STATE OF FLORIDA
DEPARTMENT OF HEALTH AND REHABILITATIVE SERVICES

ONSITE SEWAGE DISPOSAL SYSTEM RESEARCH IN FLORIDA

An Evaluation of Current OSDS Practices in Florida

March 1993

ONSITE SEWAGE DISPOSAL SYSTEM RESEARCH IN FLORIDA

An Evaluation of Current Onsite Sewage Disposal System (OSDS) Practices in Florida

Report Prepared for:

HRS

State of Florida
Department of Health and Rehabilitative Services
Environmental Health Program

Under HRS Contract No. LP-596
Dr. Kevin M. Sherman, Project Administrator

MARCH 1993

By:

AYRES ASSOCIATES

3901 Coconut Palm Drive, Suite 100
Tampa, Florida 33619

PRINCIPAL INVESTIGATORS:

Richard J. Otis, P.E.
Damann L. Anderson, P.E.
Richard A. Apfel, CPSS

ONSITE SEWAGE DISPOSAL SYSTEM RESEARCH IN FLORIDA

AN EVALUATION OF CURRENT OSDS PRACTICES IN FLORIDA

TABLE OF CONTENTS

Page

LIST OF TABLES

LIST OF FIGURES

ACKNOWLEDGMENTS

The research described herein was supported by the State of Florida under Contract No. LP-596 with the Department of Health and Rehabilitative Services, Environmental Health Program Office. This report is the final report of a series of research efforts conducted as part of the Florida Onsite Sewage Disposal System (OSDS) Research Project, begun in 1986. The funding for this project was generated from the $3 research surcharge on OSDS permits issued from 1983 through 1988, mandated by the Florida Legislature under the Water Quality Assurance Act of 1983.

Valuable input throughout the OSDS Research Project was provided by an Ad-Hoc Advisory Committee made up of interested volunteers from various locations and occupations across Florida. Their effort is gratefully acknowledged:

Randy Brown, Ph.D.
Wayne Crotty
Fred Flanders
Dr. E. Charles Hartwig
Jack Haslam
John Heber
David Heil
Wade Hurt
Senator George Kirkpatrick
Dr. Arthur Lewis
Roy Pence
Patti Sanzone
Joe Schuster
Fritz Sheffield
Kevin M. Sherman, Ph.D.
James R.E. Smith
Dr. Lillian M. Stark
Norman Tuckett
Ellen Vause
Dr. Bernard Yokel

Numerous other individuals have provided information and assistance over the duration of this project. Particular recognition must be given Dr. Arthur Lewis, Dr. Lillian M. Stark, and staff of the Florida HRS Epidemiology Research Center who spent extensive time and resources in performing the virus study. Also, Resource Soil Scientists from the USDA Soil Conservation Service, several OSDS contractors from the Florida Septic Tank Association, county environmental health unit personnel, and staff of the Florida HRS Environmental Health Office in Tallahassee all contributed to the success of the project and are gratefully acknowledged. Mr. Robert Kirkner, P.G., and staff (formerly Kirkner & Associates, Inc.) are acknowledged for their assistance in the early groundwater monitoring and modeling portions of the project.

Finally, the families of the numerous homes which were used as study sites during the research are gratefully acknowledged for their participation and patience in the projects. They allowed access to their property which made many of the study phases possible.

ACKNOWLEDGMENTS

The research described herein was supported by the State of Florida under Contract No. LP-596 with the Department of Health and Rehabilitative Services, Environmental Health Program Office. This report is the final report of a series of research efforts conducted as part of the Florida Onsite Sewage Disposal System (OSDS) Research Project, begun in 1986. The funding for this project was generated from the $3 research surcharge on OSDS permits issued from 1983 through 1988, mandated by the Florida Legislature under the Water Quality Assurance Act of 1983.

Valuable input throughout the OSDS Research Project was provided by an Ad-Hoc Advisory Committee made up of interested volunteers from various locations and occupations across Florida. Their effort is gratefully acknowledged:

Randy Brown, Ph.D.	Roy Pence
Wayne Crotty	Patti Sanzone
Fred Flanders	Joe Schuster
Dr. E. Charles Hartwig	Fritz Sheffield
Jack Haslam	Kevin M. Sherman, Ph.D.
John Heber	James R.E. Smith
David Heil	Dr. Lillian M. Stark
Wade Hurt	Norman Tuckett
Senator George Kirkpatrick	Ellen Vause
Dr. Arthur Lewis	Dr. Bernard Yokel

Numerous other individuals have provided information and assistance over the duration of this project. Particular recognition must be given Dr. Arthur Lewis, Dr. Lillian M. Stark, and staff of the Florida HRS Epidemiology Research Center who spent extensive time and resources in performing the virus study. Also, Resource Soil Scientists from the USDA Soil Conservation Service, several OSDS contractors from the Florida Septic Tank Association, county environmental health unit personnel, and staff of the Florida HRS Environmental Health Office in Tallahassee all contributed to the success of the project and are gratefully acknowledged. Mr. Robert Kirkner, P.G., and staff (formerly Kirkner & Associates, Inc.) are acknowledged for their assistance in the early groundwater monitoring and modeling portions of the project.

Finally, the families of the numerous homes which were used as study sites during the research are gratefully acknowledged for their participation and patience in the projects. They allowed access to their property which made many of the study phases possible.

SECTION 1.0
INTRODUCTION

1.1: BACKGROUND

Onsite sewage disposal systems (OSDS) are used to provide wastewater treatment and disposal where municipal sewerage is not available. Unlike municipal sewerage where central but remote treatment is provided, onsite facilities are constructed on each individual lot for homes or business establishments in an unsewered area. Most OSDS are "septic tank systems" which typically consist of a septic tank followed by a.subsurface wastewater infiltration system (SWIS). Sedimentation and flotation of wastewater solids occur in the septic tank, and some anaerobic digestion of these solids occurs with time. After passing through the septic tank, the partially treated wastewater, referred to as septic tank effluent, (STE), is discharged to the subsurface infiltration system and percolates to the groundwater. As the septic tank effluent enters and percolates through the soil final treatment is accomplished primarily in the unsaturated soil, or vadose zone, through naturally occurring biological, chemical and physical processes. Ultimate disposal occurs when the treated wastewater enters the groundwater below the SWIS.

In Florida, approximately 26% of the population is served by OSDS (U.S. Dept. of Commerce, 1993). According to the Florida Department of Health and Rehabilitative Services (HRS), over 1.7 million systems are estimated to be in use statewide and since 1980, 40,000 to 70,000 new systems have been constructed each year (HRS, unpublished data). The use of OSDS in Florida is regulated by HRS and the HRS county public health units through Chapter 381 of the Florida Statutes and Chapter 10D-6, "Standards for Onsite Sewage Disposal Systems", of the Florida Administrative Code (FAC).

Where properly sited, designed, constructed, and operated, onsite systems are effective and efficient wastewater treatment facilities. However, because of the large number of systems in use, many of which are in high density subdivisions, there are serious concerns that past and present OSDS practices may be having adverse impacts on the water resources of the state, particularly groundwater. Groundwater is the source of 87% of Florida's public drinking water supplies and 94% of its private supplies (Fernald and Patton, 1984). It is important to protect this vital resource to ensure public health, maintain the tourist industry, and provide a desirable quality of life.

Because of these concerns, the Florida Legislature authorized the Florida Onsite Sewage Disposal System Research Project under the Water Quality Assurance Act of 1983 to evaluate OSDS practices in Florida. The goal of this project is to ensure that OSDS practices in Florida protect public health and water resources through application of technically sound guidelines for the management of onsite wastewater treatment systems. To achieve this goal, the research project was divided into three major areas of study:

1. To assess the impacts of OSDS use on groundwater, particularly in locations of high OSDS densities,

2. To evaluate the capabilities of Florida soils to accept and treat wastewater, and

3. To evaluate the suitability of current OSDS design criteria and installation practices in Florida and recommend appropriate improvements to OSDS practices based on the results of the research.

A summary of the research conducted in the first two areas of study can be found in Section 4.7. Details of these studies have been reported previously by Ayres Associates (1987, 1989, 1993a, 1993b). This report presents the results of the third area of study.

1.2: OBJECTIVES AND SCOPE

The objective of this phase of the Florida OSDS Research Project was to evaluate OSDS practices in the state and provide recommendations to improve those practices. The current version of Chapter 10D-6 which was revised in 1991 and became effective March 17, 1992, represents the framework for current practices in the state. Therefore, the evaluation of OSDS practices centered on these adopted requirements. This code was reviewed and evaluated focusing on program requirements, administrative procedures and technical guidelines. The results of this research, published literature, and other available information and experience were used in the evaluation to formulate specific recommendations.

SECTION 2.0
THE NEED FOR ONSITE WASTEWATER TREATMENT SYSTEM MANAGEMENT PROGRAMS

2.1: HISTORICAL REVIEW OF ONSITE SYSTEM CODE DEVELOPMENT

Septic tank systems have been used for wastewater disposal in unsewered areas since the turn of the century. However, it was not until the late 1940's, that their use became widespread. The suburban housing boom that followed World War II outpaced the construction of sewers; consequently septic tank systems were installed in large numbers.

Programs to regulate the installation and use of onsite wastewater systems were not adequate for the increased demand. Little was known about the relation between design and performance, and system siting and design guidelines were vague. Operation and maintenance were left to the homeowner. Without adequate rules and sufficient regulatory control, many systems were installed where conditions were not suitable or where designs were inappropriate for the application. As a result, hydraulic failures became common. With high housing densities in the urban fringe developments, concerns for public health and sanitary nuisance conditions required that onsite wastewater system practices be improved and adequately regulated.

In the 1950's, states began to promulgate improved codes with the intent to provide a rational basis for the design and installation of septic tank systems. The codes, which were enforced by local public health departments, were centered around the "percolation test", and local practices and experiences. Codes were not based on scientific principles, but on empirical relationships and folklore. They were based on several incorrect assumptions:

- The design of systems could be based on a clean water percolation test which ignored the complex interrelationships between soil characteristics and conditions, character of the wastewater, biological mechanisms, and climate,
- A prescribed design could be used for all sites meeting certain minimum requirements,
- Siting, design, and construction could be performed by untrained persons,

- Operation and maintenance of the system could be performed by an uninformed owner,

- Compliance with public health objectives would meet environmental protection requirements, and

- Onsite wastewater systems would be only a temporary stage toward progressive development of central sewerage and, therefore, provisions to proactively manage the systems or to deal with failures when sewers were not available would not be necessary.

