO_4$ compound, called Sterrettite, had formed, confirming that precipitation had occurred.

Kao and Blanchar (1973) examined the distribution of P in a silt loam soil that had been fertilized for 88 years. Solid-solution data suggested that the Fe-P mineral strengite was controlling P in the system.

The role of organic matter and the distribution of sand, silt, and clay (PSD) in P sorption is not clear. Harter (1969) equilibrated 15 samples taken from A or Ap horizons with P solutions. Samples were small (0.5 g) and equilibrated for only 1 hour. Regression techniques showed that P was initially bound to organic matter, and then later transformed to Fe and Al compounds. Soils were low in DCB-Fe (< 1.4% Fe_2O_3) and clay (3-14%) but high in organic matter 5-21%. Murti et al. (1966) examined adsorption of LAS to several soils. A highly significant correlation was observed between LAS adsorption and P fixing capacity of the soils examined. Soils treated with DCB showed no adsorbing potential compared to 19 to 48 ug g^{-1} for untreated soils.

Soils treated with H_2O_2 to remove organic matter showed an increase in LAS adsorbing capacity to 17 to 65 ug/g, suggesting that Fe and Al were released during the oxidation of organic matter creating more adsorption sites.

Reddy et al. (1980) examined P sorption after applying animal wastes to A horizons of Norfolk and Cecil soils. Sorption of P was less for soils treated with higher rates of animal wastes. Reddy et al. (1980) concluded that organics in animal waste may chelate Fe and Al and reduce sorption sites. Therefore, when P and organic ions occur together, adsorption will decrease due to competition for exchange sites from the organic ion. The ability of the organic ion to decrease P sorption would depend upon the stability of the Fe or Al-P complex, and the competing Fe or Al-organic complex.

Fox and Kamprath (1970) examined P adsorption to 5 soils which included a clay loam with 5% Fe_2O_3, sandy loam, sandy loam with 10% organic matter, and a peaty muck. Clay loam adsorbed at least twice as much P as the others at equivalent P in solution. Adsorption per gram of clay however decreased as clay content increased.

Fox and Kamprath (1971) added P to a muck and sand with 10% organic matter and allowed the samples to equilibrate for 6 days. Only at high P concentrations was P adsorbed. Complete adsorption occurred with addition of $AlCl_3$ to these soils.

Batch studies are often used to determine reaction rates (P sorption kinetics), or the maximum amount of P a soil can sorb (Vm of the Langmuir equation) for waste disposal (van Riemsdijk et al., 1977; Sawhney, 1977; Nagpal, 1986; Sawhney and Hill, 1975; Reddy et al., 1980). Stuanes (1984) contended however, that the single addition with a long equilibrium period in the common batch procedure is not equivalent to the dosing used for waste disposal. Stuanes (1984) proposed that the secondary slow reaction following P sorption will continue, while a new dose of effluent is added to the soil. To account for these conditions, additions were made consecutively (10 or 20 times) to simulate wastewater application. The Langmuir equation predicted only 25 to 58% of the adsorption that occurred after 10 additions, suggesting that Langmuir underestimates P adsorption. Adsorption after 10 additions versus Vm showed a high correlation ($r^2 = .96$). Stuanes (1984) concluded that if a conversion factor can be found between Vm and adsorption after consecutive additions, Vm can be used to predict adsorption of P following wastewater application.

Although P is generally considered immobile in the soil system after initial adsorption due to secondary precipitation reactions, P desorption can occur. Stuanes (1984) observed that 6 to 35 % of P adsorbed after 10 additions of wastewater, could subsequently be desorbed. Soils with lower sorbing capacity showed greater amounts of desorption.

Hill and Sawney (1981) examined P movement under saturated and unsaturated conditions in an undisturbed test cell. Concentration of P in the effluent from the cell increased dramatically during the anaerobic (saturated) conditions. Hill and Sawhney (1981) concluded that although less adsorption of P was occurring during saturated conditions because of increased flow rates, desorption of P may also occur under saturated or anaerobic conditions.

Barrow and Shaw (1975) examined desorption of P from 10 soils. If solution:soil ratios were small desorption was complete in a short period. Desorption of samples with high solution:soil ratios however showed continued desorption with time.

Reddy et al. (1980) examined P desorption after applying animal wastes to Norfolk and Cecil soils. Desorption increased as loading rates increased. Organics in animal waste may chelate with Fe and Al. Therefore, when P and organic ions occur together, adsorption will decrease and desorption increase as organic ions compete for the exchange sites. More P was desorbed from the A horizon of the Norfolk soil (3% clay) than the A horizon of the Cecil soil (16%) clay. The DCB or ammonium oxalate extractable Fe and Al contents were not given.

Munns and Fox (1976) examined P desorption following addition of P solutions to 3 Oxisols and an Andept. Rates of desorption differed between soils, but the rates were always much slower than adsorption. Samples were allowed to react with P solutions for 1, 7, 9, or 40 days prior to desorption extractions. Samples with longer reaction times for P adsorption or precipitation, consistently showed less desorption. Samples with short reaction times (1 day), desorption was rapid, but the amount of P desorbed rapidly decreased. At longer times of P equilibration (7, 9, and 40 days), P desorption was rapid but, the decrease in desorption was much less rapid than 1 day experiment. Higher rates of desorption were also found with soils treated with higher loading rates. Similar results were reported by Cho et al. (1970), Fox and Kamprath (1970), and Barrow and Shaw (1975).

Column Studies

In order to simulate an open system similar to field conditions, soil column studies are often used to examine P sorption. Batch studies are often include with these experiments for comparative purposes. Sahwney (1977) examined P adsorption and movement in saturated columns filled with either 5 cm of silt loam or fine sandy loam soil. Phosphorus was added in concentrations of 2 x 10^{-4} M (equivalent to that in domestic wastewater). Approximately 50 pore volumes passed through the fine sandy loam column before P breakthrough had occurred. About 100 pore volumes were necessary for breakthrough to occur in the finer textured, silt loam soil. Adsorption values were equivalent to amounts predicted by Langmuir. After breakthrough, P concentrations in column effluent increased, but P continued to be adsorbed above that predicted from the isotherms. Nagpal (1986) examined P movement in 12-16 cm columns packed with loamy sand and sandy loam. Bulk density of columns packed with loamy sand were 1.61-1.67 g/cm^3, but bulk density values were only 1.18 to 1.28 g/cm^3 for the sandy loam. More P moved through sandy loam columns which contained larger pore spaces than the tightly packed columns filled with loamy sand. Sorption isotherms were not conducted for correlation with the column results.

Tare and Bokil (1982) examined P movement in columns from 7.5 to 75 cm in length. Columns were filled with a mixture of sand, silt and clay. Sand contents varied from 0 to 100%, with clay contents varying from 0 to 30%. Wastewater was applied to columns for 30 days. No P was observed below 15 cm in soils with greater than 20%

silt and clay. In sandy soils however, P was in contact with soil particles for a shorter period of time because of increased flow rates, and P moved farther in the column (Tare and Bokil, 1982). Concentrations of P at 15 and 30 cm increased as the number of additions increased.

Nagpal (1985) added STE daily to columns filled with 60 cm of gravelly silt loam. Columns were designed to represent high (partially saturated columns) and low water (unsaturated columns) tables. Phosphorus was observed in column effluent within a day or two of initial additions to columns representing high water tables. Between 15 and 20 pore volumes passed through the unsaturated column before P was consistently observed in effluent. Concentrations at P breakthrough were also twice as high in the partially saturated columns. With time, P concentration in column effluent reached an equilibrium concentration in both columns. Equilibrium concentrations were more than twice as high for the columns simulating a high water table (1.7-2.1 mg L^{-1}), compared to the unsaturated columns (0.13-0.8 mg L^{-1}). Flow rate and P concentration at equilibrium were highly correlated (r^2 =.974), with higher flow rates giving higher equilibrium concentrations. On average 92% of P was attenuated in the unsaturated columns but only 70% in the partially saturated column. Most P was adsorbed in the upper 30 cm, although P was found below this depth. After 2.7 years 60 to 90% of the P applied was still being removed. These values were not predicted by P isotherms from batch studies.

Sawhney and Hill (1975) examined P movement in undisturbed cores 90 cm in length from six Connecticut soils after application of simulated wastewater. Most adsorption occurred in the upper 7.5 cm of soil. Batch experiments showed that after 2 years of wastewater application, from 10 to 63% of the P adsorption capacity remained in upper 7.5 cm of column, and between 58 and 97% for the 7.5 -15 cm depth.

Most of the batch studies previously presented indicated that organic matter adsorbs little P. Several column studies have been conducted to confirm these results. Rock et al. (1984) applied STE to columns 10 to 90 cm in length filled with peat. Less than 10% adsorption of P was observed. Fox and Kamprath (1971) examined P movement in 40 cm columns filled with an organic muck with virtually no mineral matter and a sand with 10% organic matter. Nearly all of the P added in a fertilizer form was leached.

Larsen et al. (1958) examined P movement in columns filled with 30 cm of soil. Columns were prepared from 5 different organic soils and a mineral soil. Phosphorus was applied and the soils leached with water. For a virgin muck soil, 61% of the P applied appeared in the leachate after 38 cm of water had been applied. The mineral soil and organic soils that had been drained for 15 years retained all of the P. Phosphorus retention was related to sesquioxide content, which was much higher in the mineral soil and the drained organic soil.

Columns are often constructed to simulate alternative OSWDS such as mounds or filter beds. Magdoff et al. (1974) constructed soil columns to represent a mound system. Phosphorus concentrations were monitored for 110 days during which STE was applied. Solutions from 55 cm of unclogged columns showed a 29% reduction in P after 94 days. An additional 30% reduction was observed in the silt loam material underlying the sand fill material. Clogged columns showed a greater decrease in P concentration with effluent from columns showing 91% reduction in total P. The initial reduction in P was thought to represent adsorption of P to the fill material, but the continued removal of P with time, and the steady state conditions after time, suggested that precipitation was controlling P attenuation (Magdoff et al., 1974). Levels of Ca and P in the system, as well as solution pH, indicated that octacalcium phosphate may be controlling P in the mound system. After continuous ponding due to clogging, P levels increased in column effluent reaching 60% of the STE.

Magdoff and Keeney (1975) examined the fate of P applied as STE to columns representing mound systems. Accumulation of P was observed in the silt loam former top soil indicating that P was moving through the sand mound. Only 56% of the P applied was removed.

Willman et al. (1981) examined P attenuation in a column filled with crushed limestone, sandstone, and shale. Columns were set up to simulate a typical mound system installed in Pennsylvania. For each sand type, 0, 3, 6, and 12% clay were added. Clay added was obtained from the Bt horizon of a Hagerstown soil (high DCB extractable Fe and Al content). All of the sands and mixtures, with the exception of the 0% clay-sandstone mixture attenuated all of the P after 23 weeks. Effluent from the sandstone-0% clay-sandstone column was of equal P concentration of the column influent.

Pell and Nyberg (1989) examined P movement through columns set up representing sand filter systems. Initially all P was adsorbed suggesting a fast reaction. After time, a secondary reaction occurred which was less effective in attenuating P, and P was present in the column effluent.

Devries (1972) examined P removal from SSE applied to columns representing a sand filter system. Sands used for the columns that had substantial sesquioxide coatings removed nearly 100% of the P applied. Sands without coatings however had little effect on P removal.

Lance (1977) examined P removal from wastewater in columns filled with 250 cm of calcareous loamy sand. Columns were flooded for 9 days and rested for 5 days to simulate conditions of at the Flushing Meadows RI-wastewater treatment site. Removal decreased as infiltration rates increased. This trend was especially apparent after P adsorption capacity had been reached, suggesting that precipitation with Ca may be occurring. Growth of Bermuda grass increased P in effluent. Lance (1977) suggested several explanations for these results including: organic acids from the plants chelating Ca, organic ions competing for adsorption sites; Ca uptake; pH reduction due to organic acids reducing precipitation; or a combination of these factors. Drying time apparently increased adsorption. From 75-80% of the P was attenuated when infiltration rates were kept below 15 cm/day.

Rock et al. (1984) constructed columns to represent a peat filter system. Less than 10% of P applied in STE was attenuated by the peat filter. This attenuation was attributed to biological assimilation because Fe and Al contents were both < 0.08%. A cover crop did not increase P losses when effluent was applied subsurfacely.

Field studies

Valiela and Costa (1988) examined P concentrations in the groundwater entering a bay in Massachusetts. An estimated 38% of the P added to the bay came from STE. On average, ratios of N:P for STE were 6:1, however in the groundwater the ratios of N:P were 25:1. These data suggest that P attenuation is considerable larger than that of N. Movement of P within a soil depends upon the soil texture and structure, pH, Fe, Al, and Ca content, depth to the water table, depth to an impermeable layer, and the P loading rates. Most research has shown in soils with textures finer than loamy sand and unsaturated conditions, movement of P is limited, and most P applied as fertilizer or in wastewater is attenuated by the soil. Kao and Blanchar (1973) examined the distribution of P in a silt loam soil that had been fertilized for 88 years. Although the fertilized field showed greater P levels than the control, the distribution of the P between horizons was similar. Laboratory data from sorption isotherms showed that the fertilized soils had nearly as much capacity to sorb P as the control. These data suggest little evidence for mobility of P even after 88 years of application.

Beek et al. (1977) examined the distribution of P within a sandy soil (< 3% fine silt or clay) in which raw sewage had been applied for 50 years. Most of the P was attenuated in the upper 50 cm with minimal amounts below this point. Below 75 cm P concentrations were equal to the control profile suggesting minimal movement of P.

Alhajjar et al. (1989a, 1989b) examined P movement from ST-SAF placed in glacial sediments ranging from outwash to loess. Water tables were within 3 meters of soil surface. Median-Polish (best-fit type) statistics showed that there existed less than a 10% probability that greater than 1% of the P applied to these ST-SAF would reach the ground water.

Although uncommon, movement of P below ST-SAF with fine textured soils has been reported. Viraraghavan and Warnock (1975) examined the fate of P following addition of STE to a sandy clay to clay soil in Ottawa, Canada. Water samples were taken from below the ST-SAF for 60 weeks. Samples showed that only 25 to 50% of the P was attenuated. An explanation for the low rates of P reduction was not given (profile morphology or flow rates were not given in the paper).

Sawhney and Starr (1977) examined P movement in a 6 year old ST-SAF in Connecticut. Soil characterization data was absent. Within 40 days after the ST-SAF was re-established, concentrations of P in soil solution 15 cm below trench bottom were nearly equal to concentrations of STE. Concentrations of P in soil solution 30 cm below the trench bottom indicated that between 57 and 95% of P was being attenuated. Reduction of P was greater than 93% at 60 cm depths. Sawhney and Hill (1977) concluded that adequate P removal had occurred within 60 cm of the trench bottom, but not within 30 cm.

Attenuation of P in soils with textures of loamy sand or coarser appears to be limited. These soils generally have low concentrations of reactive Fe and Al. In addition, flow rates are generally greater than finer textured soils, which limits the contact time between P in solution and the soil particles. Whelan (1988) examined P distribution below a soak-hole and drain line placed in calcareous sands (pH 9.2). Distribution of P was nearly constant in the upper 5.5 m of the regolith, below which the P content sharply declined. Whelan (1988) concluded that the soils within 5.5 meters of the soak hole had reached their P adsorption maximum, and P was moving through these horizons without a reduction in concentration.

Whelan and Barrow (1984) examined P concentrations in soil and solutions below a soakwell and leach drain in Australia. The soil was composed of greater than 90% sand. Elevated P concentrations were observed in these sands at more than 8 meters below the soil surface. Concentrations of P in the pore water were as high as those in the STE suggesting that P was not being attenuated, and P was leaching into the groundwater. Adsorption capacity predicted by isotherms were highly correlated with P levels in the field suggesting that soils had reached their P adsorption capacity.

Bouwer et al. (1974) examined P movement in a rapid-infiltration system composed of sand and gravels in Arizona (Flushing Meadows). In the upper 9 meters of the deposits only 50% of the P was removed. The limited amount of Fe and Al in these deposits suggested that Ca-P precipitation was the primary method for P attenuation (Bouwer et al. 1974). Microprobe analysis of sediments below RI basins showed P-rich coatings and soil particles confirming precipitation.

Ver Hey and Woessner (1988) examined P concentration below a ST-SAF placed in sandy or sandy-skeletal alluvial sediments. The concentrations of P were nearly the same as the STE even at depths of 2.4 to 14.3 m. These data suggest that no treatment for P was occurring in the system.