Largely because of these flawed assumptions, the success of early codes in regulating system use and preventing system failure has been limited. Subsequent efforts to improve codes have been done largely by revising existing codes. As a result, many of these basic assumptions have been implicitly perpetuated. Much of the reason for this is that the regulation of onsite wastewater systems has occurred within the public health sector by people that have had little training in wastewater engineering. While these regulators have been experienced in public health issues and have successfully protected human health and safety from spread of disease, they have not fully adapted to the changing needs of onsite wastewater system design for increased performance and environmental protection. Thus, codes continue to be prescriptive "rule books" that provide no assurance that environmental or public health goals can be met.

Today, it is generally recognized that past approaches to managing onsite wastewater treatment system use are no longer adequate. Prescriptive codes based on empirical relationships and arbitrary standards that emphasize hydraulic performance rather than treatment are not meeting the demands for environmental protection. Regulatory complacency with system performance after installation cannot continue if treatment goals are to be met.

2.2: THE FUTURE OF ONSITE WASTEWATER TREATMENT SYSTEMS

Because many continue to believe that onsite wastewater treatment systems cannot meet public health and environmental protection goals and are designed only to be interim facilities until sewers are available, units of government have attempted to severely restrict or ban their use. However, conventional sewerage is not economically feasible in most rural areas. Onsite treatment and disposal systems are needed as cost-effective alternatives to provide safe and environmentally sound wastewater treatment in unsewered areas.

In small communities and many urban fringe developments, low housing densities result in prohibitively high costs of sewer construction and operation. The Association of State and Interstate Water Pollution Control Administrators (Buckrop, 1992) estimates that over $10 billion is needed over the next 10 years to satisfy wastewater facility needs in communities with populations of less than 5,000. It is these small communities that generally have the least ability to pay. It is not uncommon for construction costs of conventional sewerage to exceed the total assessed value of such a community. As a result, wastewater facility projects are being delayed or not undertaken. Continuing noncompliance could seriously threaten local public health. In a survey of states by the GAO, 24 of 34 responding states said that unmet wastewater needs in small communities will have significant health and environmental impacts (Hembra, 1992).

The demand for effective but low-cost wastewater facilities in low density developments is great. Onsite systems can fulfill this need, but only if they are recognized as wastewater treatment plants that must be designed and managed by qualified professionals to meet specific treatment standards. If effective regulatory controls are in place which ensure proper management, the use of onsite wastewater treatment systems can be expected to increase substantially.

2.3: ELEMENTS OF AN EFFECTIVE PROGRAM

Onsite wastewater treatment systems are a viable and legitimate alternative to conventional sewerage. The failure of these systems to gain acceptance as effective and permanent facilities is due primarily to shortcomings in management programs. Most programs are developed around minimum standards which define acceptable site characteristics for a prescribed system design. The established standards are conservative and restrictive because few programs require appropriately trained service providers to perform the work. Also, the programs do not retain regulatory control on system performance after installation. Without the assurances of proper implementation and perpetual performance monitoring and enforcement, system failures are expected and routinely occur.

If onsite systems are to provide satisfactory, low-cost wastewater treatment and disposal in unsewered areas, management programs must include five basic elements:

- Clear and specific performance standards
- Technical guidelines for site evaluation, design, construction, and operation
- Perpetual performance monitoring
- Licensing or certification of all service providers
- Effective enforcement mechanisms

Unfortunately, most programs address only some of these elements. Most codes are based on the assumption that if a site meets the minimum requirements and the system design complies with the codified technical specifications, the public health and environment will be protected. After construction is completed, any regulatory control typically is lost unless a sanitary nuisance complaint is filed.

Before onsite wastewater treatment systems can be regarded as effective and permanent facilities, management programs must include each of these elements. Onsite systems must be designed to meet specific performance standards. Each system must be designed to conform to site conditions rather than requiring site conditions conform to criteria established for system design. Once constructed, perpetual monitoring is necessary to ensure systems continue to meet performance standards. Registration of all service providers including: site evaluators, designers, contractors, operators and regulators must be established with minimum qualifications and continuing education requirements. Effective enforcement of the program through licensing, permitting for construction and operation, and assessing fines and penalties are also needed. Program emphasis must shift from a codified "cook book" approach to effective and continuous management.

SECTION 3.0
THE ONSITE WASTEWATER TREATMENT SYSTEM REGULATORY PROGRAM OF FLORIDA

3.1: CHAPTER 10D-6, FAC: "STANDARDS FOR ONSITE SEWAGE DISPOSAL SYSTEMS"

The authority to regulate onsite wastewater treatment systems in Florida has been delegated to the Florida Department of Health and Rehabilitative Services in Chapter 381.0065 of the Florida Statutes. Chapter 10D-6, "Standards for Onsite Sewage Disposal Systems" of the Florida Administrative Code provides rules for system regulation.

The objective of the current onsite wastewater treatment system regulatory program in Florida is to minimize the occurrence of sanitary nuisances and prevent pollution of groundwater and surface waters. This is accomplished through plan review, site evaluation, construction inspection, and registration of system contractors. These activities are to ensure that the minimal installation standards specified in Chapter 10D-6 are met for all new and repaired systems. Further details of the current Florida program are discussed in Sections 5.0 through 8.0 of this report.

The regulation of onsite wastewater treatment systems in Florida began in the 1920's because of the frequent occurrences of water-borne diseases. Contact with untreated sewage was often cited as the cause of diseases such as hepatitis, meningitis, cholera, dysentery and typhoid. The early regulations were general rules regarding construction of pit privies and septic tank systems and sought to prevent contact by requiring that sewage be disposed below the ground surface. Elimination of a public health hazard rather than wastewater treatment was the objective of these regulations.

In the early 1970's, a more comprehensive onsite wastewater treatment system code was developed primarily based on the "Manual of Septic Tank Practice" published by the U.S. Public Health Service. This code remained in effect with only small changes until 1983.

In 1983, the Florida Legislature passed the Water Quality Assurance Act of 1983. A part of this act stipulated that Chapter 381 of the Florida Statutes be rewritten because of concerns for potential adverse impacts on groundwater and surface water quality from high density subdivisions served by onsite septic tank systems.

The act addressed particular technical guidelines in the code. Of greatest significance were the directives that the minimum separation distance between the bottom of the drainfield and the seasonally high groundwater table be increased as well as the horizontal setback distance of the system to surface water. The act stipulated that the separation distance between the infiltrative surface of system and seasonally high groundwater be a minimum of 2 ft. The guidelines of previous statutes allowed septic tank systems to be used where seasonally high groundwater reached an elevation no greater than 36 inches below the ground surface. Burial requirements in the code permitted a minimum separation distance to groundwater of 18 inches, significantly less than the 3 to 4-foot separation required by most other states. Because of the difficulty in determining the elevation of the seasonally high groundwater table and the lack of code enforcement, many systems were probably installed with only 6 to 12 inches of separation to, and in some cases below, the surface elevation of seasonally high groundwater. The minimum horizontal setback distance was increased from 50 to 75 ft. Other significant revisions included:

- Establishment of an advisory review variance board and public health standards to be followed for granting of variances to the rules,
- Promulgation of a special rule for onsite system use in the Florida Keys, and
- Prohibition of the advertisement, sale or use of organic chemical solvents to degrease or de-clog onsite infiltration systems.

Finally, the 1983 revisions established fee surcharges on construction permits. One was to fund an accelerated soil mapping program in the state. The other was a special surcharge established for a five year period to generate funds for onsite system research. The research was "...to determine whether high density installation of systems, installation of systems under certain soil and water table conditions, and current methods of system installation are polluting state ground water..." (Stat. 381.273). This surcharge provided some of the funds for the research presented in this report.

Over the next several years, only minor modifications were made to the regulations. In 1984, utilities were allowed to waive the requirement for mandatory connection of onsite systems to sewers with approval of HRS. In 1985, the horizontal separation distance between onsite treatment systems and public potable wells, which serve residential or non-residential establishments and have a total daily flow of less than 2,000 gallons, was relaxed from 200 to 100 ft. The permit fee surcharge was re-authorized and increased from $3 to $5 in 1988. The research objectives were also modified at the same time to include alternative design and installation methods for improving system performance.

In 1989, major revisions of permitting requirements in Industrial/Manufacturing (I/M) areas were adopted. The major changes included:

1. Issuance of annual operating permits by HRS for any new system constructed after July 5, 1989, or upon any change in ownership or tenancy in existing systems for I/M or equivalent usages. (The annual operating permit gives HRS the authority to require periodic sampling and analyses from within and around the system to ensure that no toxic or hazardous chemical or industrial wastes have been disposed through the system.),

2. Written approval of occupational licenses to ensure that businesses with a likelihood to dispose of toxic, hazardous, or industrial wastes are not allowed to occupy existing buildings with onsite treatment facilities, and

3. Restriction of departmental permitting of any property which was zoned, rezoned, platted or subdivided for I/M or equivalent purposes after July 5, 1989 (*i.e.*, they must develop on a sewer system).

Major revisions of Chapter 381, Florida Statutes, occurred again in 1991. These revisions resulted in the current version of Chapter 10D-6, FAC, which took effect in March 1992. This new version removes, after October 1, 1991, the provision allowing developers of large subdivisions to use septic tank systems under a density formula (*i.e.*, allowing septic systems to be used until 50 percent of the subdivision is built at which time sewer service must be provided for all existing and future homes). This provision was removed because HRS found that it was impossible to enforce such agreements. Another provision in the new revision directs HRS and the Florida Department of Community Affairs to study the issue of "vested" lots; lots platted prior to 1972. These lots do not have to comply with minimum densities and are allowed the prior horizontal setback distance of 50 ft from surface water. Environmental groups and agencies are concerned that with the large number of "vested" lots (estimated at more than 1 million), any changes in program requirements will have minimal impact on water quality improvement in these areas. The agencies were to make recommendations by January 1, 1993. Another change in the 1991 revisions is removal of the prohibition against the department issuing an annual operating permit for properties zoned, rezoned, platted or subdivided for I/M or equivalent purposes after July 5, 1989. Also, a fee not to exceed $5,000 for field evaluation of a representative number of experimental systems has been established to aid in the approval of new onsite treatment system technology. The 1991 revisions allow HRS to permit an underground injection well for effluent disposal on lots in the Florida Keys where setback distances can not be met. Permit fees have been increased in these revisions and a fee has been added to annual operating permits for aerobic systems and for I/M zoned areas.

3.2 PROGRAM ADMINISTRATION

The OSDS regulatory program in Florida is administered through the Department of Health and Rehabilitative Services. HRS is divided into several divisions and subdivisions, and the OSDS program falls under the Florida Deputy Secretary for Health, and Office of Environmental Health, Environmental Health Program located in Tallahassee. HRS also maintains eleven district public health offices in various locations throughout the state. In addition, a county public health unit (CPHU) office is located in each of the 67 Florida counties. The function of district offices is to provide support to the CPHUs.

The CPHUs issue all OSDS permits and provide design review, site evaluation, and construction inspection for OSDS in Florida. Chapter 10D-6 is a minimum state code. All counties must enforce OSDS regulations which are to be at least as strict as 10D-6. Several local governments in Florida have adopted regulations that are more strict than state regulations; state law specifically grants local governments the authority to do this.

Further discussions of various Florida OSDS program elements are included in subsequent sections of this report.

SECTION 4.0
EVALUATION OF ONSITE WASTEWATER TREATMENT SYSTEM PERFORMANCE

4.1: INTRODUCTION

The extent to which OSDS are contributing to pollution of groundwater and surface water in Florida is unknown. An evaluation of performance data is needed if effective performance standards and technical guidelines are to be established as part of a comprehensive regulatory program. Such an evaluation will provide a basis for assessing the potential impacts of current OSDS designs on Florida's water resources and recommending appropriate changes to current technical practices.

The following is a critical review of the scientific literature relevant to OSDS performance. A summary of OSDS performance monitoring in Florida that was conducted during previous phases of this project is also reviewed. Both hydraulic and treatment performance are evaluated.

4.2: DESCRIPTION OF CONVENTIONAL ONSITE SYSTEM OPERATION

Conventional onsite wastewater treatment systems typically consist of a septic tank and a subsurface wastewater infiltration system (SWIS). Wastewater flows from the home, through the septic tank and into the SWIS where it infiltrates the soil and percolates to groundwater. The septic tank provides primary treatment of the wastewater which removes the majority of the settleable solids, grease, and other floatable solids which could clog the infiltrative surface and cause hydraulic failure of the system. Anaerobic digestion of retained solids also occurs in the tank. Soil below the SWIS provides physical, chemical, and biological treatment of septic tank effluent as it percolates to groundwater.

The infiltration system is the most critical component of a septic tank system. It provides most of the treatment and utimate disposal of the wastewater. Several different SWIS designs have been developed for use under various site and soil conditions, but all consist of buried soil infiltrative surfaces (Figure 4.2.1). The infiltrative surfaces are the bottoms and sides of excavations constructed in natural soil or imported fill materials. These excavations may be in the form of trenches or rectangular beds. Porous media, typically gravel, is placed in the excavation and around perforated piping which runs the length of

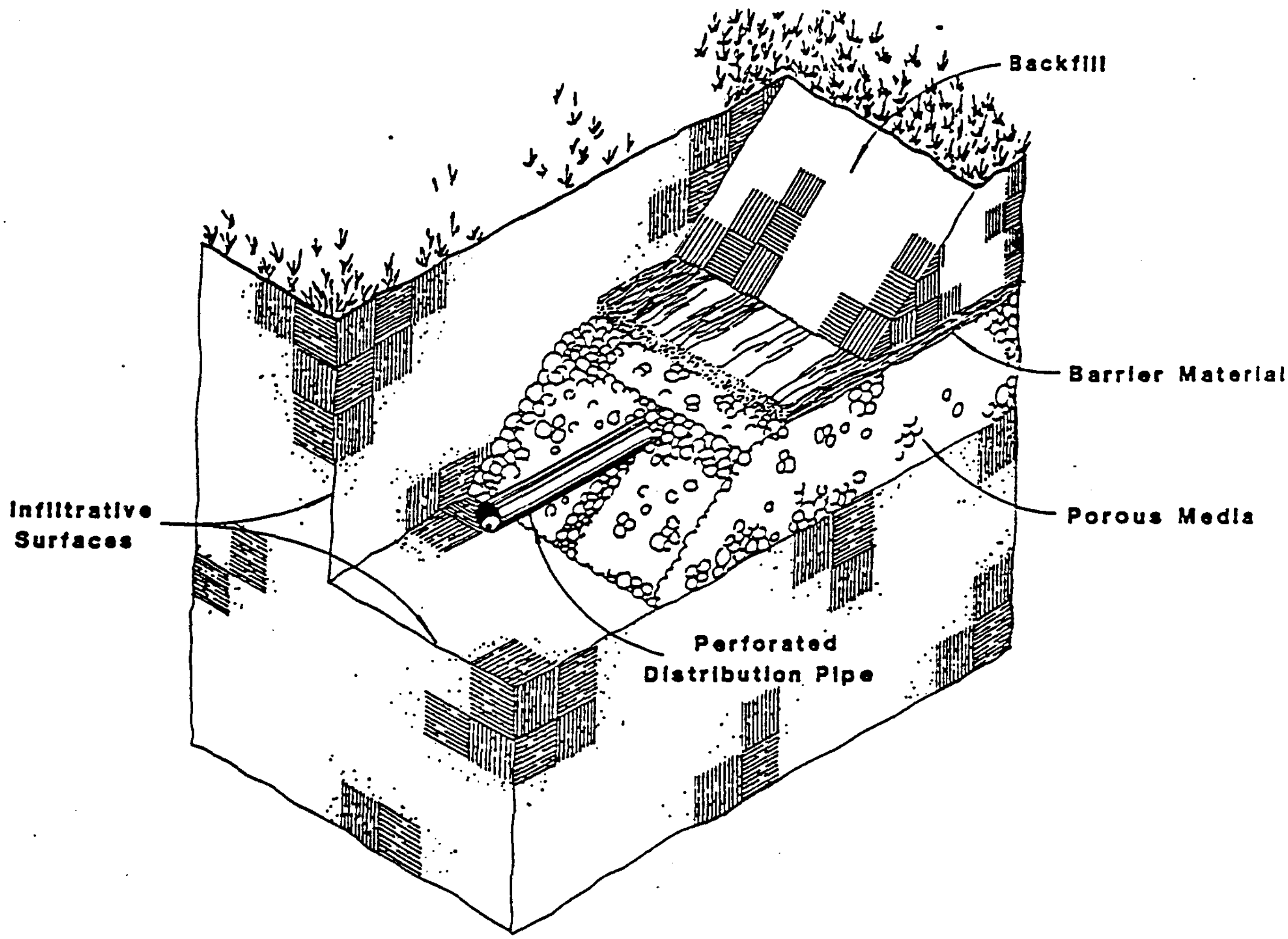

Figure 4.2.1: Typical SWIS Construction

the excavation. The piping is installed to convey wastewater from the septic tank to the SWIS. The porous media maintains the structure of the excavation, allows the free flow of wastewater over the infiltrative surfaces and provides voids for storage of wastewater during peak flows. After construction, the excavation is covered with a porous barrier material and soil backfill.

Fluid transport from SWIS typically occurs through three zones: 1) infiltration zone; 2) vadose (unsaturated) zone; and 3) saturated zone (Figure 4.2.2). Wastewater enters the soil at the surface of the infiltrative zone. This zone is a biologically active zone, usually only a few inches thick. Most of the physical, chemical, and biological treatment of the septic tank effluent occurs in this zone. Many of the particulate materials accumulate on the infiltrative surface and within pores of the soil matrix in this zone, and provide a source of food and nutrients for the active biomass. The biomass and metabolic by-products also accumulate in this zone. Blockage or filling of soil pores by accumulated solids, microbiological growths and by-products may occur which dramatically reduces the hydraulic conductivity of the soil in this zone. As a result, the infiltrative zone becomes a transitional zone where fluid flow changes from saturated to unsaturated flow.

Below the zone of infiltration, fluid enters the unsaturated, or vadose zone. Here fluid is under a negative pressure potential (less than atmospheric) resulting from capillary and adsorptive forces of the soil matrix. Consequently, fluid flow occurs over the surfaces of soil particles and through finer pores of the soil while larger pores remain gas filled (usually air under atmospheric pressure). Wastewater contact with solid surfaces of the soil is enhanced. Fluid transport in this zone occurs in response to the total gravity and pressure potentials, and is primarily downward.

From the vadose zone fluid passes through the capillary fringe and enters the saturated zone. In this zone, all soil pores are filled with fluid and flow occurs either vertically and/or horizontally under a positive pressure gradient. It is in this zone that fluid ultimately leaves the site. Mixing of wastewater with groundwater is somewhat limited because groundwater flow is typically laminar. Because of this, treated wastewater can remain as a distinct plume for some distance from its source (LeBlanc, 1982; Anderson et al., 1988; Ayres Associates, 1993a; Robertson et al., 1989, 1990; Shaw, 1991). The plume may descend into the groundwater as it travels from the source due to recharge from above as a result of precipitation. Dispersion also occurs, but mobility of solutes within the plume varies with soil-solute reactivity.