Depth of the water table and the degree of macro structure can effect the movement and distribution of P within a soil. Hill and Sawhney (1981) examined P movement in soil by adding simulated wastewater to a block of soil (surface area of 9.25 $m^{2)}$. The undisturbed soil block was a shallow, moderately-well drained fine sandy loam located at a foot slope position. Flow rates and P concentrations in soil solution were measured at 45 and 70 cm. Flow rates varied around the cell. These variations were related to the proximity of natural flow paths in the soil. About 5.6 pore volumes were necessary before P appeared in effluent at 45 cm. Only 10% of the P sorption capacity calculated from isotherms were filled at this time. About 10.8 pore volumes were necessary before breakthrough occurred at 70 cm. The time before breakthrough as well as the necessary pore volume was lower than that observed in lab columns by Sawhney (1977). These differences were related to the channels and other continuous voids that occur in natural soils. High water tables increased the flow rates as well as the amount of P in the effluent. After 2.5 years of application, 90 and 80% of the

P applied was adsorbed by the soil within 45 and 70 cm respectively. Most P was adsorbed in the upper 15 cm.

Lateral movement of P from a site is generally minimal in a well-drained site with substantial P adsorption capacity, but in soils with high water tables lateral movement of P can be considerable. Sawhney and Hill (1975) sampled soils at depths of 60 cm around a 12 year old SAF placed in a sandy soil. Batch studies showed that soils within 15 cm of the SAF continued to adsorb P. Samples taken from 2.1 m from the SAF showed similar P sorption capacity as the control suggesting movement of P did not occur beyond this point.

Jones and Lee (1979) examined P movement away from a ST-SAF located in glacial drift in Wisconsin. Wells were established down-gradient of the SAF. Closest wells were 13 m away, with some wells as far as 75 m away. Wells were installed either 1.5 or 4.5 m below water table (depths to water tables were not given). Concentrations of P in groundwater were not above background levels at the closest wells after 4 years.

Reneau (1979) examined lateral movement of P from 3 ST-SAF on the coastal plain of Virginia. Soils were either fine -loamy or coarse loamy Ochraquults. Sampling wells were placed in the direction of the groundwater flow pattern, and between each of the three ST-SAF and an agricultural drainage tile located 12 to 19 m from the ST-SAF. High water tables were observed at various times during the study period. The amount of P in water samples was shown to decrease with distance from the ST-SAF. Measurable amounts of P however were observed in wells at greater than 17 m from the ST-SAF. Using regression lines, Reneau (1979) estimated that for one of the sites a distance of 30 m was necessary before 99% of the P would be attenuated. At least 8 m would be necessary for equivalent reduction at the other two sites. From 64 to 88% of the P in the argillic horizons at distances of 15 to 456 cm from the ST-SAF, occurred in combination with Fe or Al. Argillic horizons of the control pedon showed approximately 20 % Fe and Al bound P.

Bouwer et al. (1974) examined P movement in a rapid-infiltration system composed of sand and gravels in Arizona (Flushing Meadows). Wells 30 m from effluent discharge showed only 90% reduction in P levels.

Reneau and Pettry (1976) examined P in the soil and groundwater adjacent to a 4 and 15 year old ST-SAF in the Coastal Plain of Virginia. The 15 year old site was located in a Plinthic Paleudult and the 4 year old system in an Aquic Paleudult. Concentrations of P decreased with distance from the ST-SAF. Elevated concentrations of P however were found at distances up to 13.5 m from the ST-SAF at upper sampling depths. Below the impermeable plinthic or fragic layers of these soils, little P was observed even 15 cm from the ST-SAF. Most P was found in the argillic horizons of the pedons. Sixty-four percent of the total P was found in the extractable Fe and Al form. In the Plinthic Paleudult elevated concentrations of P were observed in the soils 3 meters from the ST-SAF. These concentrations were not observed in the Aquic Paleudult even at 15 cm from the ST-SAF. This suggested that little P movement occurred in these soils after 4 years.

Cogger et al. (1988) examined P movement applied as STE by LPD to a Quartzipsamment in North Carolina. Two adjacent sites were examined in which the height of the water table differed by 30 cm. Wells were installed 0.3 to 3 m from each ST-SAF trench. Higher concentrations of P were observed in the water samples taken from wells adjacent to the ST-SAF with the high water, then ST-SAF with the lower water tables. In the summer months when water tables at both sites were at that their lowest levels, little P was found in wells of either system. Cogger et al. (1988) concluded that P adsorption was governed by flow velocity. Under high water table conditions, P moved away from the ST-SAF. In cases where the water tables were low and flow velocity was essentially zero, P adsorption occurred similar to fast reactions which are often observed in batch studies. Cogger et al. (1988) concluded that a separation distance between drainage tile and the high water table should be greater than 30 cm, and that 60 cm was most likely sufficient. Evidence of P desorption following resting was observed, but desorption following heavy rainfall was not observed (Cogger et al., 1988). These differences in desorption effects could not be explained.

Stewart and Reneau (1988) examined P movement in soils with high water tables. Low pressure distribution was used to distribute STE. Movement of P was not observed away from the ST-SAF.

Johnston et al. (1962) examined P concentrations in tile drainage effluent from irrigated fields in California. Concentrations of P were only elevated during times of increased flow following irrigation, or when the water table was higher than normal.

Rea and Upchurch (1980) examined P movement as a result of application of STE to a Grossarenic Paleudult and an Aeric Haplaquod with < 3% silt and clay. Wells were established at five points along a 30 m toposequence. P concentration in groundwater and soils were determined along the sequence. Most P was found to move in the vertical direction even though a gradient to the water table existed beyond the ST-SAF. Most P was attenuated within 15 m of the source.

Whelan and Barrow (1984) examined P concentrations in a sandy soil (> 90% sand) adjacent to a soakwell and leach drain in Australia. Soil samples obtained a meter below the soil surface showed elevated P concentrations as far as 4 m from the soakwell.

Loading rate effects

The loading rates of P applied as STE can often effect the P concentration leaving a ST-SAF and entering the groundwater. Cogger et al. (1988) examined the effect of loading rates on P movement in a Quartzipsamment in North Carolina. The STE was applied using LPD systems at rates of 10, 40, and 160 L m^{-2} per day. Significant differences were observed at .05 level for all three loading rates for the system with high water tables, and the two lower rates were significantly different than the higher rate in the soil with lower water tables. Cogger et al. (1988) recommended that in sandy soils loading rates should remain at or below 40 L m^{-2} per day.

Cogger and Carlile (1984) examined the effectiveness of 15 conventional and alternative systems in soils with high water tables. Phosphorus content in the ST-SAF effluent and groundwater were higher for the systems in which loading rates were highest. Levels of P in the groundwater around 6 of the systems were above 1 mg/L. High water tables were not correlated with high P concentrations.

Uebler (1984) examined the effect of three loading rates of 7.5, 11.3, and 15 L m^{-2} per day on P adsorption to a Cecil soil in North Carolina. No effects due to loading rates could be observed. Greater than 95% of the P applied was attenuated by the Cecil soil which has a high P adsorption capacity. An increase in P from .1 to .9 mg/L was observed by the end of the first year. Uebler (1984) suggested that this difference may have been due to adsorption of P by the ceramic sampling cups at the beginning of the experiment.

Nagpal (1986) examined the effect of P concentration and loading rates on P attenuation in a loamy sand and sandy loam. Effects were confounded by an increase in the flow rate with increased loadings, so that any loading effects were masked.

Fate of P in alternative systems

Numerous researchers have examined P attenuation and movement within and away from various alternative OSWDS. The success of the alternative systems in reducing P concentrations have met with mixed success. Stewart and Reneau (1988) examined P movement in soils treated with STE. These soils had high water tables and STE was applied using LPD system. Soils were poorly drained fine loamy, siliceous, Typic Ochraquults. Little movement of P could be detected. Stewart and Reneau concluded that because an even distribution of effluent was realized, trench bottom loading rates were low, and saturated flow conditions were eliminated, P adsorption was nearly complete.

Cogger et al. (1988) examined P movement applied as STE by LPD to a Quartzipsamment in North Carolina. Higher concentrations of P were observed in the water samples taken from wells adjacent to the ST-SAF with high water tables, then

ST-SAF with the lower water tables. Cogger et al. (1988) suggested that P adsorption was governed by flow velocity, therefore under saturated conditions and greater flow rates P will move away from the ST-SAF and into the groundwater.

Brooks et al. (1984) examined P concentrations of effluent leaving several peat filter systems. A 58% reduction in the concentration of P was observed in one of the filter systems. This value was much higher than that reported for the associated column study (Rock et al., 1984). Dilution was expected to be the leading cause. To examine the effect of dilution on P concentration in the field, another system was designed and sampled. Flow into and out of this filter could be monitored, and a mass balance of P within the system could be made. Concentrations of P were reduced by 63%, but only 33% could be attributed to attenuation, with the rest due to dilution. Effluent from the peat filters were also treated in a sand and gravel trench which was absent from the laboratory columns. The Fe, Al, and Ca, in the trench may have attributed to the additional attenuation of P. In the third system, effluent passes through the filter bed and into the shallow mineral soil below the bed much like a mound system. A reduction in this system of 96% of the P was reported. Again dilution may have played an important role in this reduction in P concentration.

Tilstra et al. (1972) examined removal of P applied as SSE by peat over the 3 month study period in a saturated soil in Minnesota. A laboratory column filled with peat and maintained in unsaturated conditions was established to determine if saturated conditions effected P removal. Removal of P in the column was nearly complete (95-99% removal), with concentrations of effluent remaining the same with time. Field studies showed from 76 to 92% reduction in P, with the amount of attenuation decreasing with time.

Nichols and Boelter (1982) examined P removal from secondary effluent from a campground for 8 years. Effluent was sprayed on a peat-sand filter bed. Almost all of the P was removed (>99%). The peat was high in Al and Fe, as well as mineral material which probably increased the sorbing capacity. Greater than 37% of the P applied was attenuated in the upper 12 cm of the filter bed. Bluegrass growing at the bed surface removed between 22 and 45% of the P applied. The amount taken up by the plants increased with time. Nichols and Boelter (1982) concluded that with time P would probably start moving through the filter bed when the P sorbing capacity of the filter medium was reached.

Rock et al. (1982) examined the efficiency of a peat bed for removal of P from STE. The first year as much as 70% of the P applied was removed by the filter. By the third year however, less than 33% of the P applied was reduced in the filter.

Pell and Nyberg (1989) examined P movement after application as STE to a sand-filter system and columns representing sand filter systems. Initially P adsorption was greater in the column then sand filter. After 4 to 5 weeks both sand filter and columns reacted similarly to their attenuation of P. After 78 days the both columns and sand filters were removing 70% of the P applied. Pell and Nyberg (1989) concluded that > 75 cm of sand is needed for attenuation of P, and columns do not always perform the same as sand filter system.

Osborne (1975) applied SSE from a campground to a sand filter bed using spray irrigation. In the first year greater than 99.6% of the P was attenuated. Blue grass was planted on the surface of the filter. Filter bed was for a campground.

Brandes et al. (1975) examined several filter beds and columns for removal of P from STE. Beds and columns were filled with mixtures of sand, crushed limestone, clayey silt, and Al mill waste called red mud which is high in Al and Fe. Red mud attenuated > 96% of P applied, with only 71 and 76% of P attenuated by the crushed limestone, and crushed limestone-sand mixture respectively.

Sikora et al. (1978) examined the efficiency of a denitrification unit filled with dolomite rock on P removal. Initially P was removed within the unit but with time the amount of P removed decreased because the adsorption sites for P were covered with a microbial mass.

In some systems ST-SAF are constructed so that for an extended period one half of the field receives the STE, and then this first field is rested, and the other half of the field begins to treat the STE. Sawhney and Starr (1977) examined the relationship between sorption capacity and resting in a 6 year old ST-SAF in Connecticut. Resting the ST-SAF for six months was shown to regenerate some of the soils P adsorbing capacity. Hill and Sawhney (1981) examined P adsorption in an undisturbed test cell over a two year period. Resting the soil over the first winter rejuvenated some of the sorption sites. Resting over the next winter, however, was not as effective.

Phosphorus conservation in the household or treatment of the wastewater in the ST may be some of the better alternative methods. Brooks et al. (1984) suggested that reducing the P in wastewater may be the best alternative system. In this study, P concentrations between the two different sources of STE were 32.2 mg/L, or an 81% difference. These differences suggest that reducing the usage of P-based products can substantially reduce the potential for P contamination in the groundwater due to OSWDS.

Alhajjar et al. (1989a, 1989b) examined P movement from ST-SAF placed in glacial sediments ranging from outwash to loess. Water tables were within 3 meters of soil surface. Eight of the households studied used P-based detergents. The other 9 sites used a CO_3 based detergent. The P-based detergent accounted for 46% of the P in the STE. Best fit type of statistical analysis (median-polish) suggested that the chance of P concentrations greater than .1 mg/L reaching the groundwater was equivalent for either of the detergents. The P-based detergents however showed a 10% probability that greater than 1% of the P applied would reach the ground water.

Brandes (1977) examined the addition of alum to a septic tank to improve the removal of P within the septic tank. Findings from this study indicated that only 4% of the P content in wastewater was reduced in the septic tank, although retention time was as much as 10 days. This is in contrast with the data presented by Pell and Nyberg (1989) who reported a reduction in P levels of 48% in the septic tank.

Addition of alum (Brandes, 1977) reduced P levels in STE by as much as 96%. Introduction of alum increased on average the sludge accumulation rates within the septic tank by 2.35 times.

Summary and conclusions

The fate and transport of P from OSWDS have been extensively studied. Detergents which are P based and human excreta are the major sources of P in domestic wastewater. Most P in STE is in the orthophosphate form and is readily adsorbed to soils which contain reactive Fe, Al, or Ca compounds. Organic or mineral soils with minor amounts of Fe and Al show minimal P sorption capacity. Primary processes in P attenuation are adsorption and precipitation. Most researchers agree that initially P is adsorbed to the soil. This reaction occurs quickly, with the secondary reaction much slower. The secondary reaction may involve precipitation or adsorption to secondary sites. Batch studies producing sorption isotherms appear to be the most efficient means of examining P sorption to soil. Typical batch studies may not effectively determine the capacity of a soil to adsorb P. This may especially be the case when P is added in consecutive doses such as in OSWDS. Phosphorus sorption sites may also be regenerated following a resting period. These effects cannot be predicted using simple batch studies.

Desorption of P is always much slower than adsorption but the effect can be considerable when anaerobic conditions exist, organic ions are present, or when the concentration of P in the soil solution is very dilute.

Many factors effect the fate and transport of P in a soil including: Fe, Al, and Ca content, flow rates, P concentrations in soil and solution, particle size distribution, pH, and depth to water table. In most soils in which Fe and Al are present in a reactive form, and flow rates are minimal, P movement is minimal and concerns for pollution of ground or surface waters from P applied in a OSWDS are unfounded. Phosphorus pollution to the groundwater can occur in cases where the water tables are near the surface, soils have coarse textures, flow rates are increased due to strong soil structure, loading rates are high, soils have a low P adsorption capacity, or when P capacity of a soil has been met.

The major factor contributing to P movement is the flow rate. Increased flow rates are generally associated with soils of coarse textures, strong structure or macro voids associated with biological activity, or high water tables. Loading rates may also effect flow rates by increasing the amount of wastewater applied. In cases with high flow rates, contact time between P and the soil particles are reduced and P adsorption and precipitation are not as effective. These factors can be considerable when the adsorption capacity of a soil has been reached and the mechanism for P attenuation is the slower precipitation reaction. Hill and Sawhney (1981) showed that P could move through a soil in which only 10% of the P adsorption capacity had been filled. Movement of P was thought to occur along soil macropores. These effects were increased in conditions of high water tables and saturated flow. In coarse textured soils P concentrations as high as those in the STE have been observed to depths of 8 m. These soils have both high flow rates and little P adsorbing capacity.

Lateral movement of P has been observed at distances greater than 30 m in coarse textured sediments, and greater than 17 m in soils with high water tables. The continued effect of high water table on P movement suggests that the separation distance between drainage tile and seasonally high water table should be maximized. Sawhney and Hill (1977) working with Connecticut soils suggested that 30 cm did not provide sufficient treatment for P, and suggested a 60 cm separation distance. Soils characterization data was absent from this paper. Cogger et al. (1988) examined P movement in sandy coastal plain soils. They concluded that 30 cm was not sufficient for these soils, and suggested a 60 cm separation distance. The OSWDS in this study used LPD to distribute the STE, suggesting that equal distribution of the wastewater is necessary. These conclusions were confirmed by other studies.

The effectiveness of alternative OSWDS such as sand or peat filters, or mound type ST-SAF on treatment of P depends upon the composition of the filter material. If significant amounts of Fe are present coating sand grains or in association with the finer fraction, significant P reduction will occur. In filter media void of reactive Fe minimal reduction in P will occur. In time, even in systems with considerable Fe, the adsorbing capacity of the filter media will eventually be filled and the degree in which P is attenuated will be reduced. Alternative methods such as P conservation or treatment of P within the ST may be the best approach to reducing the effect of P to the pollution of ground and surface waters.