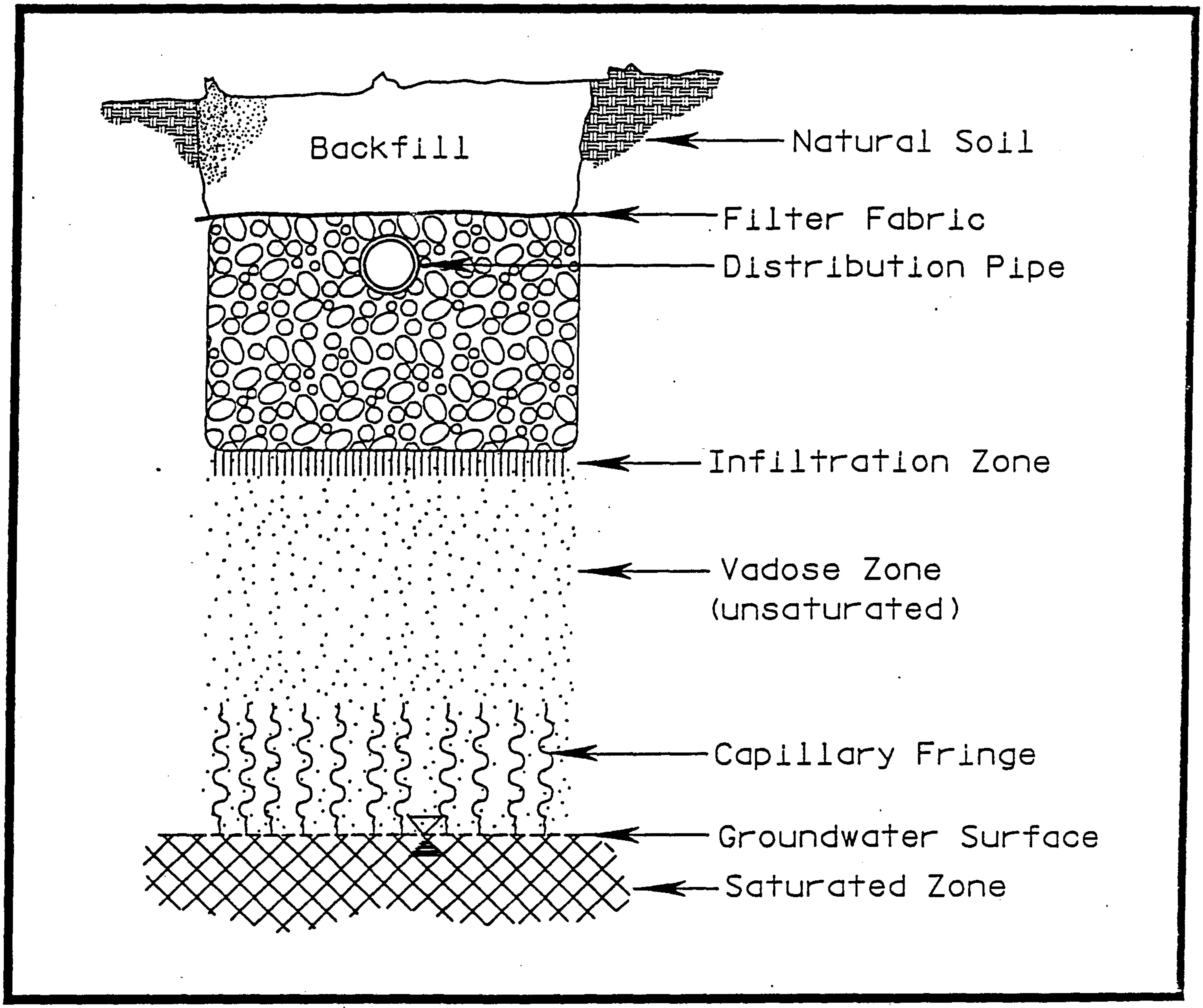

Figure 4.2.2: Fluid Transport Zones in SWIS

The performance of conventional septic tank systems should be measured by the ability of the system to accept and adequately treat applied wastewater loads within a defined boundary. The boundary is frequently defined to include unsaturated soil below the infiltrative surface of the SWIS down to the permanent water table and a portion of the saturated zone horizontally to the property line. Typically, drinking water standards must be met before the wastewater/groundwater mixture crosses this boundary. Therefore, the SWIS provides the majority of the required treatment.

The use of SWIS for wastewater treatment is limited by characteristics of the selected treatment site. The "soil is the system". Therefore, SWIS performance is difficult to predict and monitor since each site is unique. Successful performance of septic tank systems is achieved only if the soil below the SWIS accepts all wastewater it receives and provides sufficient final treatment before reaching groundwater. If failure should occur, significant environmental damage or health risks can result. Hydraulic failures, caused by excessive clogging of the infiltration zone or insufficient infiltrative surface area, can lead to wastewater backups in the building, or wastewater ponding on the ground surface and runoff from the treatment site into surface waters. Inadequate treatment by the soil matrix can result in contamination of groundwater and ultimately surface water through groundwater discharge. Therefore, the selection and design of SWIS for wastewater treatment must be based on a thorough site evaluation and understanding of the interactions between applied wastewater, soil, and hydrogeology of the selected site.

4.3: TYPICAL WASTEWATER CHARACTERISTICS

4.3.1: Composition of Domestic Wastewater

Wastewater discharged from single family homes is comprised of a number of individual wastewaters generated from various water using activities. These activities typically include toilet flushing, bathing, clothes and dish washing, cleaning activities, and in some instances, garbage disposal and water conditioning brines. The separate contributions of water using activities that result in discharge to wastewater systems have been extensively studied by several researchers (Cohen and Wallman, 1974; Ligman et al., 1974; Bennett and Linstedt, 1975; Laak, 1975; Siegrist et al., 1976). Table 4.3.1 summarizes the results of those investigations which were primarily determined from measured interior water use in rural homes.

TABLE 4.3.1: RESIDENTIAL WATER USE BY ACTIVITY[a,b]

Activity	Gal/Use	Uses/Cap/Day	gpcd[c]	% Total
Toilet Flushing	4.3 4.0 - 5.0	3.5 2.3 - 4.1	16.2 9.2 - 20.0	35
Bathing	24.5 21.4 - 27.2	0.43 0.32 - 0.50	9.2 6.3 - 12.5	20
Clothes Washing	37.4 33.5 - 40.0	0.29 0.25 - 0.31	10.0 7.4 - 11.6	22
Dish Washing	8.8 7.0 - 12.5	0.35 0.15 - 0.50	3.2 1.1 - 4.9	7
Garbage Grinding	2.0 2.0 - 2.1	0.58 0.4 - 0.75	1.2 0.8 - 1.5	2
Miscellaneous	-	-	6.6 5.7 - 8.0	14
Total	-	-	45.6 41.4 - 52.0	100

a Adapted from U.S. EPA (1980).

b Means and ranges are of results reported in Cohen and Wallman (1974), Ligman et al. (1974), Bennett and Linstedt (1975), Laak (1975), and Siegrist (1976).

c gpcd may not equal gal/use multiplied by uses/cap/day due to difference in the number of study averages used to compute each mean and range.

The average daily wastewater flow from a typical residence is approximately 45 gal/capita/day (gpcd), based on several studies (Table 4.3.2). Wastewater volumes can be expected to vary substantially from residence to residence as a result of many factors including family size, age distribution of the occupants, socioeconomic status, and type of water using fixtures in the residence. Average flows at individual residences are typically no greater than 60 gpcd and seldom exceed 75 gpcd. Low volume discharge water fixtures can be used to reduce the average daily wastewater flow. Such fixtures have been shown to reduce total wastewater volume by 10 to 30% (Siegrist, 1983; Siegrist et al., 1978; Siegrist et al., 1981; Anderson and Siegrist, 1989; Anderson and Konen, 1993).

TABLE 4.3.2: SUMMARY OF AVERAGE DAILY RESIDENTIAL WASTEWATER FLOWS[a]

Study	No. Residences	Study Duration (months)	Study Average (gpcd)	Study Range (gpcd)
Linaweaver, et al. (1967)	22	-	49	36 - 66
Anderson & Watson (1967)	18	4	44	18 - 69
Watson et al. (1967)	3	2 -12	53	25 - 65
Cohen & Wallman (1974)	8	6	52	38 - 102
Laak (1975)	5	24	41.4	26 - 65
Bennett & Linstedt (1975)	5	0.5	44.5	32 - 83
Siegrist et al. (1976)	11	1	42.6	25 - 57
Otis (1978)	21	12	36	8 - 71
Duffy et al. (1978)	16	12	42.3	-
Aher et al. (1981)	25	3	39.4	22.7 - 59.7
Anderson & Konen (1993)	25	3	50.7	26.1 - 85.2

a Adapted in part from U.S. EPA (1980). Based on interior water use monitoring and not wastewater flow monitoring.

Since wastewater is generated by discrete water use events within the home, wastewater flow and quality vary widely during the day. A typical residential wastewater hydrograph is illustrated in Figure 4.3.1. Table 4.3.3 provides figures for mass loading of selected wastewater constituents from various water use activities. Ranges of computed total mass loadings and observed constituent concentrations are presented in Table 4.3.4.

4.3.2: Composition of Domestic Septic Tank Effluent

The septic tank provides partial treatment of raw wastewater. The primary removal mechanism is sedimentation and flotation of suspended solids. Anaerobic digestion of the retained solids occurs within the tank converting some of the solids into soluble forms and allowing them to escape with tank effluent. The quality of septic tank effluent can vary substantially, but various studies have shown domestic septic tank effluent to vary within typical ranges (Table 4.3.5).