BACTERIA

Bacteria, viruses, helminths, and protozoa are the primary biological contaminants in STE. Helminths and protozoa are relatively large microbes and are easily filtered by even the coarsest of soils. These microbes are of little concern to the pollution of ground waters from OSWDS because of their relatively large size. Bacteria and viruses however, are much smaller and can move into ground and surface waters. Bacteria and viruses in groundwaters can initiate significant health problems, and promote outbreaks of waterborne disease.

Bacteria are substantially larger than viruses (Figure 2). Under certain soil and environmental conditions, bacteria can move from the ST-SAF, through the soil, and into groundwater. The fate and transport of bacteria applied as wastewater to soils has been reviewed by Bouwer and Chaney (1974), Burge and Marsh (1978), Gerba et al. (1975), and Lance (1978). Reviews specifically related to the fate and transport of bacteria applied from OSWDS have been presented by Bicki et al. (1984), Cogger (1988), Hagedorn et al. (1981), McCoy and Ziebell (1975), Miller and Wolf (1975), and Reneau et al. (1989).

Bacterial pathogens, such as *Salmonella spp.* and *Shigella spp.*, are often difficult to assay and detect in waste and drinking water samples (McCoy and Ziebell, 1975). In order to estimate the potential for pathogenic pollution, indicator organism of the human intestine, such as fecal coliforms and fecal streptococci, are most often assayed in groundwater samples (Bouma et al., 1972). Elevated levels of fecal coliforms indicate that water is contaminated and may be of human risk.

Incomplete treatment of domestic wastewater was responsible for 41% of the outbreaks and 66% of the illnesses linked to contaminated groundwater in the U.S. between 1971 and 1978 (Craun, 1981). Similar reports were made by Craun (1985) for the period of 1971 to 1981. Craun (1981, 1985) summarized the leading etiological agents contributing to waterborne outbreaks and illnesses. The primary bacterial agents were *Shigella, Salmonella, Campylobacter,* and toxigenic *E. coli.* Matthess et al. (1988) also reported these bacterial pathogens as the most important in

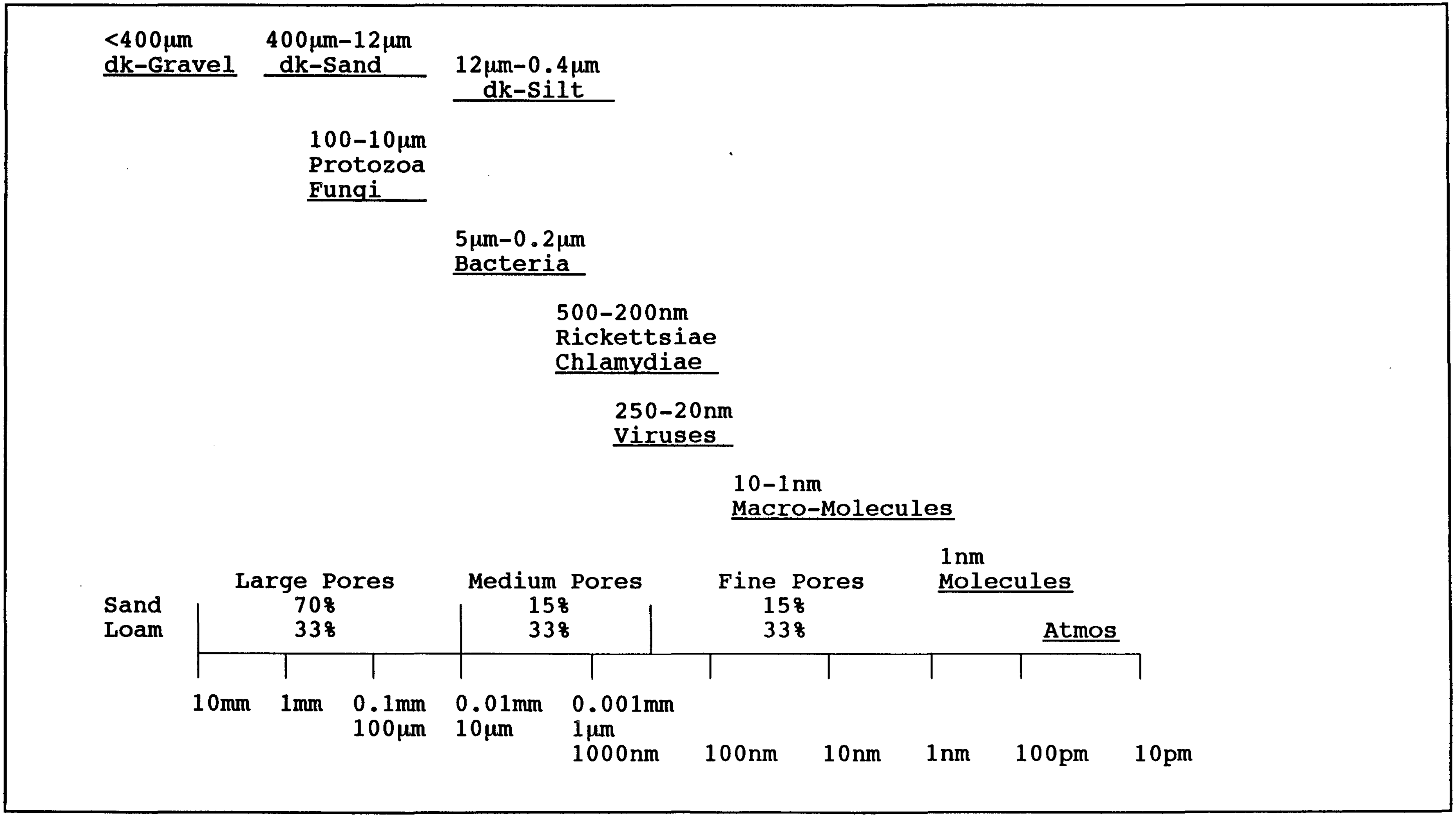

Figure 2. *Comparison of sizes of bacteria, viruses and molecules with hydraulic equivalent diameters of voids (Matthess and Pekdeger, 1981).*

groundwater. In addition, Matthess et al. (1988) included *Vibrio cholerae, Yersina, Leptospira, Francisella tularenis,* and enterotoxic forms of *Pseudomonas, and Legionella pneumophila* as important bacterial pathogens found in groundwater.

Bacterial pollution of groundwater from OSWDS appears fairly widespread. Surveys of bacterial contamination of ground and surface waters have been conducted by various researchers including DeWalle and Schaff (1980), LeChevallier and Siedler (1980), Mancl and Beer (1982), Sandhu et al. (1979), Sauer (1975), Brooks and Cech (1979), and Reneau et al. (1975). DeWalle and Schaff (1980) examined well records and water samples over a 30 year period near Takoma, Washington. The population of the area was 242,000 with 100,000 of the residents on OSWDS. The study area was underlain by glacial deposits. As many as 35% of the wells located in areas served primarily by OSWDS were contaminated with coliforms.

LeChevallier and Seidler (1980) examined drinking water from rural areas in Oregon for fecal coliforms and *Staphylococcus aureus. Staphyloccus aureous,* a common agent of food poisoning, and coliforms were found in 6 and 15% of 320 rural drinking water samples, respectively. No correlation between the presence of coliforms and *Staphylococcus aureus* was found. LeChevallier and Seidler (1980) concluded that coliform analysis may not be a satisfactory measure of drinking water quality.

McGinnis and DeWalle (1983) reported on an outbreak of typhoid fever in Yakima, Washington. The drinking water was contaminated from a ST-SAF located 64 m from the well.

Sandhu et al. (1979) examined levels of total coliforms, fecal *streptococci,* and *E. coli* in water samples from 460 wells in South Carolina. Samples, representing about 10% of the rural population, were collected from areas of the lower coastal and middle coastal plain, and sand hills of the piedmont. Only 7.5% of samples met health standards for total coliforms. Fecal *streptococci* and *E. coli* were found in 75 and 43% of the wells, respectively. Data suggested that as well depth and distance from OSWDS increased, the degree of pollution decreased. These trends however, were not significantly different. Regression analysis of total coliforms/*E. coli* versus distance from the ST-SAF, showed a strong correlation (r=.83) for the lower coastal plain sampling sites. The correlation was only .29 for the middle coastal plain data. Sandhu et al. (1979) concluded that the bacterial pollution to the wells was attributable to OSWDS.

Reneau et al. (1975) examined levels of bacteria in ground and surface waters of a small (80 ha) watershed in Virginia. Soils in the watershed were divided into three groups based on their suitability for ST-SAF. Seventeen percent of the soils were suitable, 41% marginal, and 42% unsuitable for ST-SAF. The ST-SAF constructed in marginal soils failed during periods of high precipitation. All of the ST-SAF located on unsuitable soils failed. Water samples obtained from both ground and surface waters near failing ST-SAF showed high numbers of total and fecal coliforms. Coliform concentrations decreased with distance from the ST-SAF. Reneau et al. (1975) concluded the decreases in coliforms with distance from the ST-SAF were related to dilution effects.

Mancl and Beer (1982) examined concentrations of bacteria in 68 wells within a 58 acre area of Iowa. Homes within the study area used ST-SAF for wastewater treatment. Soils had developed in sandy loam or loamy sand terrace deposits. Only one well was contaminated with fecal coliforms, although 53% of the water samples contained > 10 mg L^{-1} -N

Brooks and Cech (1979) investigated $NO_3^-$and bacteria levels in 53 rural water supplies in Texas. The study area was composed of sandy quaternary and tertiary coastal plain deposits. Nearly all of the 21 wells within 15 m of the surface tested positive for fecal coliforms and *streptococci*. Fecal bacteria were also found in some of the deeper wells (up to 80 m deep). All wells with elevated NO_3^- levels tested positive for fecal bacteria. Brooks and Cech (1979) concluded that domestic wastewater from OSWDS was the primary source of pollution for these wells.

Sayler et al. (1975) examined levels of fecal indicator organisms in the upper Chesapeake Bay. Five stations were sampled from the conflux of the Susquahanna River and Chesapeake Bay, to Annapolis. All stations were polluted with fecal organisms.

Total concentrations of bacteria in STE range from 25 to 48 x 10^8 L^{-1} (McCoy and Ziebell, 1975). Average concentrations of fecal coliforms and *streptococci* in STE are 4.2 x 10^7 and 3.8 x 10^4 L^{-1}, respectively (McCoy and Ziebell, 1975; Ziebell et al., 1975). Reduction in these bacterial concentrations following addition of STE to a ST-SAF is dependent upon flow conditions, survival rates of microbes, temperature, competitive microbial population, degree of development of clogging mat, as well as soil moisture content, particle size distribution, pH, and degree of soil structure. These factors combine to both physically remove through filtration and adsorption, and inactivate the bacteria.

Rates of bacterial inactivation or natural die off are an important consideration in groundwater pollution from OSWDS. If the pathogenic bacteria cannot survive long enough to travel from the ST-SAF to the groundwater, or from the point of entry into the groundwater to the well, these bacteria will not be a viable problem. Rates of bacterial inactivation vary between bacteria and soils. These rates can be affected by several factors including both temperature and soil moisture content. Bitton et al. (1983) studied survival of *Salmonella typhimurium*, fecal *streptocci*, and *E. coli* in water obtained from a Florida well. The well was 145 m deep. Water temperature was 22°C, with a pH of 7.6. Bacterial survival was investigated over a 15 day period. *Salmonella* and *E. coli* showed similar die off rates, with survival constants of $k=-0.0054$ and -0.0066, respectively. Fecal *streptococcus* showed longer survival rates ($k=-0.0012$). Similar findings were reported by Chandler and Craven (1980) for *Salmonella typhimurium* and *E. coli*. In a second study, Bitton et al. (1983) examined survival rates of fecal coliforms and fecal *streptococci* found in well water 50 m from SSE discharge into a cyprus strand. Bacteria were detectable 70 days after discontinuation of SSE discharge, suggesting that bacteria are relatively stable in groundwater.

McFetters et al. (1974) compared survival rates of indicator bacteria and enteric pathogens in well water over 3 and 4 day periods. Coliform death rates were greater, and showed more variability than enterococcus. Results showed that survival times of *Aeromonas sp.* > *Shigellae* > *Fecal streptococci* > *Coliforms* = *Salmonella* > *Streptococcus equinus* > *Vibrio cholera* > *Salmonella typhi*.

Hagedorn et al. (1978) introduced antibiotic resistant bacteria into soil at depths of 30 and 60 cm. Soils were somewhat poorly drained, with clay contents ranging from 27 to 45%. Both *E. coli* and *S. faecalis* were found in soil after 32 days.

Parker and Mee (1982) examined survival of *Salmonella adelaide* and fecal coliforms in two coarse sands amended with STE. *Salmonella adelaide* and fecal coliforms showed similar survival rates for one soil but not another. Average survival of > 10% of fecal coliforms was 64 days, with 46 days for equivalent survival of *Salmonella.*

Tate (1978) studied survival of *E. coli* over an 8 day period in a muck and fine sand soil. Greater than 3 times more *E. coli* survived in the muck soil than in a fine sand. Initial bacterial population was shown to affect survival. The smaller the initial bacterial population, the greater the number of bacteria that survived after 8 days.

Several studies have shown relationships between bacterial survival and temperature, or soil moisture content. Kibbey et al. (1978) examined survival of fecal *streptococci* under various soil moisture contents and temperatures. Five soils, collected from A horizons, were used in the study. Soil moisture contents ranged from saturation to air-dried, while temperatures ranged from 4 to 37° C. Bacteria survived longer under cooler moister conditions regardless of the soil type. Longer survival under cooler conditions was related to the lower activity of the other competitive soil organisms. On average, a 95% reduction in bacteria occurred within 53 days for 4 of the soils. In the other soil, 5% of bacteria were still alive after 120 days. A series of freeze-thaw treatments substantially reduced bacterial numbers.

Parker and Mee (1982) found that survival for fecal coliforms and *Salmonella adelaide* was less at saturated conditions than at 5% moisture content. These differences however, were not significant at the .05 level. Parker and Mee (1982) concluded that survival times for bacteria could not be predicted by soil moisture content.

Chandler and Craven (1980) examined relationships between soil moisture and survival of *E. coli* and *Salmonella typhimurium* in soils. Soil moisture contents of 40% were optimum for survival of these 2 bacteria. At this moisture content, 10% of bacteria lived longer than 17 days.

Other studies have shown increased survival of bacteria at lower temperatures. Stenstrom and Hoffner (1982) observed that survival rates for *E. coli* and *Streptococcus faecalis* are much greater at 4°C than 20°C. Survival was greater for *S. faecalis* at 4°C than for *E. coli.* Geldreich et al. (1968) found that bacteria in storm water survived longer at 10°C than at 20°C.

Movement of bacteria through soil is dependent upon size of the bacteria, soil pore size and distribution, and the interactions between soil and bacteria (Bitton et al., 1983). If bacteria are larger than the soil pores, filtration will occur. In soils with pores larger than bacteria, adsorption is the primary mechanism for retention of bacteria. Factors that affect pore sizes and bacteria adsorption include bacteria type and strain, flow conditions, soil pH, moisture content, particle size distribution, degree of soil structure, and nature and concentration of electrolytes (Bitton et al., 1983; Strenstrom and Hoffman, 1982; McCoy and Ziebell, 1975). Bitton et al. (1974) examined movement of two strains of *Klebsiella Aerogenes* in 2 cm diameter, 11.5 cm long soil columns. Columns were filled with four soils ranging in particle size distributions from 90% sand to 58% clay. Different bacteria strains showed different degrees of retention. Sandy soils showed less retention of bacteria than clayey soils. The highest degree of bacterial retention occurred at lower pH values. Soil water content was positively correlated with depth in which the bacteria moved. Bitton et al. (1974) concluded that both adsorption and filtering of bacteria were occurring.

Tare and Bokil (1982) examined the effect of various particle size distributions on removal of bacteria in columns from 7.5 to 75 cm in length. Sand, and silt and clay particles were mixed at various ratios from 0 to 100% sand. Bacterial removal was greatest in mixtures with higher percentages of clay and silt particles. Tare and Bokil (1982) concluded that a mixture of 40% < 75 u and 60% > 75 u soil particles was the most efficient mixture for attenuation of bacteria.

Smith et al. (1985) examined movement of bacteria through 28 cm disturbed and undisturbed columns of soil. Disturbed columns removed ≥ 93% of bacteria, while undisturbed cores removed only 21-78%. Increases in flow rates increased the number of bacteria moving through the columns. Increasing the loadings of bacteria did not change the ratio of the concentration of bacteria in the column effluent to initial bacterial concentration ($C:C_0$). These data suggested that bacteria sorption sites were not filled to capacity. Smith et al. (1985) concluded that movement of bacteria and water in the undisturbed columns was occurring along macropores and by-passing treatment.

Strenstrom and Hoffner (1982) suggested that bacteria size is not as important as bacteria surface characteristics in explaining reductions in bacteria a soil. Using a sand soil as an example, Strenstrom and Hoffner (1982) pointed out that most of the soil pores were larger than bacteria (Figure 2). Therefore, in soils such as these, bacteria will not be effectively filtered, and adsorption is the primary mechanism for bacterial attenuation. Adsorption can occur as bacteria actively attach to soils using extracellular polymers or fibria, or as a result of electrical charges. Strenstrom and Hoffner (1982) examined the surface charge and attenuation of 3 strains each of four bacteria to soils. Although bacteria have both positive and negative surface charges, most strains were highly negative. The degree of positive and negative charge varied between strains and types of bacteria.