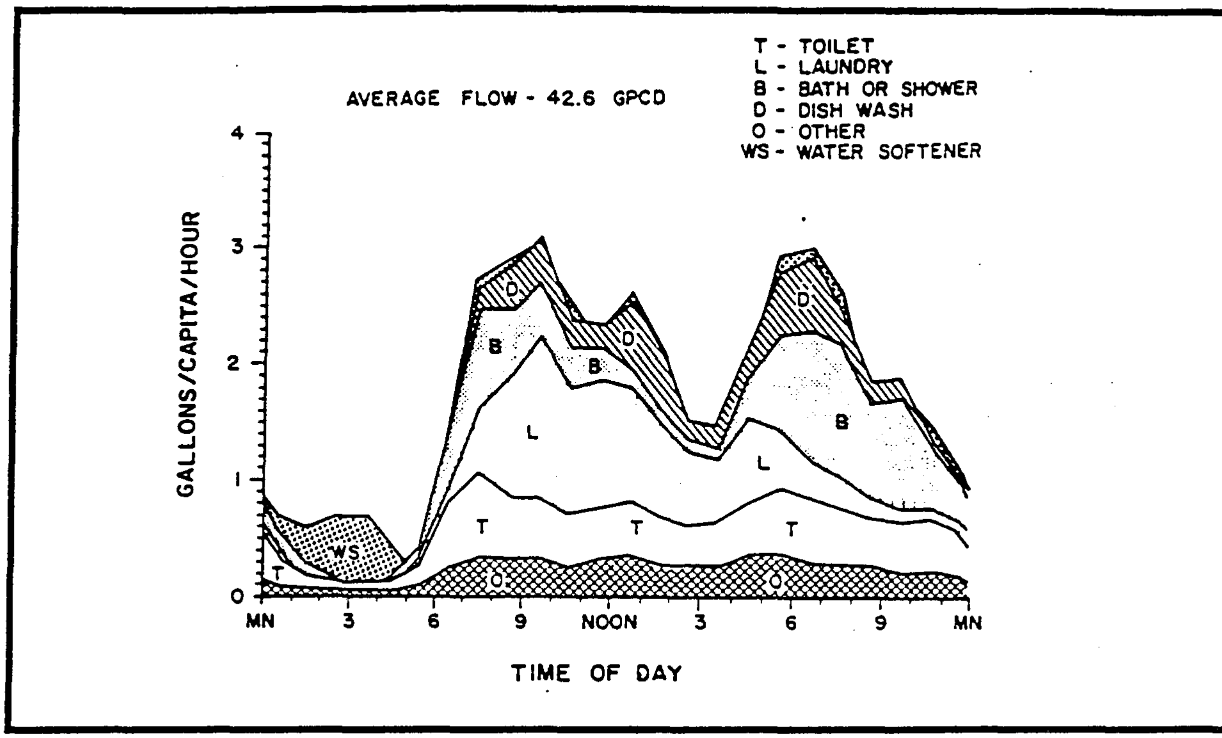

Figure 4.3.1: **Typical Household Wastewater Hydrograph (University of Wisconsin, 1978)**

TABLE 4.3.3: CONTRIBUTION OF MAJOR DOMESTIC WASTEWATER SOURCES TO CONSTITUENT LOADINGS[a,b]

Parameter	Garbage Disposal	Toilet	Basins, Sinks, Appliances	Approximate Total
	(gm/cap/day)			
BOD_5	18.0 10.9 - 30.9	16.7 6.9 - 23.6	28.5 24.5 - 38.8	63.2
Suspended Solids	26.5 15.8 - 43.6	27.0 12.5 - 36.5	17.2 10.8 - 22.6	70.7
Nitrogen	0.6 0.2 - 0.9	8.7 4.1 - 16.8	1.9 1.1 - 2.0	11.2
Phosphorus	0.1 0.1 - 0.1	1.2 0.6 - 1.6	2.8 2.2 - 3.4	4.0

a After U.S. EPA (1980).

b Means and ranges are of results reported in Olsson et al. (1968), Cohen and Wallman (1974), Ligman et al. (1974), Bennett and Linstedt (1975), Laak (1975), and Siegrist et al. (1976).

TABLE 4.3.4: CONSTITUENTS OF TYPICAL DOMESTIC WASTEWATER[a]

Constituent	Mass Loading (gm/cap/day)	Concentration[b] (mg/l)
Total Solids	115 - 170	680 - 1,000
Volatile Solids	65 - 85	380 - 500
Suspended Solids	35 - 50	200 - 500
Volatile Suspended Solids	25 - 40	150 - 240
BOD_5	35 - 50	200 - 290
Chemical Oxygen Demand	115 - 125	680 - 730
Total Nitrogen	6 - 17	35 - 100
Ammonia	1 - 3	6 - 18
Nitrites and Nitrates	<1	<1
Total Phosphorus	3 - 5	18 - 29
Phosphate	1 - 4	6 - 24
Chlorides	9 - 13	50 - 75
Metals	-	NA[c]
Volatile Organic Compounds	0.02 - 0.07	0.1 - 0.4
Surfactants	3.2	19
Total Coliforms[d]	-	$10^{10} - 10^{12}$
Fecal Coliforms[d]	-	$10^{8} - 10^{10}$

a For typical residential dwellings equipped with standard water-using fixtures and appliances (excluding garbage disposals). Based on the results presented in Bennett and Linstedt (1975), Laak (1975), Laak (1986), Siegrist et al. (1976), Bauer et al. (1979), and Tchobonoglous and Burton (1991).

b Assumed water use of 45 gpcd (170 lpcd).

c Not available.

d Concentrations presented in Most Probable Number per 100 ml.

TABLE 4.3.5: CONCENTRATIONS OF SELECTED WASTEWATER PARAMETERS IN SEPTIC TANK EFFLUENT[a]

Parameter (Units)	Weibel et al. (1949)	Univ. Wis. (1978)	Harkin et al. (1979)	Ronayne et al. 1982
Location	Ohio	Wisconsin	Wisconsin	Oregon
No. Tanks	10	7	33	8
Temp	-	-	13	-
(^{0}C)	-	-	0-23	-
	-		141	-
BOD_5	138	138	132	217
(mg/L)	64-256	7-480	-	-
	44	150	145	70
TSS	155	49	87	146
(mg/L)	43-485	10-695	-	-
	55	148	164	70
TKN	-	45	82[b]	57.1
(mg-N/L)	-	9-125	-	-
	-	99	127	57
NO_2+NO_3	-	0.4	0.95	0.42
(mg-N/L)	-	0.1-74	-	-
	-	114	215	59
Total P	-	13	21.8	-
(mg-P/L)	-	0.7-90	-	-
	-	99	215	-
Chloride	-	-	164	-
(mg/L)	-	-	-	-
	-	-	215	-
FOG	-	-	-	-
(mg/L)	-	-	-	-
	-	-	-	-
MBAS	-	-	-	-
(mg/L)	-	-	-	-
	-	-	-	-
F. coliforms	-	7.7	6.8	6.4
(Log#/L)	-	3.0-9.2	-	-
	-	151	205	56

a Data shown for each parameter correspond to average, range, and number of samples
b Total Nitrogen, not TKN

4.3.3: Composition of Commercial Wastewater and Septic Tank Effluent

Commercial wastewater can vary significantly from domestic wastewater. The wastewater varies with the type and use of the establishment. Unfortunately, few data exist to provide reasonable projections of wastewater characteristics from each type of establishment. Some data exist for raw recreational vehicle wastewater, highway rest area septic tank effluent, and restaurant septic tank effluent. These data are presented in tables 4.3.6, 4.3.7 and 4.3.8. As these data show, the characteristics of some wastewaters that are treated by onsite systems can be considerably stronger than domestic wastewater.

TABLE 4.3.6: CHARACTERISTICS OF RAW RECREATIONAL VEHICLE WASTEWATER

Parameter	Units	Pearson et al.[a] (1980)	Kiernan et al.[b] (1983)
COD	mg/L	6,209	8,230
BOD_5	mg/L	3,080	3,110
Organic N	mg-N/L	202	-
Ammonium-N	mg-N/L	767	-
Phosphate	mg-P/L	114	-
pH	-	7.4	-
Total Solids	mg/L	6,460	-
Volatile Solids	mg/L	4,353	-
TSS	mg/L	3,847	3,120
VSS	mg/L	3,329	2,460
Oil & Grease	mg/L	189	-
Formaldehyde	mg/L	18	170
Phenol	mg/L	0.5	-

a Study of 9 sanitary dump stations in California during 1978-1979
b Study of 3 sanitary dump stations in Washington during 1981

TABLE 4.3.7: CHARACTERISTICS OF HIGHWAY REST AREA SEPTIC TANK EFFLUENT (Ayres Associates, 1991)

Parameter	Units	Conventional Toilets	Ultra Low Volume Flush Toilets
COD	mg/L	2,470	3,190
BOD_5	mg/L	560	695
TKN	mg-N/L	207	302
NH_4	mg-N/L	157	216
NO_2&NO_3	mg-N/L	2.98	3.31
TSS	mg/L	1,740	606
VSS	mg/L	1,300	195

TABLE 4.3.8: CHARACTERISTICS OF RESTAURANT SEPTIC TANK EFFLUENT (Siegrist et al., 1984)

Parameter	Units	Value
COD	mg/L	1,027
BOD_5	mg/L	506
TKN	mg-N/L	66
Phosphorus	mg-P/L	20
TSS	mg/L	177
Oil & Grease	mg/L	83

4.4: HYDRAULIC PERFORMANCE

Hydraulic performance of a SWIS is measured by its ability to accept all the wastewater received over the design life of the system. The capacity of an infiltration system to accept wastewater can change due to soil clogging which develops as repetitive applications of wastewater occur. Excessive clogging can reduce the rate of infiltration below the wastewater application rate and cause hydraulic failure. Many studies have investigated soil clogging to better understand the process and to find ways to prevent excessive clogging. Only a brief review of these studies, adapted largely from Otis (1985), is provided here.

4.4.1: Process of Soil Clogging

4.4.1.1: Phases of Clogging

The permeability of soil is a function of the number, size, and continuity of soil pores. If water cannot freely enter or leave these pores, its movement through the soil will be slow. Processes which restrict the openings or reduce the size of soil pores will reduce permeability of the soil. Compaction and smearing of the infiltrative surface during construction or deposition of suspended solids entering the system with the wastewater can seal the entrances to the pores. Gases produced from biological activity in the soil or air trapped below the wetting front can prevent liquid from entering the pores. Soil swelling from prolonged wetting can close the pores. Biological activity stimulated by the carbonaceous material and nutrients in the wastewater, can degrade soil structure resulting in the reduction of macropores. The biomass and metabolic by-products produced by microbial activity can fill or reduce the size of the pores. All of these processes probably occur to some degree in subsurface infiltration systems.

Soils in which construction damage is not significant, clogging has been described to occur in three or four phases (Allison, 1947; Jones and Taylor, 1965; Thomas et al., 1966; Okubo and Matsumoto, 1979). However, the definitions of these phases differ between investigators.

Allison (1947) found that infiltration rates of groundwater recharge basins receiving river water initially decline and then increase before gradually declining to a small fraction of the initial infiltration rate. The initial decline of the first phase was attributed to structural changes in the soil resulting from swelling and dispersion of clay minerals. The gradual increase in the second phase was explained as the result of dissolution of entrapped air in the soil profile. In the third phase, the rates decreased rapidly at first and later more slowly. Based on a laboratory investigation of this phenomenon, in which sterilized and unsterilized water was applied to sterilized and unsterilized soil columns, Allison concluded that biological activity was responsible for the loss in permeability due to clogging resulting from the production of active biomass and metabolic by-products such as slimes and polysaccharides.

Jones and Taylor (1965), applied septic tank effluent intermittently to sand columns and also observed a three phase reduction in infiltration rates, but in a different pattern. When wastewater was dosed daily on the columns, the rate of decline was directly proportional to the volume of infiltrated wastewater. During this first phase, the cause of infiltration rate decline was thought to be due to the accumulation of particulate organic materials. In the second phase, the decline of infiltration rates proceeded more slowly, apparently due to a quasi-equilibrium state which was reached between organic decomposition and new solids accumulations. The infiltration rates declined rapidly during the third phase and stabilized at approximately 0.5 to 1.0 percent of the original rates. This decline appeared to be independent of effluent loading or initial rate of infiltration. In columns that were continuously ponded, the second phase was of short duration or was absent. Infiltration rates decayed rapidly to a small fraction of the initial rates, without passing through the intermediate phases.

Thomas et al. (1966) intermittently applied septic tank effluent to columns of sand. They also found three distinct phases of hydraulic behavior, but the way in which the infiltration rates declined in response to daily dosing of septic tank effluent was unique. During the first phase, the rates of infiltration declined slowly over an extended period of time. The second phase was short during which time the infiltration rates declined sharply and continuous ponding of the infiltrative surface occurred. In the third phase, the infiltration rates asymptotically approached a lower limit. It was noted that the change from the first to second phase coincided with a shift from an aerobic to an anaerobic soil atmosphere below the infiltrative surface. The total organic matter present also seemed to be indirectly related to infiltration rates.

Okubo and Matsumoto (1979) defined four phases of soil clogging from their experiments with application of synthetic wastewater to columns of medium sand. The columns were continuously inundated with a prepared wastewater containing glucose as the only carbon source. Ammonium chloride was added to produce a carbon to nitrogen ratio of 1.44. Other micronutrients were also added. In the first phase, infiltration rates decreased rapidly and were followed by almost constant and sometimes increasing rates in the second phase. The third phase showed rapid rates of decline. Finally in the fourth phase, the infiltration rates slowly decreased to a fraction of the initial rate. During the third phase, a change from aerobic to anaerobic conditions was observed in the soil gas.

Differences between the observations of these four studies can be explained by differences in experimental methods, applied water quality, method of liquid application, and soil materials used. However, it is the similarities which are important. Each of the studies showed a slow decline in infiltration rates which asymptotically approached a rate, a fraction of the initial rate. At some time during this decline, rates decreased rapidly. This period of rapid decline is likely the result of significant chemical or biological change within the soil below the infiltrative surface due to the cumulative effects of applied wastewater.

4.4.1.2: Mechanisms of Clogging

McCalla (1945, 1946, 1950) performed a series of studies from which he concluded that microorganisms are the cause of reduced water percolation through soils. In one study, three sets of columns were prepared. Each set contained three columns, one each of Sharpsburg silty clay loam, Peorian loess, and Hisperia sandy loam (McCalla, 1950). One set received distilled water only, another was covered with a cotton gin waste mulch before distilled water was applied, and the third had mercuric chloride added to the water to act as a disinfectant. All were continuously ponded. Dramatic decreases in the rates of water infiltration resulted in the first two sets, but the columns to which mercuric chloride was added maintained infiltration rates near the initial rates. McCalla concluded that microorganisms caused the reduced water percolation either by producing gases or organic materials, such as slimes, which interfere with water movement, or by decomposing or changing the binding agents responsible for stabilizing soil structure.

Allison (1947) also concluded that soil clogging is largely a result of microbial activity. Columns of three soils, Hanford loam, Exeter sand loam, and Hesperia sandy loam, were prepared and sterilized. Half of the columns were re-inoculated with microorganisms after sterilization. Sterilized tap water was ponded continuously over the columns. Only the sterilized columns receiving the sterilized tap water remained at maximum permeability throughout the test. Both the control soil and the re-inoculated soil clogged readily.

Gupta and Swartzendruber (1962) confirmed the results of McCalla and Allison, but went on to show that clogging occurs near the infiltrative surface. Boiled deionized water with and without phenol was injected into the bottom of columns filled with clean Ottawa Testing Sand. Piezometers showed that within one day head losses through the first centimeter of the columns injected with the phenol-free water were substantial and increased with time. The remainder of the columns changed little. When phenol was added or the temperature reduced from 23 to 1.5°C, little headloss was observed. Bacterial counts taken from increments throughout the column showed maximum numbers at the inlet. However, it was concluded that cell mass alone could not account for the reductions in permeability observed. By-products of microbial metabolism were suggested, but not identified.

The column studies conducted by Jones and Taylor (1965) and Thomas et al. (1966) also showed clogging to be primarily a surface phenomenon. In both studies, septic tank effluent was applied to columns of sand. Jones and Taylor placed gravel on the sand surface to simulate subsurface infiltration system construction. A zone of low conductivity developed at or just below the gravel/sand interface. Incremental analyses for organic and inorganic materials with depth in the gravel and sand columns revealed two distinct zones of accumulation: the gravel/sand interface region, and the top few centimeters of the gravel. The lower ratio of organic to inorganic materials in the gravel as compared to the interface region suggested that biological degradation was more rapid in the ground due to better aeration. No gravel was used by Thomas and his co-investigators, but the results were similar. Impedance measurements showed that the top centimeter of the sand accounted for 87 percent of the total impedance measured across the column. Chemical analyses further showed that organic materials including polyuronides and polysaccharides accumulated in this zone. The organic matter seemed to be the dominate agent of clogging because recovery of the infiltration rate by drying was accompanied by a decrease in the concentration of the total organic matter in the soil.

To isolate the agents of clogging, studies were undertaken by Mitchell and Nevo (1964), Avnimelech and Nevo (1964) and Nevo and Mitchell (1967) using synthetic wastewaters applied to columns of coarse dune sand. Mitchell and Nevo (1964) were able to correlate the amount of polysaccharides, particularly polyuronides directly with clogging of the sand. These by-products of microbial metabolism are known to enhance soil aggregation by binding soil particles together (Martin, 1946). In a later study, Nevo and Mitchell (1967) found that polysaccharides can be formed at very low redox potentials while degradation only occurs at positive potentials.

In other studies, Bliss and Johnson (1952) and Johnson (1957) investigated methods to maintain high infiltration rates in soils under prolonged submergence in percolation ponds used for groundwater replenishment. They found that infiltration rates were improved when organic materials were added to the soil, but only after a period of preliminary decomposition or "incubation" and air drying. During the incubation period, the infiltration rates decreased, but during the reapplication of water following the drying period, infiltration rates increased substantially. With time, rates once again decreased to low levels at which time incorporation of fresh organic materials became necessary. This phenomenon was described as a three phase process consisting of an incubation period, post-drying phase, and final infiltration rate decline (Bliss and Johnson, 1952). During the incubation period, microbial activity increases when water with the incorporated organic matter is applied to the soil. Large numbers of slimes, gums, gases and other by-products are produced which seal the soil pores. At this point, drying of the soil is necessary, or no increase in rates will follow. During the drying phase, the microbes and by-products contract, forming a somewhat water stable coating, and pull the soil particles together into aggregates. Soil treated in this manner can show large increases in total and non-capillary porosity. With the reapplication of liquid, enlarged and more stable pores permit more rapid infiltration. Eventually, the binding agents are also decomposed and the aggregates break down to cause a dramatic decline in infiltration rates.

In another study, the type and amount of wastewater solids applied to the soil were confirmed to be factors affecting the rate of clogging (Siegrist, 1987). Also, the infiltration rate response patterns observed paralled the previously observed three-phase soil clogging process. In this study, gravel-filled, cylindrical, field test-cells constructed in native silty clay loam subsoil were hydraulically loaded for several years with one of three different domestic wastewater types: tap water, grey water septic tank effluent, or conventional septic tank effluent. Undisturbed soil samples and cores were collected from the cells at selected depths for the purpose of characterizing a wide variety of physico-chemical, morphological, and micromorphological soil properties as they related to clogging. The clogged infiltrative surface zones observed in this study were found to have elevated water contents and organic matter accumulations at and immediately beneath the soil infiltrative surface. In all cases, the matrix organic C contents measured were <0.074 kg/kg. Organic materials concentrated near the infiltrative surface were effective in blocking and filling soil pores, thereby reducing native soil infiltration rates. All soil cells that experienced clogging exhibited a variable-length initial period of operation characterized by infiltration rates which gradually declined from near-initial levels. Subsequently, there were substantial and steady declines. Long-term infiltration rates approached zero as intermittent and then continuous ponding of the soil infiltrative surface ensued and grew in magnitude.

The mechanisms responsible for soil clogging development were not elucidated, but Siegrist concluded that soil clogging may have been caused by processes similar to humus development in native soils. Humus is known to form in the soil from a wide variety of precursors including readily degradable organic compounds (Stevenson, 1985). Conditions that favor humus development include cool temperatures, high humidity, and restricted aeration with an influx of organic materials and nutrients. Wastewater infiltration into subsoils, such that the water, organic, and nutrient loading rates greatly exceed native loading rates, would be expected to stimulate the accumulation of residual materials and synthesized humus at the infiltrative surface.

4.4.1.3: Control of Clogging

The investigations of soil clogging suggest that to maintain infiltration rates in SWIS, the organic and suspended solid loadings to the soil are as important as the hydraulic loading. The rate of organic decomposition within the infiltration system must be equal to or greater than the rate of organic applications. Anaerobic soil gas conditions should also be avoided because organic decomposition is slowed and the microbial by-products produced under these conditions seem to promote clogging. Therefore, suspended solid loadings should be controlled such that soil pores are not mechanically sealed by the solids, which could lead to continuous ponding and the exclusion of oxygen to the infiltrative surface. Where low soil temperatures are encountered, organic loadings should be reduced due to slower microbial activity.

Control of clogging in aggregated soil appears to require more management than is necessary in sands. Prolonged submergence apparently leads to the breakdown of soil aggregates due to microbial degradation of the binding agents. Periodic drying of the infiltrative surface may be necessary to re-form these aggregates and reestablish soil permeability.