Field studies

Movement of bacteria through well drained, fine and medium textured soils is minimal due to both adsorption and filtration. Concerns of groundwater pollution from bacteria applied to OSWDS in such soils is unfounded. In soils with coarse textures, considerable structure, or high water tables, movement of bacteria may be significant. Brown et al. (1978) examined the movement and distribution of bacteria below 3 ST-SAF established in soils with textures of sandy clay, clay, and sandy loam. Soil solutions were collected in ceramic cups 120 cm below drain lines. During the second year of the study, only 3 of the 133 water samples collected were positive for fecal coliforms. Soil analysis indicated high levels of fecal coliforms along channels to depths of 90 cm below drain lines (soil samples were only collected to 90 cm). Similar concentrations of fecal coliforms with depth were not observed in soils without channels.

Bouwer et al. (1976) examined movement of fecal coliforms in an RI system (Flushing Meadows). Most fecal coliform were attenuated in upper 60 cm. Movement of bacteria was observed especially after a drying period. Drying removed clogging material and reduced natural microbial populations. The clogging material acted as a filter and microbial population were competitive with the fecal coliforms. Most bacteria were removed within 91 meters of the point of application of the SSE.

Gilbert et al. (1976) examined the removal of bacteria in an RI System (Flushing Meadows). After percolation through > 9 m of sandy soil 99.9% of bacteria were removed.

Schaub and Sorber (1977) examined the efficiency of an RI System in removing bacteria. Most bacteria were removed in the coarse textured sediments. Fecal *streptococci* however, were found in the groundwater 29 m below the soil surface.

Peterson and Ward (1988, 1989) presented results from simulation models used to predict bacterial movement in coarse soils. Results suggested that in unsaturated coarse textured soils, bacteria may move more than 1.2 meters from the point of application. Therefore, in coarse textured soils water table depth should be greater than 1.2 m below drain lines.

Ver Hey and Woessner (1988) examined movement of bacteria from ST-SAF placed in coarse-textured alluvial soils in Montana. Bacteria were found in samples collected just above the water table at depths of 2.4 and 4.3 m below the surface.

Most of the field research on bacteria applied from OSWDS has focused on the effect of saturated soil conditions on bacterial movement. During saturated conditions water flows primarily through the larger soil pores and channels. If these pores are larger than the bacteria, and conditions for adsorption are less than ideal, significant movement of bacteria may occur. Cogger and Carlile (1984) examined movement of bacteria away from 15 ST-SAF in the coastal plain of North Carolina. Wells were placed at distances of 1.5 and 7.5 m from the OSWDS. Continuously saturated systems showed bacterial concentrations significantly higher than systems with lower water tables. Substantial decreases in bacterial numbers were observed between samples collected at 1.5 and 7.5 m from the ST-SAF. Highest levels of bacteria were observed when groundwater gradients were highest. Higher gradients occurred during saturated conditions, or in summer months when groundwater mounds formed under the ST-SAF. Under saturated conditions, systems using LPD showed better removal of bacteria than conventional systems.

Cogger et al. (1988) examined movement of bacteria in a Quartzipsamment in North Carolina. Two ST-SAF within the same area were examined in which the water table depth differed by 30 cm. Water tables were within 30 cm of the trench bottom 75% of the study period in the poorer drained soils. Wells were installed at distances of 0.3 and 3 m from each ST-SAF trench. Considerably more bacteria were found in wells adjacent to the ST-SAF with higher water tables, than wells located near the better drained system. Bacteria numbers decreased with distance from the ST-SAF in both systems. Higher loading rates showed higher numbers of bacteria in both sites. Cogger et al. (1988) concluded that a 30 cm separation distance between the bottom of the adsorption trench and water table was inadequate for removal of bacteria. A 60 cm separation distance would be sufficient for removal of bacteria in these soils, if the effluent was applied using LPD, and a low (< 4 cm/day) loading rate was maintained.

Hagedorn et al. (1978) examined the movement of fecal bacteria under saturated flow. Bacteria were introduced into soils at 30 and 60 cm depths in a simulated ST-SAF. Water samples were collected from wells placed at set distances down-gradient from the point of application. Soils were somewhat poorly drained on a 2% slope. Clay contents increased with depth, and ranged from 27 to 45%. Within 24 hours of each application, bacteria reached wells 3 m from the simulated ST-SAF, and in some cases reached the wells at 5 m. The greatest numbers of bacteria were observed following rainfall. In the soil, bacteria were observed within 50 cm of point of application in all directions. At greater distances however, bacteria were only recorded down gradient. Bacteria were detected in wells 15 m from the ST-SAF, but no bacteria were detected within 32 days of the well 30 m away. Hagedorn et al. (1978) suggested that a longer sampling period may have shown positive results for the 30 m well. Bacteria introduced at the lower depth took longer to reach wells than those applied into the upper soil. These results were related to the heavier textured, lower B horizon. Hagedorn et al (1978) concluded that antibiotic resistant bacteria were an excellent bacteria tracer because these bacteria are not naturally found in the soil, and therefore could be detected from the natural bacteria.

McGinnis and DeWalle (1983) reported on the movement of typhoid organisms in saturated soils leading to an outbreak of typhoid fever in Yakima, Washington. The drinking water was contaminated from a ST-SAF located 64 m from the groundwater well. The ST-SAF was located in terrace deposits with a water table within 2.1 m of soil surface. Dye added to the septic tank reach the groundwater well within 36 hours.

Rahe et al. (1978) investigated the movement of *E. coli* through saturated soil. Two sites at foot slope positions, and sloping at 14 and 20%, were examined. Saturated soil conditions were maintained by water application. One hour after addition of bacteria to the A horizon of one site, detectable numbers of bacteria were recorded at a depth of 12 cm, and a distance of 15 m from the point of application. After 2 hours, detectable numbers of bacteria were observed at a 100 cm depth. Addition of bacteria into the B horizon and saprolite showed similar results. Both A and B horizons had silty clay loam textures. In the second site, bacteria moved 20 m in 12 hours. Bacteria moved faster in the A horizon (silt loam texture), than B and C horizons (silty clay loam and clay textures). Transport in the first site was much faster than in the second. Rahe et al. (1978) concluded that in the saturated soil, water and bacteria were moving through soil macropores where very little filtering of bacteria will occur. Site 1 had many more macropores and better soil structure than site 2, therefore bacteria moved faster through site 1.

Reneau (1978) examined the movement of bacteria away from several ST-SAF placed in Aeric and Typic Ochraquults in the Virginia coastal plain. The study occurred over a three year period. Wells were placed at distances of 1.5, 4.6 and 10.4 m from the ST-SAF in the direction of the groundwater gradient. The gradient was maintained by a tile drain. Bacteria concentrations decreased logarithmically away from the ST-SAF. Bacteria moved farther and in greater numbers during periods of high water tables, than under low water tables. Reneau (1978) concluded that during high water tables bacteria were moving with minimal attenuation under saturated flow conditions.

Reneau and Pettry (1975) examined movement of bacteria away from 3 ST-SAF placed in soils classified as Plinthic Paleudults (Varina series), Aquic Paleudults (Goldsboro series) and Typic Fragiudults (Beltsville series). For Varina soils, the number of fecal coliforms decreased with distance from the ST-SAF. Positive identification of coliforms was observed as far as 6.1 m from the ST-SAF. No coliforms were found at any distance below the plinthic horizon. Bacteria were positively identified in water samples collected from Goldsboro and Beltsville soils at distances of 13.5 and 28 m from the ST-SAF, respectively.

Stewart and Reneau (1981) examined movement of fecal coliforms away from 2 ST-SAF in the coastal plain of Virginia. Both sites were located on soils with high water tables and classified as Typic Ochraquults. Wells were installed to depths of 152 and 305 cm at set distances from the ST-SAF to collect water samples. In one system a groundwater gradient of 2-3.2% was maintained by a drainage tile. In this system bacteria were observed as far as 10 m from the ST-SAF. Bacteria at the 305 cm sampling depth directly below the ST-SAF were < 30 MPN L^{-1}. Higher levels of bacteria (< 50 MPN L^{-1}) were found in the 152 cm deep well 10 m away from the ST-SAF. These results suggested that most bacteria were moving laterally from the ST-SAF. Bulk density values of the subsurface horizons averaged over 1.7 g/cm^3, which would restrict vertical movement. The groundwater gradient at the second site was minimal, and vertical movement of bacteria was of greater importance. During periods of high water tables, increases in bacteria numbers were observed in the wells of both sites.

Stewart and Reneau (1988) examined bacteria movement in soils treated with STE. Soils had high water tables and STE was applied < 30 cm below the soil surface using LPD. Soils were poorly-drained fine loamy, siliceous, Typic Ochraquults. Wells were placed at distances of 2.8 and 8.4 m from the ST-SAF. All wells showed ≤ 0.3 MPN L^{-1}. These low numbers of bacteria were attributed to the equal distribution of STE by the LPD system as well as increased treatment in the shallower, biologically active, near surface horizons.

Viraraghavan (1978) examined the movement of microbes away from a ST-SAF placed in poorly drained soils with sandy clay and clay textures. Water samples were collected at several distances from the ST-SAF along the groundwater flow path. Numbers of bacteria decreased with distance from the ST-SAF, but high levels of bacteria were still observed at the last sampling point 15 m from the ST-SAF.

Wilson et al. (1982) examined bacterial concentration in drains placed 3 and 6 meters from ST-SAF to artificially lower the water tables. In cases where water tables remained high, levels of bacteria were well above acceptable levels.

Fate of bacteria in alternative systems

Most of the research addressing the fate of bacteria in alternative systems has focused on the usage of mound systems or some form of a filter bed for bacterial attenuation and reduction. The effectiveness of alternative systems treatment for bacteria varies between the system and the research.

The effectiveness of mound systems has been investigated by Magdoff et al. (1974), Stewart et al. (1979), Willman et al. (1981), and Ziebell et al. (1975). Ziebell et al. (1975) examined the efficiency of a mound system in removing bacteria. Although minimal numbers of fecal coliforms passed through the mounds, detection of *Ps. aeruginosa* suggested the presence of pathogenic bacteria. Ziebell et al. (1975) concluded that filtration of bacteria from the soil below the mound would be necessary before treatment for bacteria would be adequate.

Magdoff et al. (1974) examined movement of bacteria in columns designed to simulate mound-type OSWDS. Columns were monitored before and after a clogging mat had formed at the interface between gravel and sand. All fecal coliforms and *streptococci* were attenuated in the simulated mound regardless of the presence or absence of the clogging mat. Distribution of bacteria within the column was not determined at the end of the experiment. At the base of the simulated mound, 30 cm of silt loam soil was placed in the column to represent the original soil surface. This fine textured material may have attenuated any fecal organisms that passed through the sandy mound fill material.

Stewart et al. (1979) examined the efficiency of mounds in removing bacteria from STE. Columns were constructed to simulate a mound system. Fecal bacteria were not observed below 30 cm. Columns constructed with finer textured soils showed no bacteria below 15 cm.

Willman et al. (1981) examined various sand and clay mixtures for renovation of STE. These mixtures were packed in columns to simulate mound systems. Regardless of the mixture, > 99% of the bacteria were removed.

Filter beds are a commonly used alternative system for pretreatment of domestic wastewater or in some cases complete treatment of STE. Filter beds are generally composed of sand, peat, or a sand-peat mixture. Kristiansen (1981c) investigated the distribution of bacteria within a sand-filter trench following application of STE. Numbers of bacteria decreased with depth with the highest number occurring at the gravel-sand interface of the sand filter. The system with the most developed clogging mat showed the least number of bacteria with depth. Larger bacteria were concentrated closest to the drain with smaller bacteria showing some movement. Bacterial biomass in the most clogged system was strongly correlated with carbon content (r = 0.999). This was not the case for the least clogged system (r = 0.11). Kristiansen (1981c) concluded that clogging was important to the purification process in OSWDS, and mechanisms that reduce the clogging mat also reduce the efficiency of the system. Similar conclusions were offered by Bouwer et al. (1974) for an RI system.

Sauer and Boyle (1978) examined the efficiency of several sand filters in treating STE and effluent from an aerobic unit. Bacterial numbers in effluent from the sand filters were too high to allow for surface discharge and disinfection would first be necessary.

Ziebell et al. (1975) examined the efficiency of several alternative and conventional OSWDS in removing bacteria. Sand filters showed more bacteria in effluent than mounds or conventional ST-SAF.

Effert et al. (1985) investigated the efficiency of sand filters, up-flow gravel filters, and subsurface gravel beds in removal of bacteria applied as STE. Relatively high numbers of bacteria occurred in filter effluent of each system. Effert et al. (1985) concluded that disinfection is necessary for these systems to provide adequate treatment for pathogens.

Brooks et al. (1984) examined the effectiveness of a peat filter for removing bacteria in STE. Greater than 99% of bacteria applied were attenuated in the 75 cm of peat. Brooks et al. (1984) suggested that peat may be a better material for filter beds than sand because of the greater efficiency of bacterial attenuation.

Rock et al. (1982, 1984) investigated the efficiency of peat filter beds for the removal of bacteria. In both field and column studies, total coliforms were reduced by 99.9%. Rock et al. (1984) concluded that peat provided excellent treatment for bacteria so that disinfection of effluent may not be necessary.

Nichols and Boelter (1982) studied bacterial removal from SSE from a campground for 8 years. Effluent was sprayed on a peat-sand filter bed. Almost all (>99%) of bacteria were removed by the filter.

Osborne (1975) applied SSE to a sand-peat filter using spray irrigation. The filter removed > 99% of fecal coliforms. Greater than 75% of the time no bacteria were detected in the filter bed effluent.

Several other alternative methods have been used to remove bacteria from domestic wastewater. Brandes (1977) studied the effect of alum on the population of total and fecal coliforms in a septic tank. Alum reduced total coliforms by 35% and fecal coliforms by 80%. Brandes et al. (1975) examined the effectiveness of columns or filter beds constructed of sand, or sand mixed with Al mill waste (red mud), silty clay, or limestone. The finer the particles in the columns or filter beds, the more bacteria were removed. Ziebell et al. (1975) investigated the effectiveness of an aeration unit on reducing bacteria prior to application to a soil. Ziebell et al. (1975) noted that aeration did not reduce bacteria sufficiently, and further purification would be required prior to discharge into ground or surface waters.

Some alternative systems, such as composting toilets, separate domestic black water from grey water. Siegrist (1978) enumerated the bacterial populations in domestic grey water. Both bath and laundry waters contained significant numbers of enteric organisms. Indicator bacteria in grey water STE were very high after a year of operation suggesting the need for additional treatment before disposal of greywater STE into ground or surface waters.

Summary and conclusions

Contamination of ground and surface waters from bacteria is one of the primary concerns with the use of OSWDS. Under certain soil and environmental conditions, bacteria can move away from ST-SAF and into nearby ground and surface waters. Bacteria in groundwaters can initiate significant health problems, and promote outbreaks of waterborne disease. The primary bacterial agents leading to waterborne outbreaks and illnesses are *Shigella, Salmonella, Campylobacter, Vibrio cholerae, Yersina, Leptospira, Francisella tularenis, as well as* toxigenic *E. coli, Pseudomonas,* and *Legionella pneumophila.*

Bacterial pollution of groundwater from OSWDS appears fairly widespread. Surveys of bacterial contamination of ground and surface waters have been conducted in numerous states including Washington, Oregon, Iowa, Maryland, Virginia, Texas, and South Carolina. These studies have shown that shallow aquifers, in areas where OSWDS is the primary means of treating domestic wastewater, have considerable bacterial pollution.

Average concentrations of fecal coliforms and *streptococci* in STE are 4.2×10^7 and 3.8×10^4 L^{-1}, respectively. The reduction in the concentration of these bacteria is primarily dependent upon the processes of inactivation (natural die-off), filtration, and adsorption occurring in the ST-SAF. Rates of bacterial inactivation are an important consideration in groundwater pollution from OSWDS. If pathogenic bacteria can survive long enough to travel from a ST-SAF to a groundwater well, a serious health problem may result. Rates of bacterial inactivation vary between bacteria, groundwaters, and soils. Inactivation rates are affected by both temperature and soil moisture content. In general, cooler moister soils show longer survival of bacteria. Bacteria were shown to survive in groundwater at least 70 days (Bitton et al., 1983). In soil, bacteria were reported to survive greater than 120 days (Kibbey et al., 1978).

Attenuation of bacteria in the soil is primarily a function of the processes of adsorption and filtration. Factors that affect adsorption and filtration include bacteria type and strain, flow conditions, soil pH, moisture content, particle size distribution, degree of soil structure, and nature and concentration of electrolytes. Soils with fine textures, minimal structure, and low pH values adsorb and filter nearly all bacteria. Therefore, movement of bacteria through well drained, fine and medium textured soils is minimal. Concerns of groundwater pollution from bacteria applied to OSWDS in such soils is unfounded. Numerous studies however, have shown that in soils with coarse textures, considerable structure, or high water tables, (conditions associated with high flow rates) movement of bacteria is significant resulting in groundwater pollution. The distances and rates at which bacteria move varies with the OSWDS and associated soil conditions. Movement of fecal streptococci to depths of 29 m has been reported in coarse textured RI systems (Schaub and Sorber, 1977). Under saturated conditions, Rahe et al. (1978) reported bacteria moving 15 m in one hour within an A horizon.

The most studied types of alternative systems used to treat STE for bacteria were mounds and filter beds. Mound systems were effective in removing bacteria applied as STE. These systems were especially effective if soil was available below the mound for additional treatment. Most sand filters were not very effective, and some additional treatment prior to introduction of effluent into ground or surface waters would be necessary. Sand filter and RI systems with well developed clogging mats showed much better treatment of bacteria than those without these organic concentrations. Peat filters, or filters combining both sand and peat showed significant reductions in bacteria. These systems were much more effective than the sand filters. The use of LPD in soils with high water tables appears to be more effective than conventional systems under similar environmental conditions.

VIRUSES

Viruses are very small microbes (< 250 nm) that behave in the soil and soil solution as a colloid. The largest virus is smaller than any bacteria (Figure 2). Because of their small size, filtration by the soil is minimal and adsorption is the primary process for removal of viruses applied in OSWDS. Under conditions unfavorable for adsorption, viruses can move away from an OSWDS and into nearby groundwater. Viruses in groundwater can initiate significant waterborne disease.

There are over 100 known enteric viruses excreted in the feces of man (Gerba, 1984). The fate and transport of these viruses applied in wastewater has been reviewed by Bouwer and Chaney (1974), Gerba (1984), Gerba et al. (1975), Keswick and Gerba (1980), and Berg (1973). Reviews specifically addressing viruses applied from OSWDS have been offered by Sproul (1975), Reneau et al. (1989), Bicki et al. (1984), Hagedorn et al. (1981), and Miller and Wolf (1975).

Pathogenic viruses are primarily Hepatitis A, Norwalk agent (rotavirus), Poliomyelitis, Coxsackie, Rota, and Adeno (Matthess et al., 1988; Gerba, 1984; Craun, 1985; Craun, 1981; Craun, 1979; Safferman, 1982). Craun (1985) reported that 9% of the outbreaks of disease reported in the U.S. between 1971 and 1982 were attributable to Hepatitis A (5%) and Norwalk agent rotavirus (4%). Craun (1981) reported an outbreak of viral hepatitis in Arkansas in 1971. The immediate source of Hepatitis virus was a contaminated well. The well was contaminated from a ST-SAF which was shallow to rock strata allowing lateral movement of the virus. The residents of the home with the malfunctioning ST-SAF had infectious hepatitis six weeks prior to the outbreak.

An outbreak of hepatitis was reported by Craun (1974) to have occurred in 1973 in a small Alabama community. The immediate source of the hepatitis virus was two springs which served as the community water supply. The springs were contaminated from an improperly designed OSWDS 61 m from the springs.

Viruses may be more of a pollution problem in groundwater then is presently known. Virus assessment in groundwater is difficult because of the large number of viruses excreted by man. Making assays of all of these is not possible. These assay problems are magnified when the difficulties in concentrating viruses from groundwater into a sample small enough to assay are considered (Wellings et al., 1974). In addition, viruses tend to clump and are therefore not well distributed in the groundwater (Wellings et al., 1975b). Another perspective problem is the differences between viruses prepared in the lab versus the respective natural virus. These differences could be especially important in survival rates (Wellings et al., 1974).

Other perspective problems in determining the extent of pollution to the groundwater from viruses may also be the sampling techniques. Wang et al. (1980) examined several ceramic soil-water samplers for virological sampling. Samplers with pore sizes > 20 u recovered 82 to 100% of the viruses introduced. Samplers with pores smaller than 20 u however, trapped considerable numbers of viruses. Powelson et al. (1990) reported that stainless steel solution samplers showed no attenuation of viruses, but ceramic cup samplers showed a 74% loss in viruses attributed to adsorption. Assays made from ceramic samplers with < 20 u pores would suggest much less virus activity than actually occurs.

Wellings et al. (1976) present data indicating that many viruses in wastewater are attached to solids. These solids may be suspended in water samples. During concentration techniques, viruses attached to suspended particles may be filtered, discarded, and therefore left undetected. These assay methods would indicate much less virus pollution than actually exists. Similar conclusions were reached by Gerba et al. (1975) and Gerba (1984).

Virus survival

Survival of viruses is an important consideration in addressing the possibility of ground and surface water pollution from OSWDS. If viruses survive long enough to pass from the ST-SAF into the groundwater, serious health problems may result. Hurst et al. (1980a) examined various soil and environmental factors effecting survival of viruses in soils. Survival of 5 virus and 2 bacteriophages were examined in 9 soils. Soil characteristic and properties were well documented. Temperature, soil moisture content, and O_2 conditions were varied during the studies. Temperature was the most important factor in explaining survival regardless of other properties. Lower temperatures showed higher survival rates.

Yates et al. (1985) collected groundwater samples from 11 wells located throughout the U.S. Samples were maintained at the temperature they were collected, as well as 2 other temperatures. Polio, Echo, and MS-2 viruses were added to the samples and survival rates documented. Samples were analyzed for NO_3^-, NH_4^+,

SO_4^-, Fe, Ca, Mg, TDS, and pH, and multiple regression techniques were applied to data. Over 77% of the variability in survival rates could be explained by the temperature. Viruses maintained at lower temperatures survived longer. Survival of polio, MS-2, and Echo viruses were similar.

Several studies have examined the relationships between virus survival and degree of adsorption, soil type, virus type, microbe competition, and soil moisture content. Hurst et al. (1980) reported that adsorption was second only to temperature, as the factor most important in explaining virus survival. Increases in adsorption showed increases in virus survival. Step-wise multiple regression techniques indicated that levels of resin-extractable P (labile-P), exchangeable Al, and pH were found to positively correlate with virus survival. These soil factors also affect virus adsorption supporting the positive correlation between virus survival and adsorption. The presence of SSE did not affect survival rates. Presence of aerobic microbes reduced survival rates of viruses but not significantly at the 0.1 level. A correlation between moisture content and survival was not observed.

Murray and Laband (1979) examined degradation of poliovirus by adsorption to several inorganic compounds. Poliovirus was adsorbed to SiO_2, Fe_2O_3, Al_2O_3, Mn_2O and CuO. Inactivation of polioviruses was monitored following desorption. No inactivation, and only minor amounts were observed following desorption of viruses from SiO_2 and Fe_2O_3, respectively. Significant amounts of inactivation were observed following desorption of MnO_2, Al_2O_3 and CuO. Murray and Laband (1979) suggested that compounds with high Van der Waals forces may inactivate viruses.

Yeager and O'Brien (1979) examined inactivation of viruses following adsorption to soil. Poliovirus type 1 were applied to dry and moist sand and sandy loam soils. Inactivation of viruses occurred in both dry and moist soils. In dry soils however, viruses could be eluted that were not inactivated. Yeager and O'Brien (1979) suggested that at least two separate mechanisms may be involved in virus inactivation depending upon soil moisture conditions.

Hurst et al. (1980b) studied survival of viruses in an RI System (Flushing Meadows). Viruses and sand were placed in a tube and the tube was buried vertically in the RI System. A two day resting (drying) period accounted for inactivation of 83% of the viruses at a 2.5 cm depth and 50% inactivation at depths from 2.5 to 20 cm. Virus survival was greater at 60 cm depths than shallower depths. Hurst et al. (1980b) concluded that differences in virus survival in this system was related to lower aerobic conditions, lower levels of aerobic microbes, and lower degree of drying at the 60 cm depth.

Sobsey et al. (1980) examined survival of polio and reo viruses following application of domestic sewage wastewater to soils. Sterile soils showed greater survival times than unsterile soil, suggesting microbial activity affected survival rates. Average survival of reovirus was 35 and 123 days and poliovirus was 42 and 95 for unsterile and sterile soil, respectively.

Jorgenson and Lund (1985) examined survival of viruses in sludge and soils. Experiments were conducted at temperatures between 4 and 7°C. Viruses were detected in sludge up to 21 weeks from inoculation. A one log reduction was observed in population of viruses in a sand soil within 8 weeks, but a similar reduction in virus population was not recorded for a sandy loam soil until 20 weeks had passed. Viruses were still active after 34 weeks.

Bitton et al. (1983) examined survival of f2 coliphage and poliovirus over a 15 day period. Viruses were kept in groundwater taken from a 475 foot well in Florida and maintained at 22°C. Poliovirus showed similar slow die-off rates as bacteria and f2 died within 5 days.

Adsorption of viruses

Most researchers agree that the attenuation of viruses in soil is primarily restricted to the process of adsorption (Sproul, 1975; Gerba, 1984; Gerba et al., 1975; Bouwer and Chaney, 1974; Powelson et al., 1990). Gerba (1984) offered an excellent review of the aspects of virus adsorption to soils. Because of their small size and variable charge, viruses behave in soil solution similar to a colloidal particle (Gerba, 1984; Gerba et al., 1975). In general, the factors that affect a colloid in soil will also affect a virus in a similar manner. These factors follow the attractive or repulsive forces associated with the diffuse double layer (DDL), and the attractive forces associated with van der Waals forces (Gerba, 1984; Moore et al., 1981). Therefore, adsorption of viruses to soil is primarily a physical process (Preston and Farrah, 1988; Gerba et al, 1975, Gerba, 1984; Moore et al., 1981). Adsorption of viruses to soils, and the factors that affect this process, have been extensively examined in both batch and column studies.

Batch studies

The variable charge of the viruses originates from the protein coat (Sproul, 1975; Gerba, 1984; Gerba et al., 1975). The viral coat is composed of proteins that contain amino acids. These amino acids have both acid and basic groups resulting in an electrical charge. The charge varies depending upon the pH at which the various amino acids dissociate. The pH at which the charge of the virus is neutral, is termed the isoelectric point (IEP). Below this point viruses are positively charged, and above the IEP a negative charge is exhibited.

The morphology of the protein coat and associated IEP may differ between viruses as well as strains. Gerba et al. (1981) used several statistical techniques to assess adsorption of viruses to soils. Ten viruses, and 5 bacteriophages of E. coli were used to study adsorption to 9 soils. Two groups of viruses were easily separated using factor analysis. Group I showed an average 44% adsorption to soil. Group II on average showed 78% adsorption. Bacteriophages occurred in both groups, and different coxsackie strains appeared in both groups. Regression analysis of Group I showed significant negative correlation (r^2=.83) for pH, and organic matter, and a positive correlation for exchangeable Fe (alpha = 0.1). Regression analysis for Group II viruses showed no significant correlations with any soil property. Gerba et al. (1981) concluded that differences in adsorption between the groups were related to the configuration of the outer-capside proteins.

Goyal and Gerba (1979) compared adsorption of 28 viruses and 5 bacteriophages to 9 soils. Soils ranged in clay contents from 3 to 54%. Strains of Echo virus ranged in adsorption from 0 to 97%. Strains of coxsackie ranged from 0 to 30% adsorption. No significant differences in adsorption were observed for strains of poliovirus. These data suggested that virus strain is as important as type in explaining adsorptive properties of viruses.

Numerous studies have investigated the effect of pH, and cation concentration and valence, on adsorption of viruses to soils. Gerba and Lance (1978) examined adsorption of polioviruses to a loamy sand. An increase in adsorption was observed with additions of 0.01 M $CaCl_2$, but not 0.001 M $CaCl_2$. Most viruses showed a significant positive correlation with pH (at the .05 level). All of the viruses showed the most adsorption to two soils with pH values < 5.

Sobsey et al. (1980) examined adsorption of polio and reo viruses to soils. Virus adsorption was shown to occur within 15 minutes. Lower pH values or addition of Mg^{2+} increased adsorption. Koya and Chaudhuri (1977) examined adsorption of bacteriophage MS-2 to 3 soils. Soils were either silt or clay loams. Most virus sorption occurred in the first 20 to 30 minutes. Minimal adsorption occurred after 80 minutes. Increases in soil pH decreased virus adsorption. Introduction of a divalent cation increased viruses adsorption.

Cookson and North (1967) examined adsorption of viruses (phage T4) to activated carbon. Optimal adsorption occurred at ionic strength of .08 and a pH of 7. Varying temperature between 23 and 33°C did not affect virus adsorption. Adsorption could be explained by Langmuir isotherms suggesting that mono layer coverage was occurring. At maximum adsorption, only 18% of the surface area of activated carbon were covered. Desorption and adsorption reached the same point of equilibrium suggesting that adsorption is an electrostatic process. Adsorption was concluded to occur according to 2nd order kinetics.

Drewry and Eliassen (1968) examined adsorption of bacteriophages T1, T2, and f2 to 9 soils varying in clay contents from 6 to 39%. Adsorption followed Freundlich isotherms. Adsorption of viruses was enhanced below pH 7 to 7.5. Increasing concentration of cations increased virus adsorption. Drewry and Eliassen (1968) concluded that a single soil property could not be used to predict virus adsorption.

Bengtsson et al. (1964) examined adsorption of M^+ and M^- strains of poliovirus to cation and anion exchangers. Adsorption of the M^- strain occurred below but not above the IEP of the exchangers. Elution of viruses was only 8-12% in the initial buffer, but 65-75% of the viruses could be eluted with 0.2 m NaCl. Adsorption of M^- strain occurred above and below the IEP of the anion exchanger, but only below in the presence of anionic buffer suggesting an ion-virus complex.

Taylor et al. (1981) examined adsorption of poliovirus to soils and minerals. Isoelectric points (IEP) of poliovirus were determined using electrophoresis. The IEP was 7.5 at high pH values and 4.5 at low pH values. Adsorption of viruses to the silt loam, sandy loam, clay, and sand soils were similar over the pH range of 4 to 10. Almost complete adsorption occurred below pH 7.5. Above pH 8 adsorption was minimal. Muck soil showed maximum adsorption below pH 6.5, but an increase between pH 8 and 10. Increasing the electrolyte concentration increased adsorption. Electrolyte NaCl, $CaCl_2$, and Na_2SO_4 were all effective which is consistent with DDL theory.

The increases in adsorption at lower pH values in the preceding studies can be related to the IEP of the viruses. Most viruses have IEP above 5 (Gerba, 1984). A reduction in the soil pH reduces the negative charge on the virus. If the charge is reduced sufficiently the virus may approach a soil particle, be attracted by van der Waals forces (Moore et al., 1981), and be adsorbed. If the pH is reduced below the IEP of the virus, a positive charge will result and the virus can be adsorbed on a cation exchange site of the soil.

Effects related to addition of cations can be explained by cation bridging between viruses and multivalent cations (Koya and Chaudhuri, 1977; Gerba, 1984; Bengtsson et al., 1964), or reduction in the thickness of the diffuse double layer. Increases in cation concentration reduce the thickness of the diffuse double layer between the negatively charge virus and soil particle. If the layer is substantially reduced, attraction between the particles, associated with van der Waals forces, will occur resulting in virus adsorption (Moore et al., 1981; Gerba, 1984).

Adsorption of viruses has been shown to be effected by the mineral composition of the soil as well as the presence of organic matter. Green and Cliver (1975) examined adsorption of viruses in the presence of STE. Adsorption to fresh sand was 96%. Adsorption of viruses to sand after STE had been applied was reduced to 50%.

Goyal and Gerba (1979) reported that less adsorption of viruses occurred in the presence of SSE, than deionized water. Virus adsorption in both primary and secondary sewage effluents was similar and followed Freudlich isotherms.

Lo and Sproul (1977) examined the adsorption of poliovirus to várious silicate minerals in batch studies. Six minerals were used in the experiment: actinolite; enstatite; kyanite; microline; olivine; and sillimonite. Minerals were ground so that 90% of the particles were between 1 and 6 u. Adsorption of viruses ranged from 51 to 78%. Additions of SSE before and after viruses were applied reduced the degree of adsorption. Additions of SSE following virus adsorption resulted in 38 to 84% adsorption. These results suggest that organics have replaced the viruses on the exchange sites.

Sobsey et al. (1980) examined adsorption of polio and reo viruses following application of domestic sewage wastewater to soils. Virus adsorption occurred within 15 minutes. Clayey soils, and kaolinite and bentonite adsorbed the most viruses. On average, sandy soils or organic soils adsorbed < 40% of the viruses. Organic soils adsorbed > 80% of the reoviruses, but < 35% of the polioviruses. Substantial elution of viruses was observed with distilled water in the organic and sandy soils, and to a lesser extent in the other soils. Degree of elution was dependent upon the virus.