4.4.2: Soil Clogging Control Measures in Infiltration Systems

4.4.2.1: Methods of Control

Control of soil clogging in subsurface infiltration systems can be accomplished, in part, by design. Important design factors include hydraulic, organic, and solid loading rates, infiltrative surface geometry and depth, wastewater application method, pretreatment, and multiple cells for infiltrative surface resting.

4.4.2.2: Wastewater Loading on the Infiltrative Surface

Some clogging of the infiltrative surface will always occur in subsurface systems. This is desirable for effective waste treatment because it slows infiltration and forces the water into the finer soil pores enhancing filtration and soil/liquid contact. However, there has been a consensus reached by previous investigators that an equilibrium infiltration rate through the clogged zone can be achieved if the hydraulic application rates are controlled. The rate of infiltration is dependent on the resistance of the clogged zone and the hydraulic gradient across this zone. Equilibrium infiltration rates have been estimated by several investigators (USPHS, 1967; Healy and Laak, 1973; Bouma, 1975). Generally, the accepted hydraulic rates have been correlated to percolation rates or soil texture.

Correlating design infiltration rates only to percolation rates or soil texture is inappropriate because several very important factors which can affect the rates are not considered. First, texture or percolation rates alone do not adequately describe the hydraulic characteristics of soils. Structure, clay mineralogy, bulk density, and soil moisture have profound effects on water movement. Second, the characteristics of wastewater are ignored. Most recommended rates were established from the study of systems receiving only domestic residential septic tank effluent and, therefore, should not be applied to higher strength wastewater such as restaurant wastes (Siegrist et al., 1985). Third, the presence of oxygen at the infiltrative surface is important in maintaining an equilibrium condition. The rate of oxygen reaching soil below the infiltrative surface must equal the rate of applied oxygen demand of the wastewater. Therefore, organic and solids loading rates to the infiltrative surface must also be considered. System geometry, depth to the infiltrative surface, and methods of wastewater application can all be important factors in selection of an appropriate wastewater loading rate because of their influence on maintaining an adequate oxygen flux below the system. Mahuta (1991) showed a direct relationship between oxygen concentration below SWIS and system width, unsaturated zone thickness, and wastewater strength. Finally, seasonal, as well as climatic differences in soil temperatures will affect the development of clogging (Kropf et al., 1975; University of Wisconsin, 1978; Simons and Magdoff, 1979).

4.4.2.3: Geometry and Depth of the Infiltrative Surface

The geometry and depth of the infiltrative surface can be quite significant in long term performance of subsurface infiltration systems. Several studies showed that trench designs are superior to bed designs (McGauhey and Winneberger, 1965; University of

Wisconsin, 1978). Recently, the use of shallow narrow trench systems have gained popularity because of their superior performance in marginal soils (Carlile, 1980; Hargett, 1984). Several factors may be of significance in explaining why shallow trenches perform better. The ratio of sidewall to bottom area is greater, thus increasing potential infiltrative surface area, shallow soil horizons are usually more permeable, and shallow placement enhances evapotranspiration which will reduce hydraulic loading. However, the most significant factor acting to reduce the resistance of the clogged zone appears to be the aeration status in the surrounding soil.

Several studies showed that maintenance of aerobic conditions at the infiltrative surface reduces the severity of clogging. McGauhey and Winneberger (1964) compared percolation rates through aerobic and anaerobic columns of sand. The respective conditions were maintained at the infiltrative surface by continuously inundating the columns with septic tank effluent and bubbling either oxygen or nitrogen through the ponded liquid. The columns which received the oxygen maintained infiltration rates several times higher than the columns receiving nitrogen, but eventually all columns clogged to the same degree. Black ferrous sulfide deposits which form under reduced (anaerobic) conditions were observed deep within the columns suggesting that the aerobic columns clogged well below the infiltrative surface where oxygen was unable to penetrate.

A similar phenomenon was described by Wood and Bassett (1975) of a groundwater recharge basin in Texas. They observed when oxygen saturated river water was infiltrated that a reduction in permeability first occurred well below the infiltrative surface. They surmised that some organic material penetrated deep into the profile and microbes, utilizing this as substrate, eventually depleted the oxygen. Under anaerobic conditions, permeability of the soil was reduced. The reduced permeability slowed the percolation of the oxygen-saturated water so that the oxygen supply was limited allowing the anaerobic zone to migrate upward until it reached the infiltrative surface.

In a study by Simons and Magdoff (1979) sand columns of 10, 30, 60 and 90 cm in length were dosed daily with septic tank effluent. The 10 cm columns failed within 18 days to accept their daily doses while the deeper columns continued for over 100 days. The 90 cm column never failed over the course of the experiment. The rapid failure of the shorter columns was attributed to poor aeration because of very low moisture tensions,*i.e.* high moisture content.

Field investigations of a small community SWIS consisting of large beds (100 by 130 ft) also resulted in observation of significantly reduced infiltration rates (Siegrist et al., 1985). Operating infiltration rate of the system was 1.4 cm/day, 72% less than the rate predicted from past observations of single family home system operation. Minimal sidewall area was indicated as one potential cause of the reduced infiltration rate. Anaerobiosis, caused by both wastewater-induced oxygen demand and the reduction of subsurface soil aeration due to the wide rectangular infiltrative surface geometry, was also implicated as a contributing factor. Indicators of anaerobic conditions observed, included high levels of methane gas in soil pores and high levels of ammonium in groundwater below the infiltrative surfaces.

As part of a study of gas transport in the unsaturated zone below a SWIS, Mahuta (1991) developed a model to predict oxygen concentrations below infiltrative surfaces for various physical dimensions associated with SWIS installations. The model was calibrated in a laboratory study using a bench scale SWIS in sandy soil. Based on a loading rate of 0.29 gpd/ft^2 with STE concentrations of 80 mg/L TOC and 50 mg/L TKN, the model predicted anaerobic conditions in SWIS wider than 12 ft or with groundwater closer than 5 ft below the infiltrative surface.

4.4.2.4: <u>Methods of Wastewater Application</u>

Storing the wastewater for intermittent discharge or dosing to the infiltration system as an alternative to gravity distribution has been shown by many investigators to reduce clogging (Bendixen et al., 1950; Winneberger et al., 1960; McGauhey and Winneberger, 1964; Jones and Taylor, 1965; Thomas et al., 1966; Popkin and Bendixen, 1968; Hills and Krone, 1971; Bouma et al., 1975; Hargett et al., 1982). The period between doses allows the soil surface to drain before receiving the next dose. Better results have been observed for larger, less frequent doses than for smaller, more frequent doses (Bendixen et al., 1950; Popkin and Bendixen, 1968). Dosing intervals of one day or more were most effective in retarding clogging.

The lower resistance of the clogged zone under a dosing regime is attributed to the aeration the infiltrative surface receives between doses. This was demonstrated in studies by Roats (1975) and Simons and Magdoff (1979). In these studies, septic tank effluent was intermittently applied to soil columns which were continuously maintained under atmospheres of either N_2 or air. After 56 days, Roats found that in his columns of

Alderwood gravelly sandy loam, infiltration rates were reduced to 29.5 percent and 53.5 percent of their initial rates for the anaerobic (N_2) and aerobic column atmospheres, respectively. Simons and Magdoff found that sand columns with the N_2 atmosphere failed to infiltrate the daily doses within 94 days while the aerobic columns operated satisfactorily over the entire course of the 220 day study.

Both diffusion and mass flow in response to drainage of water from the pores are important mechanisms in drawing oxygen into the soil (Lance et al., 1973; Hills and Krone, 1971). Oxygen supplied in this manner must be sufficient to balance the oxygen demand from materials added with each dose. If the soil moisture remains high, oxygen transfer is inhibited (Meek and Grass, 1975). Mahuta (1991) predicted anaerobic conditions below SWIS once air filled porosity in sandy soil dropped to 16 percent, based on a laboratory study and subsequent modeling of gas transport. Therefore, small, frequent doses which do not allow the soil to drain adequately before the subsequent dose are undesirable.

Although it has been amply shown that intermittent dosing reduces soil clogging, the hydraulic capacity of the infiltrative surface may not be increased. Kropf et al. (1975) argue that the infiltration time lost during the aeration phase may offset the gains from higher infiltration rates during the dosing phase. In experiments with sand columns which were either continuously flooded or dosed with frequencies from one-half to 36 times per day, the continuously flooded columns consistently infiltrated as much or more septic tank effluent than the dosed columns. These results are corroborated to some degree by Hargett et al. (1982). Using *in situ* lysimeters installed in silty clay loam, they found that daily dosing did not permit hydraulic loading rates greater than the infiltration rate which resulted from continuous ponding. Therefore, the apparent advantage of dosing appears to be in prolonging the life of the infiltration system rather than increasing its hydraulic capacity.

4.4.2.5: Wastewater Pretreatment

Improving the quality of the wastewater applied to the infiltration system is often suggested as a means to control clogging. Intuitively, reduced organic and suspended solids loading should reduce the biological and physical clogging of the soil.

Laak (1970, 1976) compared the rates of infiltration reduction produced by septic tank and extended aeration unit effluents in hand packed columns of medium sand, sandy loam, and garden soil. The five-day biochemical oxygen demand (BOD_5) concentrations were 189 mg/L and 124 mg/L and total suspended solids (TSS) concentrations were 69 mg/L and 70 mg/L, respectively, for the septic tank and extended aeration effluents. The extended aeration effluent had a dissolved oxygen concentration of 2.4 mg/L. Columns receiving the lower hydraulic loads and higher quality effluent performed longer. The hydraulic loading was the most significant factor in prolonging the life of the columns, but the BOD_5 and the TSS loading was also significant. Decreasing the sum of BOD_5 and TSS concentrations in the applied wastewater increased the infiltration rates of the sands by a factor of $[BOD_5(mg/L) + TSS\ (mg/L)]^{1/3} / 250^{1/3}$. This factor did not apply in soils with low permeability (Laak, 1976).