Bixby and O'Brien (1979) examined the effects of fulvic acid on adsorption of viruses to soil. In batch studies fulvic acid and viruses (bacteriophage) were added to a loamy sand soil. Fulvic acid was abstracted from a spodic horizon. Fulvic acid-virus complexes were examined using energy dispersive x-ray analysis (EDXA) and scanning electron microscopy (SEM). Viruses showed a preference for fulvic acid. At low concentrations, 99% of viruses adsorbed to fulvic acid versus 84% for soils. At high concentrations, fulvic acid and soils showed similar adsorption of viruses, 40 and 45%, respectively. A mixture of soil and fulvic acid showed > 99.5% adsorption of bacteriophage. The highest degree of adsorption occurred when viruses were first added to fulvic acid and the mixture was added to the soil. Bixby and O'Brien (1979) concluded that fulvic acid will compete for some of the virus adsorption sites in soils. In addition, viruses adsorption to fulvic acid may increase virus mobility in soil.

Sobsey and Hickey (1985) examined adsorption of poliovirus to humic and fulvic acids. Both humic and fulvic acids reduced poliovirus adsorption to either electro-positive or electro-negative filters. For positive filters at pH 7.5, organic acid adsorbed to exchange sites. For negative filters at pH 3.5, humic acid, which is much larger than fulvic acid, and was trapped in filters. At this low pH, viruses were positively charged and adsorbed to humic acid.

Taylor et al. (1980) examined adsorption of bacteriophage R 17 and reovirus type 3 onto allophane. An adsorption equilibrium between allophane and viruses occurred in about 15 minutes. Adsorption followed the Freudlich isotherm. At high virus concentrations, percent virus adsorbed versus percent added decreased. Above pH 7, adsorption of viruses to allophane decreased.

Koya and Chuadhuri (1977) examined adsorption of viruses to soils. The highest percentage of virus adsorption occurred in the soil with the highest clay content. This soil was also mostly kaolinite, quartz, and Fe and Al oxides. The effects of Fe or Al content, or low CEC, on virus adsorption was not discussed.

Moore et al. (1981) examined adsorption of poliovirus to 34 minerals or soils. Adsorption of viruses at high concentrations followed Langmuir isotherms and under low concentrations followed Freudlich isotherms. At maximum adsorption, only 1% of surface area was covered with viruses. Most minerals or soils adsorbed > 95% of the viruses. Minerals with low negative charges, or a positive charge showed the most adsorption. Hematite and magnetite were the most effective minerals for adsorption of viruses. Soils with high concentrations of organic matter showed lowest amounts of adsorption. Moore et al. (1981) suggested that because organic matter has a low van der Waals force, organic matter is a poor adsorber of viruses. Moore et al. (1981) concluded that the best adsorbents have little organic matter, low cationic capacities, high IEP, and strong van der Waals forces.

Column studies

A considerable amount of the research examining virus adsorption and transport has been conducted in column studies. The use of columns to simulate field conditions reduces inherent field problems. These problems are primarily associated with the difficulties involved in tracking viruses away from an OSWDS, and concentrating viruses from the large samples taken in the field, to a sample small enough that viruses can be detected. Bitton et al. (1976) examined adsorption and desorption of viruses from A, E, and Bh horizon of a Florida Ochraquult. These soils contained 36 to 97% sand. Bacterial phage T2 and poliovirus type 1 were applied to the top of soil columns and leached with either SSE or tap water. The SSE easily displaced viruses. Minimal losses of viruses occurred during leaching with tap water. Bitton et al. (1976) concluded that organic substances were reducing virus retention to the sands.

Bitton et al. (1973) applied viruses and sludge to disturbed and undisturbed soil columns. The soil contained 13 to 36% clay. In disturbed columns amended with sludge, breakthrough was observed after 7 pore volumes had been added. After 10 pore volumes had been added 99.99% of viruses added in $CaCl_2$ solution were attenuated. Subsequent leaching with rain water did not cause virus desorption. In undisturbed columns, viruses were observed in column effluent after only 1 or 2 pore volumes had been added. Undisturbed columns were not treated with $CaCl_2$ suggesting virus retention may be affected by divalent cations. Addition of rain water after initial breakthrough did not show evidence of desorption.

Duboise et al. (1976) examined the fate and transport of viruses in undisturbed 19.5 cm long columns of a loamy fine sand soil. Virus movement was greater in distilled water than in SSE due to the high ionic strength of the effluent. Intermittent additions were shown to increase adsorption compared to continuous flow of either distilled water or SSE. Enhancement of adsorption occurred when pH values were reduced to 5.5. Most adsorption of viruses occurred in the upper 2 cm of the columns, but secondary peaks were observed with depth. Secondary peaks indicated continued release and movement of viruses during subsequent applications of SSE resulting in changes in equilibrium conditions. Cycles of addition of effluent followed by distilled water, eluted 22% of the viruses initially adsorbed. Desorption increased when pH values were raised to 9.

Green and Cliver (1975) examined adsorption of viruses in the presence of STE. Columns treated with STE prior to addition of viruses showed enhanced movement. After 3 days viruses had moved 45 cm, but in untreated sand only 20 cm. Continuous flow of effluent showed 96% retention of the viruses. Intermittent dosing of STE, at the same loading rates as the continuous flow experiments, caused ponding in the columns, and only 61% of the viruses added were adsorbed.

Lo and Sproul (1977) examined the effect of SSE on the adsorption of poliovirus to microcline. Microcline was ground to pass a 20 mess screen and packed in 16.5 cm long columns. Adsorption of viruses was totally to nearly complete in columns regardless if SSE was added. In columns with SSE added however, breakthrough of viruses was observed first.

Lance and coworkers (Lance and Gerba, 1980, 1984a, 1984b; Gerba and Lance, 1978; Lance et al., 1982) examined adsorption and desorption of viruses in column studies. Columns were filled with 250 cm of loamy sand to simulate the Flushing Meadows RI system. Lance and Gerba (1984a) reported that viruses moved farther in columns in which deionized water was applied, than those columns in which tap water or SSE were applied. Cations in the higher ionic strength tap water and SSE most likely promoted virus adsorption. Most (95-100%) adsorption occurred in the upper 20 to 40 cm of the columns. Additions of anions enhanced adsorption of viruses with molecular anions NO_3^-, SO_4^-, and $H_2PO_4^-$ more effective than Cl^-. The Al^{3+} cation was the most effective cation in enhancing adsorption. Lance and Gerba (1984a) concluded that Al^{3+} probably promoted flocculation of viruses and could be used as an additive to increase treatment of viruses.

Lance and Gerba (1984b) and Gerba and Lance (1978) examined adsorption and desorption of poliovirus. Poliovirus was added with SSE to the columns and solution samples were obtained through ceramic samplers at depths of 2, 10, 20, 40, 80, 160, and 240 cm. Lance and Gerba (1984b) reported that at the lowest infiltration rates, viruses did not move beyond 40 cm. At the highest infiltration rates viruses moved to 160 cm. In only 5 out of 43 samples collected, were positive results observed for viruses in the column effluent. Additions of deionized water caused desorption of viruses. Elutions of viruses with $CaCl_2$ solutions however, were unsuccessful. Drying the columns for 1 day reduced virus desorption, and drying for 5 days eliminated virus desorption. Gerba and Lance (1978) reported that the degree of virus adsorption was similar for addition of either primary and secondary effluents. Most viruses were attenuated in the upper few cm of the columns.

Lance and Gerba (1980) observed that increasing the loading of viruses increased the number in column effluents but the ratio of concentration in the effluent to concentration applied (C/C_0) remained the same. Increasing flow rates within columns increased the depth to which viruses were leached as well as the respective numbers. Lance et al. (1982) reported similar results. Lance and Gerba (1980) concluded that flow rate may be the single most important factor governing movement of viruses in the RI system studied. Lance and Gerba (1980) suggested that viruses with the lowest net negative charge would be the first to adsorb and the other viruses would move deeper in the column.

Lance et al. (1982) reported that 90% of the viruses were adsorbed in top 2 cm, and 99.9% in upper 160 cm of the columns. Distribution of different types and strains of Echo and polio viruses within columns were similar. Much less variability was observed in adsorption behavior between strains of Echo virus 1 and 29, in the columns, than comparable batch studies performed by Goyal and Gerba (1979).

Lefler and Kott (1974) examined adsorption of poliovirus 1 and bacteriophage f2 to sand in small, 10 cm long columns. Low cation concentrations or monovalent cations such as Na^+ showed minimal effects on virus adsorption. Divalent cations showed significant effects in enhancing virus adsorption. A single washing with tap water desorbed 62 to 68% of viruses.

Sobsey et al. (1980) examined adsorption of polio and reo viruses to soils. Greater than 99.995% of viruses were adsorbed to soils with high clay contents. Sandy and organic soils adsorbed 95% of viruses applied, but subsequent desorption was significant. Desorption in clayey soils was non existent.

Wang et al. (1981) examined movement of Echo and Polio viruses in columns filled with sands, or sandy loams. Soils were screened at 0.8 mm to remove coarse sand and larger particles. Columns were 1 m long and sampled at various depths along the column. Columns were sampled with ceramic samplers that were shown in Wang et al. (1980) to trap < 10% of the viruses. Both Echo and Polio viruses were attenuated at similar rates in columns. Batch studies however, showed that poliovirus adsorbed to a greater extent than Echo virus to these soils. Sandy loam soil adsorbed most of the viruses in the upper 7 cm. Flow rates were found to be the most important factor in virus attenuation. High flow rates were negatively correlated with virus retention.

Jorgensen (1985) studied the moment of several viruses applied in sludge to columns packed with sand and sandy loam soil material. In unsaturated columns no viruses were detected below 3.5 cm. Saturated sandy loam columns showed virus breakthrough (at 30 cm) before 0.7 pore volumes had been applied to the columns. Viruses were not observed in the effluent of sand columns. Jorgensen argued that the sandy loam soil had a greater percentage small pores, and therefore a faster linear flow velocity. This is contrary to most other research. Data of the study were not well documented. The pH of sand column effluent was 4.5, while the pH of the effluent from the sandy loam column was 7. These data may better explain the movement, or lack of, for the viruses in the respective columns.

Matthess et al. (1988) investigated the transport of viruses in 10 cm columns filled with sand. The distance in which the viruses moved depended upon type of virus, pH, and flow velocity. Higher pH values and flow velocities showed greater movement of viruses.

Pancorbo et al. (1988) applied poliovirus type 1 to 27 cm columns filled with sandy clay loam soil. Columns were amended with electrolyte conditioned sludge, chemical sludge coagulated with lime, Fe, and Al, and chemical sludge (pH 1). No viruses moved through the columns after leaching with rainwater. Greater than 97% of the viruses were recovered from sludges.

Powelson et al. (1990) applied viruses to 105 cm long columns filled with loamy sand. Movement of viruses were monitored at depths of 20, 49, 80 and 105 cm under saturated and unsaturated flow. Samples were collected with stainless steel samplers to avoid problems associated with viral retention in ceramic cup samplers. In columns maintained at saturated conditions, 2 pore volumes were added before the concentration of viruses in the influent was equal to that of effluent. In unsaturated columns, most viruses were attenuated within the soil column.

Drewry and Eliassen (1968) examined virus adsorption in 43 to 50 cm long columns. Most viruses were adsorbed in the upper 2 cm. All soils showed > 99%

removal of viruses. Drewey and Eliassen (1968) concluded that for most agricultural soils separation distances of 30 to 46 m from ST-SAF to wells would be sufficient for adequate virus treatment.

Field studies

The amount of field research on virus movement in soils has been minimal. Most of the field studies that have been conducted were focused on virus movement in conditions conducive to saturated flow, or in soils with coarse textures (RI systems). Cogger et al. (1988) examined movement of viruses within a Quartzipsamment in North Carolina. Two sites were examined in which the height of the water table differed by 30 cm. The highest water table was within 30 cm of the soil surface 73% of the study period. Wells were installed at distances of .3 and 3 m from the ST-SAF. The STE was applied under LPD. High concentrations of viruses were observed in soil with higher water tables. Cogger et al. (1988) concluded that in these sandy soils a 30 cm separation distance between drainlines and high water table was inadequate for significant virus removal.

Stramer and Cliver (1981) examined virus attenuation from ST-SAF under saturated conditions. Viruses were detected to a depth of 60 cm in a saturated silt loam soil as well as samples from the nearby lake. Stramer and Cliver (1981) concluded that under saturated conditions soils are not able to filter all viruses, and they may move into ground and surface waters.

Scandura and Sobsey (1981) examined the survival and fate of enteric viruses in OSWDS in coastal plain soils. Four ST-SAF with high water tables were examined. Viruses survived in septic tanks for 5 to 6 weeks. Die-off rates followed first order kinetics. Viruses were detected in ground water wells near ST-SAF following rainfall events. Scandura and Sobsey (1981) concluded that OSWDS placed in coastal plain soils with high water tables are contributing to pollution of the groundwater in these areas. Wellings et al. (1975a) examined transport of viruses applied in SSE to a cypress dome. Wellings et al. (1975a) found that 84% of the viruses recovered were attached to filters, or solids attached to filters, during concentration of samples.

Viruses survived at least 28 days and moved at least 7 m from the point of application.

Vaughan et al. (1983) examined lateral movement of viruses applied as STE in sandy soils. Groundwater was 3.6 m below the point of application of the STE. Wells were installed at 11 distances between 1.5 and 67 meters from the ST-SAF. In general, the number of positive tests for enteric viruses decreased with distance from the ST-SAF. During one sampling period, samples collected at 60.4 m were 9% positive for enteric viruses. Samples collected at depths of 18 m below the surface, and 67 m from the ST-SAF were positive for enteric viruses 7.7% of the time. Coliforms were rarely found at distances > 1.5 m.

Brown et al. (1978, 1979) examined the movement of viruses within 3 ST-SAF. The ST-SAF were placed in a sandy loam soil, a sandy clay textured Udifluvent, and a Haplustalf with primarily clay textures. Coliphage numbers were lower in STE during the winter months than the rest of the year. Leachate was collected in ceramic cups 120 cm below drain lines. In the first year of the study, of the 230 leachate samples collected, only 2 tested positive for viruses. Coliphages were found in soil below all lines, and in some cases 30 to 60 cm in the horizontal direction from drain lines. Viruses survived 3 to 4 weeks after STE ceased to be applied.

During the second year of the study (Brown et al., 1979), viruses were only found in leachate samples after the STE had been spiked with viruses. In the soil, viruses had moved along macropores about 1 m.

Gilbert et al. (1976) applied wastewater to an RI System. All viruses were removed after percolation through 9 m of primarily sandy soils.

Schaub and Sorber (1977) examined the fate and transport of coliphage f2 and enteric viruses after application of wastewater to an RI System in Massachusetts. Coliphage f2 was observed in groundwater 28.8 m below the point of application within 48 hours. Fifty percent of the phages, and about 10% of the enteric viruses were recovered in the groundwater. Viruses were detected at distances > 183 m from their entry into the groundwater.

Vaughan et al. (1978) examined virus movement in an RI System in Long Island, NY. Recharge basins were primarily coarse sand and fine gravel. Three sites were examined. Viruses were detected at 5.5 and 10.4 m in two of three sites. No viruses were detected at 24.4 m at the other site. Viruses were detected in wells 46 m down gradient of the point of application.

Vaughan et al. (1981) examined movement of viruses applied as tertiary treated wastewater to an RI System. Highest infiltration rates showed the most movement of viruses. At high rates, viruses were recovered at 7.6 m below the point of application.

Martin and Thomas (1974) introduced a bacteriophage into sandy, shallow groundwater aquifer. The bacteriophage was monitored with distance from the point of application by sampling groundwater from wells along the suspected flow path. The phage was observed 680 m from the initial source.

Fate of viruses in alternative systems

The amount of research focused on alternative OSWDS for removal of viruses has been minimal. Green and Cliver (1975) examined movement of viruses in columns constructed to simulate mound systems. Columns treated with STE prior to addition of viruses showed enhanced movement. After 3 days viruses had moved 45 cm, but in untreated sand only 20 cm. Intermittent dosing of STE caused ponding in the columns and 61% attenuation of viruses. Continuous flow at the same loading rates showed 96% retention of the viruses. At room temperature, 97% of viruses were inactivated, but at 8°c only 37% of viruses were inactivated.

Gross and Mitchell (1985) examined the effectiveness of columns, constructed to simulate a sand filter, in removing viruses applied in STE. In columns treated with STE for 3 months prior to introduction of viruses, viruses showed a breakthrough when virus loading rates were increased to 33 million PFU/L. Average removal of viruses was 99.996%. Columns that were not preconditioned with STE showed no virus breakthroughs. Most viruses were attenuated in upper 2.5 to 3.8 cm of the columns.

Schaub et al. (1982) examined the fate of viruses in wastewater applied using spray irrigation. Bacteriophage f2, poliovirus 1, and enteric viruses were applied to sandy loam and silt loam soils with < 5% clay. Greater than 99.9% of f2 bacteriophage was adsorbed. Bacteriophage f2 was recovered at 45 and 170 cm within 4 days of application. Significant amounts of viruses were collected at 45 and 170 cm depth following heavy rains. Almost all of the f2 bacteriophage introduced into the system had disappeared after 53 days. Similar results were reported for poliovirus.

Wellings et al. (1974) examined the fate of viruses applied using spray irrigation of SSE. Viruses were shown to survive both chlorination and UV light preceding spray irrigation. Viruses were found at 6.1 m below the surface of a sandy soil.

Virus standards for groundwater

Currently, most states in the U.S. do not have a drinking or recreational water standard for viruses. Several papers have noted elevated concentrations of viruses in groundwater samples, when coliform counts were minimal or totally absent. El-Abagy et al. (1988) examined drinking water from various sources in Egypt. In 147 samples tested, 78 samples were positive for coliphages but negative for total and fecal coliforms. El-Abagy et al. (1988) concluded that survival of coliphages are greater than those of pathogenic bacteria, and standards for coliphages in drinking water are necessary.

Bell (1976) examined survival of bacteria and coliphages in effluents and river waters. After 7 days, the ratio of fecal coliforms:coliphages in raw effluent decreased from 87:1 to 1:1. These data suggest that survival times of viruses are much higher than those of fecal bacteria.

Sobsey et al. (1980) examined attenuation of fecal bacteria and viruses in columns. Columns were filled with one of four soils with varying properties and characteristics. Retention of bacteria were 1.3 to 1.1 orders of magnitude greater than the retention of viruses. Sobsey et al. (1980) concluded that fecal bacteria may not be an adequate measure of virus contamination of drinking water. Vaughn et al. (1983) examined the movement of coliforms and enteric viruses from an OSWDS located 3.6 m above a shallow, sandy aquifer in New York. Over 7% of the samples collected from the groundwater at distances greater than 67 m from the OSWDS tested positive for enteric viruses. Coliforms were rarely detected in the groundwater at distances greater than 1.5 m.

These studies indicate that the potential for pollution stemming from OSWDS may be greater for viruses than for bacteria. Many researchers and reviewers (Safferman, 1982; Berg, 1973; Gerba et al., 1975; Keswick and Gerba, 1980; Sobsey et al., 1980; El-Abagy et al., 1988) suggest that virus standards for groundwater are necessary. Until available methods are developed to measure viruses in ground and surface waters, changes in setback regulations may be more appropriate. Yates et al. (1986) and Yates and Yates (1988) collected water samples from 71 wells in Arizona to determine safe set back distances of septic tanks from wells. Yates et al. (1986) used a spatial geostatistic technique, called krieging, to determine the necessary set back distances to protect wells from pollution due to OSWDS. Factors incorporated into statistical analysis included virus decay rates, hydraulic gradients, hydraulic conductivity and well water temperatures. The thickness of the unsaturated zone was not included in the model. Results suggest that set back distances of 15 to 150 m are necessary to protect groundwater wells from virus pollution.

Summary and conclusions

Viruses are small microbes, generally less than 250 nm, that behave in the soil system as a colloid. There are over 100 different viruses excreted by man. The most pathogenic viruses are hepatitis A, Norwalk agent (rotovirus), poliomyelitis, coxsackie, rota, and adeno. Hepatitis A and Norwalk agent have been the most infectious waterborne viruses in recent years. Nine percent of the outbreaks of water borne disease reported between 1971 and 1982 could be attributed to these viruses (Craun, 1985).

Natural die-off and adsorption are the two most important means of reducing the number of viruses applied to a ST-SAF. If viruses cannot survive during the travel period between the ST-SAF and a groundwater well, these pathogens will not be a viable problem. Rates of survival vary between both type and strain of virus. Several factors including temperature, degree of adsorption, soil type and composition, soil moisture content, and amount of microbial competition have been shown to affect survival rates. Temperature is the most important factor. Viruses maintained at low temperatures have considerably higher survival rates. Degree of adsorption, although not nearly as important as temperature, is positively correlated with virus survival rates. Sobsey et al. (1980) reported average survival times of reo and polio viruses in sterile soils of 123 and 95 days, respectively. Jorgenson and Lund (1985) reported survival times of viruses maintained at temperatures from 4 to 7^0 C of greater than 34 weeks.

Adsorption is the primary process of attenuation of viruses in soils. The physical process of adsorption is governed by two types of forces. These forces are the attractive and repulsive forces between the virus and soil particle within the diffuse double layer, and van der Waals forces. Extensive batch studies have been conducted investigating the factors which affect virus adsorption to a soil. The most important factors are the charge of the soil and virus, pH, IEP, concentration and valence of cations in the soil solution, and the amount of organic colloids in the system. Viruses are amphoteric particles having either a positive or negative charge depending upon the pH. If the virus is positively charged and the soil negatively charged, the virus will be adsorbed. The pH at which the virus is neutral is called the IEP. Most viruses have IEP greater than 5, so that reducing the pH increases the amount of virus adsorption. Multivalent cations may act as a bridge between the negatively charged virus and soil particle, and thus enhance adsorption. An increase in the concentration of cations will also increase adsorption by reducing the thickness of the diffuse double layer, and thereby enhancing adsorption. Organic colloids in the system can reduce the amount of adsorption by competing for virus adsorption sites.

In columns constructed to simulate field conditions, viruses acted in some respects similar to the batch studies, but in others completely different. In most studies in which viruses were added with STE, SSE, or other sources of organic colloids, the amount of adsorption decreased with the addition of an organic source. In cases where the STE or SSE had a high ionic strength, addition of SSE or STE increased adsorption. An increase in desorption of viruses was shown to occur with the addition of STE or SSE for most systems. In some cases however, distilled water was shown to desorb more viruses than STE, because the STE had a high ionic strength.

Flow rates were the single most important factor governing movement of viruses. Columns constructed from undisturbed soils material, or columns maintained so that saturated flow occurred, showed the most virus movement. Several studies showed virus movement in saturated columns with minimal reduction in the number of viruses. Soils with finer textures showed greater reductions in viruses. Batch studies indicated that sandy or organic soils were poor adsorbers of viruses. In column studies however, even sandy soils, or soils with considerable organic matter adsorbed > 95% of the viruses applied, if columns were maintained to assure unsaturated flow. Sandy soils, or soils with high organic matter showed considerably more desorption than soils lacking organic matter and of finer textures. Sandy soils showed as much as 68% desorption of viruses with a washing of distilled water. Batch studies indicated that virus adsorption differed between types and strains of viruses. In soil columns, most viruses behaved in a similar manner regardless of type or strain.

The number of field studies, or studies of the fate of viruses in alternative systems, have been minimal. Field studies suggest, that in soils conducive to rapid flow, such as soils with coarse textures or high water tables, movement of viruses can be rapid resulting in groundwater pollution. Schaub and Sorber (1977) reported coliphages had moved as much as 29 m beneath an RI system. Once in the groundwater viruses can move even greater distances. In the same study, Schaub and

Sorber (1977) detected viruses > 183 m from their point of entry into the groundwater. Vaughn et al. (1983) detected viruses in groundwater at a depth of 18 m, and a distance of 67 m from a ST-SAF placed in sandy soils. In studies where bacteriophages were introduced directly into the groundwater (Martin and Thomas, 1974), phages were record 680 m from the application point.

The potential for pollution of the groundwater from OSWDS may be greater for viruses than bacteria. Several studies indicated that survival rates, and degree of movement were greater for viruses than those of fecal bacteria. Numerous researchers have called for drinking water standards for viruses. Because of the difficulty in assessing virus pollution in the groundwater, changes in setback distances may be the best immediate approach to assure that pollution from viruses attributed to OSWDS is maintained at a minimum.

OTHER POLLUTANTS

Bacteria, viruses, N, P, and organics are the major components of STE that are of concern as pollutants to ground and surface waters. Several other components of STE may also be considered as potential pollutants. Those of most concern are heavy metals, cations such as K, Ca, Mg, and Na, the anions Cl^- and SO_4^{2-}, and methylene blue active substances (MBAS). Bouwer and Chaney (1974) reviewed the fate and transport of heavy metals applied to soil in waste waters. In general, heavy metals in domestic wastewaters are very low. Mobility of these metals in soils is limited and best explained by pH, exchange capacity, and organic matter content (Bouwer and Chaney, 1974). High pH conditions favor metal immobility. Metals readily adsorb to the exchange sites of organic matter or clay particles. Therefore, soils with organic matter, clay, or an elevated pH will attenuate most heavy metals.

Sandhu et al. (1977) surveyed 217 rural water supplies in Chesterfield County, South Carolina for inorganic pollutants. Contamination from metals such as Pb, As, Hg were common, and in some cases above regulatory limits. Arsenic was found in 93% of the water samples. Sandhu et al. (1977) concluded that OSWDS were at least partially responsible for the contamination.

Bouwer et al. (1974) examined the movement of heavy metals through the Flushing Meadows RI System. Concentrations of Zn and Cu in the renovated wastewater were much lower than in the initial SSE. Levels of Cd, Hg, and Pb however, did not show considerable reduction following waste water renovation.

Brown et al. (1978) examined the movement of heavy metals from 3 ST-SAF. Most metals were found at the soil-gravel interface of the adsorption trench. Metal concentrations in soils directly adjacent to drain lines were as much as 2 times greater than background levels. Metal concentrations in water samples however, were at background levels. Brown et al. (1978) concluded that because levels of metals in STE were low, and metals were concentrated directly adjacent to the ST-SAF drain lines, pollution of soil or groundwater from heavy metals associated with OSWDS should be of little concern.

Substantial increases in cations or anions can reduce the consumptive or irrigation quality of ground and surface waters. Although not considered a serious problem, with time (especially in areas with fairly saline ground or surface waters), addition of ions from OSWDS may be of some concern. DeWalle and Schaff (1980) examined well records and water samples over a 30 year period near Tacoma, Washington. The area examined was underlain by glacial deposits. Of the 242,000 residents of the area, 100,000 used ST-SAF for wastewater treatment. Comparisons of average well data with average STE data indicated 20X less Na and Cl^-, 6X less K and SO_4^{2-}, and 5X less Ca and Mg in the well waters. Over the 30 year period, specific conductance, Ca, and Cl^- were shown to increase in well water due to additions from OSWDS.

Hill (1972) examined movement of Cl^-, K, SO_4^{2-}, Mg, Na, and Ca in undisturbed cores treated with STE. Soil cores were 1 m in length and represented 6 different soil series in Connecticut. Initially 90% of the SO_4^{2-} was removed. After 2 years however, no SO_4^{2-} was being removed. No Cl^- was removed by the soil columns. Between 75 and 85% of the K, Mg, and Ca applied as STE was attenuated by the soils. Only 10% of the Na was removed.

Reneau et al. (1979) examined the changes in pH, base saturation, and exchangeable Na percentage (ESP) of soils (Ochraquults) adjacent to a ST-SAF. In general, the ESP, pH, and base saturation decreased with distance from the ST-SAF. Increases in ESP, pH, and base saturation however, were observed at distances > 17 m from the ST-SAF. The largest increases occurred in the B horizons of the Ochraquults.

Alhajjar et al. (1989) examined STE in 17 septic tanks using either P-based or CO_3-based detergents. Concentrations of Na, K, Ca, Mg, and Cl^- were greater for systems using P-based detergents than those using CO_3-based detergents.

Willman et al. (1981) examined the effect of various sand and clay mixtures for treatment of STE. Clay contents from 0-12% were added to either limestone, sandstone, or shale sands. These mixtures were packed into columns to simulate a mound system. After 11 weeks no differences in Na content between STE and column effluent were observed. After 23 weeks, no reduction of K or Ca occurred in the columns. Columns filled with limestone-clay mixtures always showed Ca effluent levels higher than those of STE. No reductions in Cl^- concentration occurred during the study.

Methylene blue active substances (MBAS) are anionic surfactants added to domestic detergents. These surfactants can account for 10 to 30% of the detergent's mass. Most detergents switched from the alkylbenzene sulfonate (ABS) to linear alkylbenzene sulfonate (LAS) because LAS was reported to degrade more readily than ABS (Reneau and Pettry, 1975). Alhajjar et al. (1989) examined MBAS concentrations in effluent from 17 septic tanks. Surfactants (MBAS) were absent in nearly all of the STE suggesting that MBAS are degraded in the septic tanks.

Reneau and Pettry (1975) examined the movement of MBAS substance in soils adjacent to 3 ST-SAF in the Coastal Plain of Virginia. Decreases in MBAS were observed with distance from the ST-SAF. These decreases were related to adsorption, biodegradation and dilution. Concentrations of MBAS were lower in the summer, than the cooler months. Lower concentrations in the summer suggest greater biodegradation of MBAS during periods of warmer temperatures. These differences in degradation rates were not observed in wells placed at the greatest distance from the ST-SAF (6.1 m), suggesting that some of the MBAS were more resistant to degradation than others. In batch studies, Reneau and Pettry (1975) examined adsorption of LAS to soil. On average, between 56 and 66% of LAS added to the soils were adsorbed. Greater adsorption was related to higher clay, exchangeable acidity, and Fe contents.

SUMMARY AND CONCLUSIONS

Virginia relies heavily on OSWDS to treat domestic wastewaters. Approximately 30% of Virginia households use OSWDS to treat domestic wastewater. Over 40,000 new OSWDS permit applications are received each year by the VDH. Annual increases of this degree strongly suggests that the number of Virginia households using OSWDS will only increase in the future. Increased use of OSWDS is accompanied by an increase in potential pollution to the ground and surface waters by these systems. Concerns over health and potential degradation of water quality dictate that OSWDS contribute as little as possible to pollution of ground and surface waters of Virginia.

Most OSWDS are conventional type systems using a septic tank and a subsurface soil adsorption field (ST-SAF). Many soils however, are unsuitable for conventional OSWDS because of high water tables, shallowness to bedrock, extremely slow or high percolation rates, or restrictive layers at shallow depths such as fragipans. Several alternative systems have been developed to make these sites more suitable for domestic waste disposal. The more commonly used alternative systems are mounds, low pressure distribution, aerobic units, sand filters, electro-osmosis, evapotranspiration, and spray irrigation.

The degree of purification and filtration of wastewater by soil is primarily dependent upon the hydraulic characteristics of a soil. In general, the slower the wastewater moves through the soil the more likely soil-wastewater interactions will occur leading to increased filtration and purification. Movement of water through soil is dependent upon the type and distribution of voids in a soil, and the amount and frequency of water applied. The larger the voids the faster the flow rate. Therefore, soils with coarse textures or well developed structure have the highest flow rates. These soils are least suited for conventional OSWDS. The moisture content of a soil also dictates the rate of flow in a soil. The higher the moisture content, the higher the rate of flow. In soils that are saturated or nearly saturated, the majority of the flow occurs in the large voids. Minimal treatment occurs during these periods because potential contaminants have limited contact with soil particles. This type of flow, termed by-pass flow, often occurs in ST-SAF and can lead to short-circuiting of the purification process.

Clogging mats are universal to all ST-SAF. A clogging mat is a layer of organic material that accumulates at the interface between the gravel envelope around drain lines, and soil receiving the STE. Clogging is initiated by an accumulation of suspended solids from the STE. These suspended solids are linked together with soil particles by biologically produced material forming a restrictive layer of organic material. Clogging mats have both beneficial and detrimental effects in an OSWDS. A well developed clogging mat often forms a restrictive layer leading to hydraulic failure (ponding) of the system. In systems in which the clogging mat is poorly developed, treatment efficiency of the system is sacrificed. Clogging mats increase treatment efficiency by filtering organic matter and microbes as well as controlling saturated flow. This is especially true for coarse-textured soils.

The fate and transport of various components in STE have been studied in both the field and laboratory. Laboratory investigations can be grouped as either batch or column type studies. Most batch studies are performed in a closed system in order to examine the interactions between STE constituents and soils. Column studies are designed primarily to simulate field conditions. Under simulated field conditions, many of the important factors and parameters that effect interactions in the soil can either be maintained at a constant (such as temperature or soil moisture content), or easily monitored during the study period. Field studies examining the effect of

OSWDS to the environment can be limited to the immediate area around a drain line, or expanded to cover the entire recharge area of an aquifer. Changes in the chemical characteristics or concentrations of soil, soil solution, or groundwater are usually monitored in field studies. These changes are examined to determine if a cause and effect relationship can be attributed to an OSWDS.

The composition of STE varies depending upon the household. The STE components with the most potential as ground and surface water pollutants are N, P, bacteria, viruses, and organics. Heavy metals, cations, anions, and surfactants may also be of some concern as pollutants in certain systems.

Domestic wastewaters contain various organic components that are usually expressed as biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), and total suspended solids (TSS). The source of most of these organics is the garbage disposal and toilet. Reductions in organic concentrations can occur in either the septic tank or the ST-SAF. In septic tanks, organics are reduced through the processes of sedimentation, filtration, or biodegradation. Removals of > 60% of BOD, > 70% of the SS, and > 85% have been reported for standard septic tanks. Aerobic units have been used to reduce the BOD levels to even greater degrees.

Organic material applied as STE to a soil is generally filtered or biodegraded by the microflora. Factors which affect the fate of organic material in a ST-SAF are the type of organic material, population and type of microbes, surface area available, aeration status, loading rate of STE, and temperature. In properly sited and operating OSWDS most of the organic components are easily filtered or degraded. Therefore, the potential for pollution related to organic material from an OSWDS is minimal.

Nitrogen is the most widely studied ground and surface water contaminant attributed to OSWDS. Most N released from a ST-SAF is in the NO_3^- form which is very mobile in the soil system and can easily move into ground and surface waters.

Nitrogen contaminated ground water is the primary cause of methemoglobinemia, which is toxic to infants. Nitrate in surface waters can promote eutrophication.

Most of N in STE is NH_4^+ (75-85%) or organic-N (15-25%). Several mechanisms and processes effect the fate and transport of N following application of STE. The complexity and degree in which each process affects N in the ST-SAF is dependent upon various soil and environmental factors. Mineralization, nitrification, denitrification, and cation exchange are the most important processes contributing to the form and mobility of N in the system.

Mineralization is important in releasing organically bound N. This process is primarily restricted to the clogging mat and upper few centimeters below the gravel-soil interface of the ST-SAF, where most of the organic matter accumulates. Most N released during mineralization occurs in the NH_4^+ form.

Cation exchange is important in holding NH_4^+ on the exchange sites until nitrification occurs. Leaching of NH_4^+ can occur if the exchange sites become saturated with respect to NH_4^+, and conditions are not conducive for nitrification. Leaching of NH_4^+ into the groundwater occurs under saturated conditions, following application of STE at high loading rates, or in soils with a low CEC. High loading rates and saturated flow conditions increase flow rates which enhance movement of NH_4^+ away from a ST-SAF and into groundwater.

In most properly sited and operating OSWDS, NH_4^+ is rapidly oxidized to NO_3^-. In most aerobic soils NO_3^- moves quickly and without reduction away from the ST-SAF and into the groundwater. In such cases dilution is the only form of N treatment between the ST-SAF and the groundwater. The rate at which NO_3^- moves away from the ST-SAF, and into the groundwater is dependent primarily on the flow rate. Greater flow rates increase movement of NO_3^-. Soils with coarse textures, strong structure, or under saturated conditions are the most susceptible to rapid transport of NO_3^-.

Nitrate contamination of groundwater from OSWDS is more than a potential problem. Quan et al. (1974), Miller (1975), and Mancl and Beer (1982) all reported elevated levels of NO_3^- in groundwater attributed to OSWDS.

Denitrification is the most important process for removing N. In this biological process, NO_3^- is reduced to a gaseous N form which is released into the atmosphere.

Alternative OSWDS systems have shown mixed success in adequately reducing N levels. The amount of success a particular alternative is often dependent upon the individual OSWDS. Several studies indicated that nitrification occurred within mounds and subsequent denitrification occurred at the mound-soil interface. The amount of denitrification however, decreased with time as the carbon within the old soil surface became exhausted. These studies suggest that if mound systems are to provide adequate treatment for N, an additional carbon source such as methanol will be necessary.

Alternatives such as recirculating systems and denitrification units (RUCK system) may provide the best methods to reduce N from OSWDS. These systems have been reported to provide from 50 to 85% reduction in N concentrations. Addition of a soluble carbon source has been shown to increase the effectiveness of these units.

Restricting the use of OSWDS in high density areas, or increasing the lot size of homes in recharge areas of aquifers susceptible to N groundwater pollution, is an immediate step that can be taken to reduce pollution from OSWDS.

The fate and transport of P applied to soils as STE has been extensively studied. Sources of P in STE are primarily P-based detergents and human excreta. Most P in STE is in the orthophosphate form and is readily adsorbed to soils which contain reactive Fe or Al. Following adsorption P may precipitate to a separate mineral form. Organic or mineral soils with minor amounts of Fe and Al show minimal P sorption capacity. Desorption of P is always much slower than adsorption but the effect can be considerable when anaerobic conditions exist, organic ions are present, or when the concentration of P in the soil solution is very dilute.

In most soils in which Fe and Al are present in a reactive form, and flow rates are minimal, P movement is minimal and concerns of pollution to ground or surface waters from P applied in to ST-SAF are unfounded. Phosphorus pollution to the groundwater however, can occur in cases where the water tables are near the surface, soils have coarse textures, flow rates are increased due to strong soil structure, loading rates are high, soils have a low P adsorption capacity, or when P capacity of a soil has been met. The major factor contributing to P movement in soil is the flow rate. Increased flow rates are generally associated with soils of coarse textures, strong structure or macro voids associated with biological activity, or high water tables.

The effectiveness of alternative OSWDS such as sand or peat filters, or mound type ST-SAF on treatment of P depends upon the composition of the filter material. If significant amounts of Fe are present coating sand grains or in association with the finer fraction, significant P reduction will occur. In filter media void of reactive Fe or Al, minimal reduction in P will occur. Alternative methods such as P conservation or treatment of P within the septic tank may be the best approach to reducing the effect of P to the pollution of ground and surface waters.

Contamination of ground and surface waters from bacteria is one of the primary concerns with the use of OSWDS. Under certain soil and environmental conditions, bacteria can move away from ST-SAF and into nearby ground and surface waters. Bacteria in groundwaters can initiate significant health problems, and promote outbreaks of waterborne disease.

Bacterial pollution of groundwater from OSWDS appears fairly widespread. Surveys of bacterial contamination of ground and surface waters have been conducted in numerous states including Washington, Oregon, Iowa, Maryland, Virginia, Texas, and South Carolina. These studies have shown that shallow aquifers, in areas where

OSWDS is the primary means of treating domestic wastewater, have considerable bacterial pollution.

Reduction in bacteria concentrations in STE is primarily dependent upon the processes of inactivation (natural die-off), filtration, and adsorption occurring in the ST-SAF. Rates of bacterial inactivation are an important consideration in groundwater pollution from OSWDS. If pathogenic bacteria can survive long enough to travel from a ST-SAF to a groundwater well, a serious health problem may result. Most studies indicate that bacteria will survive for the longest period in cool and moist conditions.

Attenuation of bacteria in the soil is primarily a function of the processes of adsorption and filtration. Soils with fine textures, minimal structure, and low pH values adsorb and filter nearly all bacteria. Therefore, movement of bacteria through well drained, fine and medium textured soils is minimal. Numerous studies however, have shown that in soils with coarse textures, considerable structure, and high water tables (conditions associated with high flow rates), movement of bacteria is significant resulting in groundwater pollution.

Mound systems were effective in removing bacteria applied as STE. These systems were especially effective if soil was available below the mound for additional treatment. Most sand filters were not very effective, and some additional treatment prior to introduction of effluent into ground or surface waters would be necessary. Sand filter and RI systems with well developed clogging mats showed much better treatment of bacteria than those without these organic concentrations. Peat filters, or filters combining both sand and peat showed significant reductions in bacteria. These systems were much more effective than the sand filters. The use of LPD in soils with high water tables appears to be more effective than conventional systems under similar environmental conditions.

Viruses are small microbes, generally less than 250 nm, that behave in the soil system as a colloid. There are over 100 different viruses excreted by man. Hepatitis A and Norwalk agent have been the most infectious waterborne viruses in recent years. Nine percent of the outbreaks of water borne disease reported between 1971 and 1982 could be attributed to these viruses (Craun, 1985).

Natural die-off and adsorption are the two most important means of reducing the number of viruses applied to a ST-SAF. If viruses cannot survive during the travel period between the ST-SAF and a groundwater well, these pathogens will not be a viable problem. Temperature is the most important factor in explaining the survival rate of viruses. Viruses maintained at low temperatures have considerably higher survival rates.

Adsorption is the primary process of attenuation of viruses in soils. Adsorption is governed by attractive and repulsive forces between the virus and soil particle within the diffuse double layer, and van der Waals forces. The most important factors in explaining these forces and virus adsorption are the charge of the soil and virus, pH, IEP, concentration and valence of cations in the soil solution, and the amount of organic colloids in the system. Most viruses have an isoelectric point (IEP) greater than 5, so that reducing the pH increases the amount of virus adsorption. Multivalent cations may act as a bridge between the negatively charged virus and soil particle, and thus enhance adsorption. An increase in the concentration of cations will also increase adsorption by reducing the thickness of the diffuse double layer, and thereby enhancing adsorption. Organic colloids in the system can reduce the amount of adsorption by competing for virus adsorption sites.

Flow rates are the single most important factor governing movement of viruses. Columns constructed from undisturbed soils material, or columns maintained so that saturated flow occurred, showed the most virus movement. Several studies showed virus movement in saturated columns with minimal reduction in the number of viruses. Soils with finer textures showed greater reductions in viruses. Most soils showed nearly complete adsorption of viruses. Sandy soils, or soils with high organic matter however, showed considerably more desorption than soils lacking organic matter and of finer textures. Sandy soils showed as much as 68% desorption of viruses.

The number of field studies, or studies of the fate of viruses in alternative systems, have been minimal. Field studies suggest that in soils conducive to rapid flow, such as soils with coarse textures, or high water tables, movement of viruses can be rapid, resulting in groundwater pollution. Once in the groundwater viruses are mobile and can move considerable distances.

The potential for pollution of the groundwater from OSWDS may be greater for viruses than bacteria. Several studies indicated that survival rates, and degree of movement were greater for viruses than those of fecal bacteria. Therefore, numerous researchers have called for drinking water standards for viruses. Methods for assessing virus pollution in the groundwater are not well developed. Therefor increases in setback distances may be the best immediate approach to assure that pollution from viruses attributed to OSWDS is maintained at a minimum.

Several other components of STE may also be considered as potential pollutants. Those of most concern are heavy metals, cations such as K, Ca, Mg, and Na, the anions Cl^- and SO_4^{2-}, and methylene blue active substances (MBAS). In general, heavy metals in domestic wastewaters are very low. High pH conditions favor metal immobility. In addition, metals readily adsorb to the exchange sites of organic matter or clay particles. Therefore, soils with organic matter, clay, or an elevated pH will attenuate most heavy metals.

Substantial increases in cations or anions can reduce the consumptive or irrigation quality of ground and surface waters. Although not considered a serious problem, with time (especially in areas with fairly saline ground or surface waters), addition of ions from OSWDS may be of some concern.

Methylene blue active substances (MBAS) are anionic surfactants added to domestic detergents. These surfactants can account for 10 to 30% of the detergent's mass. Most detergents switched from the alkylbenzene sulfonate (ABS) to linear alkylbenzene sulfonate (LAS) which readily degrade in the septic tank. Some evidence exists however, suggesting that some of the MBAS were more resistant to degradation than others. These concerns however are of only minor importance.

RESEARCH AND SAMPLING NEEDS

The literature and research indicate that ground and surface waters in Virginia are being contaminated from OSWDS. Several of the studies that extensively sampled ground or surface waters, and directly attributed pollution of these waters to OSWDS, were conducted in states neighboring Virginia with similar soils (Maryland, Delaware, South Carolina). Studies conducted by Reneau et al. (1975) in a small watershed in Virginia indicated that pollution of surface and shallow-ground waters was occurring from OSWDS. Preliminary research examining bacterial pollution in Virginia groundwaters (Alexander, 1991) has suggested extensive bacterial pollution to the groundwater attributable to OSWDS.

The majority of groundwater studies have indicated that biological contaminants (bacteria and viruses) and N are leading pollutants of groundwater attributable to OSWDS. These contaminants are also of primary concern in terms of health aspects. Studies in Virginia have also indicated that these contaminants are of the most concern. Therefore, research efforts should concentrate on the fate and transport of bacteria, viruses, and N in the soil-groundwater system.

RESEARCH NEEDS

Remediation of polluted groundwater is very difficult as well as costly. An assessment is necessary to determine the major aquifers in Virginia in which OSWDS have the greatest potential to pollute. Much of this assessment is possible through previously published information discussing sources of groundwater. The amount of recharge through precipitation and wastewater disposal should be estimated to determine the effective dilution of NO_3^- possible within the aquifers. Research has shown that in properly sited and operating conventional OSWDS, dilution of NO_3^- is the primary means of reduction. Studies addressing these issues may aid in determining allowable densities of OSWDS within a recharge basin, without excessively polluting the groundwater with N. Similar studies have been proposed by Bauman and Schafer (1985).

Research in South Carolina (Sandhu et al., 1979), Delaware (Miller, 1975), and preliminary research in Virginia (Alexander, 1991) has indicated that groundwaters are already polluted with N and biological contaminants from OSWDS. An extensive survey is necessary to determine the current status of the groundwater of Virginia with respect to N, bacteria, and viral concentrations. A survey of this type will provide the necessary information to evaluate the effects OSWDS have had on this valuable resource. Extensive sampling from an aquifer or groundwater source in which most of the residences are served by OSWDS could be compared with an aquifer in which most of the households use a municipal system for wastewater disposal. Similar studies have been conducted by Miller (1975) in Delaware and Sandhu et al. (1979) in South Carolina, examining N and bacterial pollution, respectively. Research indicates that the greatest potential for groundwater pollution is in areas where soils are of coarse texture and shallow to groundwater. Therefore, the coastal plain is the most likely area to begin a groundwater survey program. The most important aquifers in Virginia (those serving the largest number of people) would be another area to consider initial groundwater surveys and research.

A positive correlation between density of OSWDS and declining groundwater quality was presented in numerous papers (Bicki et al., 1984; Morrill and Toler, 1973; Ford et al., 1980; Geraghty and Miller, 1978; Bauman and Schafer, 1985). The cause and effect relationships between OSWDS density and declining water quality should be examined in Virginia. These data would provide another method to estimate the maximum number of OSWDS that can safely operate in a recharge basin without threatening the groundwater resource. In order to examine the relationships between groundwater quality and density of OSWDS, comparisons of the groundwater quality between several small aquifers or groundwater sources on similar soils and landscapes should be conducted. In this survey three aquifers could be examined in which: one aquifer has a high density of OSWDS; one has a low density of OSWDS; and the third aquifer is located in a nearly uninhabited recharge basin.

In most properly sited and operating OSWDS considerable amounts of NO_3^- are produced. In these systems NO_3^- moves easily through the soil and into the groundwater. Nitrate can be easily converted to N gas by the process of denitrification. Denitrification is a process that has potential for reducing the quantity of NO_3^- transported to groundwaters as the result of OSWDS. Several systems with high water tables or alternative technology have shown nearly total reduction in NO_3^- due to denitrification. These results suggest that the various alternative systems for denitrification should be evaluated to determine which is the most cost efficient and easily maintained.

Numerous studies have shown groundwater samples containing essentially no fecal coliforms, but elevated levels of enteric viruses. Methods should be developed to assay viruses in groundwater in a cost effective manner. In addition, standards for enteric viruses in groundwater should be established.

Although a plethora of information exists on the reaction of viruses with soil, minimal field tested research exists. Studies examining the fate and transport of viruses in field situations need to be conducted.

Biological contamination of groundwaters from OSWDS is the most serious concern from a public health standpoint. Research should be conducted and methods developed to reduce the number of bacteria and viruses in domestic wastewater before the wastewater is applied to the ST-SAF. These methods should be applied to OSWDS in which soil treatment of biological contaminants is not sufficient.

Most of the research indicated that soils with high water tables, coarse textures, restrictive layers, or creviced bedrock have the greatest potential for movement of contaminants away from ST-SAF and into groundwater. The minimum separation distance between the ST-SAF trench bottom and a restrictive layer or high water table varies depending upon state and local regulations. Several research studies addressed this issue. In sandy soils, Cogger et al. (1988) concluded that 60 cm was an adequate separation distance if loadings were maintained at a low level and low pressure distribution was used. Similar findings were presented by Sawhney and Hill (1978) for silt loam textured soils. Numerous field and column studies however, have reported chemical and biological movement below 60 cm. Bicki et al. (1984) reviewed much of the literature pertaining to separation distances and potential groundwater contamination. These authors concluded that at least a 90 cm separation distance is necessary for adequate treatment of STE. Field research specifically aimed at examining this problem has been minimal. Studies should be established to examine the separation distance between ST-SAF trench bottoms and high water tables. If possible, these studies should include both bacteria and virus indicator organisms.

The importance of maintaining unsaturated conditions within a given distance of the ST-SAF trench bottom cannot be over emphasized. Estimating the location of the high water table in some soils is fairly difficult. This can be especially difficult in sandy soils. Research should be conducted to increase the knowledge of soil morphology as an indicator of seasonally high water tables.

Considerable research has indicated the importance of macropore flow in structured soils. In Virginia, percolation rates and separation distances are only estimated using soil textural classes. Research addressing the inclusion of structural sizes, grades, and classes into definitions and regulations for separation distances and percolation rates should be initiated.

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