Roats (1975) compared the effects of septic tank and extended aeration effluents on the infiltration capacity of hand packed columns of Alderwood gravelly sandy loam. The average concentrations of BOD_5 were 173 mg/L and 4 mg/L and concentrations of TSS were 43 mg/L and 63 mg/L, respectively, for the septic tank and extended aeration effluents. The extended aeration effluent had a dissolved oxygen concentration of 6.4 mg/L. The effluents were applied to the columns once per day at the rate of 5 cm/day (1.2 gpd/ft^2). After 56 days, the infiltration rate of the column receiving extended aeration effluent was only 13.3 percent of the original value, while in the column receiving septic tank effluent the infiltration rate was still 53.5 percent of its original value. Close inspection of the columns revealed that a significant surface mat had developed on the column loaded with the extended aeration effluent. The column receiving the septic tank effluent did not display these same characteristics and column effluent was higher in suspended solids. Therefore, the differences in infiltration rate decline were attributed to differences in the nature of the suspended solids in the effluents.

Nykiel (1983) investigated differences in the hydraulic performance of coarse sand columns loaded with domestic greywater septic tank effluent and domestic septic tank effluent. Average effluent concentrations for the greywater were 83, 31, and 21 mg/L for BOD_5, TSS, volatile suspended solids (VSS), respectively. Average effluent concentrations for the domestic STE were 110, 66, and 60 mg/L for BOD_5, TSS and VSS, respectively. Time to continuous ponding in the greywater columns was observed to be 600% longer (360 days) at 20 cm/day and 200% longer (100 days) at 40 cm/day than in columns receiving the septic tank effluent. In addition, the location of soil clogging was observed to differ with effluent type. Clogging occurred more in-depth in the greywater columns as compared to the surficial clogged zone in the septic tank columns. He postulated that biological activity may have been the primary factor driving clogging in columns receiving greywater, and physical straining the primary factor driving clogging in columns receiving septic tank effluent.

In another study, Siegrist et al. (1985) observed infiltration behavior differences in sand-filled columns receiving domestic versus restaurant septic tank effluent. Average concentrations of BOD_5, chemical oxygen demand (COD), TSS, and total volatile suspended solids (TVSS) for domestic septic tank effluent were 140, 356, 88, and 61 mg/L, respectively. Restaurant septic tank effluent concentrations averaged 377, 772, 247 and 173 mg/L for BOD_5, COD, TSS and TVSS, respectively. Reduction of infiltration rates were observed in the columns receiving either effluent, however, the reduction was much greater in columns receiving restaurant septic tank effluent. The maximum observed reduction of infiltration rate due to loading of restaurant effluent was nearly 100% for eight days of operation, whereas for columns loaded with domestic effluent, infiltration rates were reduced nearly 37% within 67 days of operation.

In the study of soil clogging by Siegrist (1987) described earlier, results indicated that clogging development was accelerated at higher hydraulic loading rates or with more concentrated effluents. Siegrist demonstrated that soil clogging development was significantly correlated with the cumulative mass density loadings of tBOD (ultimate carbonaceous and nitrogenous BOD) and suspended solids. As a result, soil clogging may be retarded by reducing the applied mass loading rates of these materials either through lower hydraulic loading rates or reduced effluent concentrations. Thus, wastewater effluents possessing concentrations of tBOD and TSS lower than typical domestic septic tank effluent (e.g. sand filter effluent) may be applied at hydraulic loading rates higher than the 1 to 5 cm/d used for domestic septic tank effluent without causing accelerated soil clogging. Conversely, effluents possessing higher concentrations of these materials (e.g. restaurant septic tank effluent) should be applied at correspondingly lower rates.

These studies indicate that the organic strength and suspended solids concentration of the applied wastewater are important factors in reducing or retarding clogging, but the relative significance of each was not demonstrated. To determine the effects of suspended solids on soil clogging, Rice (1974) continuously loaded columns of loamy and coarse sands with secondary effluents having high and low suspended soils concentrations. High

TSS concentrations were achieved by adding activated sludge. To obtain lower TSS concentrations, the effluent was filtered. Results showed that below 10 mg/L TSS, small increases in TSS concentrations resulted in a large reduction in the hydraulic capacity of the columns. At TSS concentrations above this threshold, the reduction in hydraulic capacity was much less dependent upon TSS.

Frankenberger et al. (1979) investigated the relationship between biological clogging and applied wastewater quality on packed columns of Nicollet loam, a fine loamy soil, and Tama silty clay loam, a fine silty soil. Sterile water, distilled water, and distilled water with either glucose or glucose plus KNO_3 were added. Bacterial counts and measurement of phosphatase activity implicated biological activity as the cause of the clogging. Both the bacterial population and phosphatase activity were highest with applications of the glucose plus KNO_3 and water solution.

Okubo and Matsumoto (1983) prepared synthetic wastewater to investigate the effects of both organic strength and suspended solids concentrations on infiltration rates of sand. The wastewaters were prepared from glucose and homogenized activated sludge flocs. The wastewaters were dosed continuously to the columns. Results showed that to maintain reasonably high infiltration rates over prolonged inundation periods, the soluble organic carbon should be less than 10 mg/L and the suspended solids concentration less than 2 mg/L.

Mahuta's (1991) studies of gas transport below SWIS suggested that increased organic strength in applied wastewater had a significant effect on oxygen concentration below the SWIS. At a 0.29 gpd/ft^2 loading rate on sandy soils, modeling of gas transport predicted the development of anaerobic conditions by increasing wastewater organic strength from 80 to 160 mg/L TOC.

4.4.2.6: Infiltrative Surface Resting

Prolonged resting of the infiltrative surface is promoted as a good management technique to restore hydraulic capacity of the system. During the resting phase, wastewater application is stopped to allow the infiltration system to drain and the clogged zone to degrade. Thomas et al. (1966) rested columns of Ottawa sand by exposing the surfaces to air for 125 days after they had become continuously ponded following over 200 days of daily dosings with septic tank effluent. By the end of the first 23 days of the resting period, the infiltration rate recovered to nearly the same rate it was just before the columns went into the anaerobic phase. This partial recovery corresponded to a decrease in the concentration of total organic matter in the soil, but polysaccharide and polyuronide concentrations did not change. They concluded that the air drying promoted biochemical degradation of the organic matter which was a significant agent of clogging. The polysaccharides and polyuronides may have merely changed their physical properties by irreversibly dehydrating which would also contribute to the recovery.

Simons and Magdoff (1979) found that the degree of recovery is related to the length of time the infiltrative surface is inundated and the soil moisture content during the resting phase. They found that resistance of the clogged zone (thickness of the biomat divided by its hydraulic conductivity) of sand columns should be reduced to at least 1.5 hours if the columns were not to fail rapidly upon restarting daily application of the septic tank effluent. The length of time required to reach this point varied from 10 to 250 days and correlated closely to the length of time the columns had remained ponded. The length of the resting period was also inversely related to the length of the columns suggesting that good drainage or low moisture tensions are needed for effective resting.

Although resting is a good management technique and may be necessary in aggregated soils to maintain acceptable infiltration rates, it is seldom employed because an alternate system is required. Installing two systems increases the cost and land area needed. If the area of two alternating systems could be reduced significantly, resting would probably be used more widely. The lysimeter study performed by Hargett et al. (1982) suggests this may be possible. When silty clay loam soil was dosed once per day with 2 and 4 times the recommended loading rate, the infiltrative surface had just begun to pond after 12 to 14 months of operation. If resting were effective, a two cell infiltration system operating alternately on a 12 month schedule might require equal total area of or even one-half the area of a single cell system. Unfortunately, alternation was not investigated in this study.

4.4.3: Summary of Factors Affecting Hydraulic Performance

Studies under a variety of conditions have shown that soil clogging begins immediately with wastewater application and proceeds slowly with time. Eventually, the liquid infiltration rate is reduced to a small percentage of the initial rate. Clogging seems to be a surface phenomenon due primarily to biological activity stimulated by the nutrients in wastewater. Suspended solids also contribute to clogging, but appear to be significant only during the initial stages. Suspended solids, microbial cell mass, and decomposition by-products accumulate on or just below the infiltrative surface. It appears that if the rate of organic material additions exceed the rate of decomposition, the infiltration rate declines more rapidly. Once permanent ponding occurs, aeration is inhibited and the high oxygen demand of accumulated materials creates anaerobic conditions in the soil below the infiltrative surface. As a result, the infiltration rate declines more rapidly because anaerobic decomposition of clogging agents is less efficient, thus, slower.

Control of soil clogging requires consideration of many factors. It begins with site selection. Well drained, sandy soils with thick unsaturated zones are preferred to other types. Air diffusion into soil pores is promoted in soils with high permeability and low moisture content. Avoidance of saturated zones caused by soil stratification or proximity to the water table is necessary to maintain high moisture tensions. This is not to imply that subsurface infiltration is not effective in finer textured, aggregated soils. However, the shortcomings of these soils must be addressed by emphasizing design and management.

Design can affect the aeration status of the clogged zone. Shallow, narrow trenches or mounds promote good soil aeration by placing the infiltrative surface in more permeable soil materials, providing maximum separation from the water table, and maximizing moisture loss through evapotranspiration during the growing season. Narrow trenches are more effective than wide beds because the distances for oxygen diffusion to soil below the infiltrative surface are shorter.

Sizing criteria for infiltration systems are usually specified as a function of the percolation rate of the soil or of the soil texture. However, some studies have shown that other factors such as depth, geometry of the infiltrative surface, and soil temperature, structure and bulk density may be equally important. Construction of shallow, narrow trenches, in warm soil materials with granular or strong, fine structure appears to provide superior performance due to the warm, aerobic conditions in the subsoil.

Improving the quality of the wastewater by reducing suspended solids and organic carbon concentrations has been shown to reduce the rate and possibly the intensity of soil clogging. The effect is probably greatest in granular soils. The hydraulic conductivity of granular soils changes dramatically with small changes in moisture tension. Therefore, small reductions in the resistance of the clogged zone can have a profound effect on the rate of infiltration. This is not true in fine textured soils. Also, in structured soils, prolonged periods of wastewater application may degrade the aggregates and decrease the size of water carrying pores. In fine textured soils, flow reduction through water conservation may be more effective in maintaining the hydraulic capacity of the system.

Intermittent dosing of wastewater onto the infiltrative surface and periodic resting of the system are two effective techniques to prolong the life of infiltration systems. Selection of a proper dosing frequency will permit the infiltrative surface to be periodically exposed to air providing the period between doses is sufficient to permit the surface to drain completely. Complete drainage also aids in aeration of the subsoil by drawing air in behind the percolating liquid. The period of aeration should be sufficient to balance the oxygen supply with the oxygen demand of the materials added with each dose. Thus, the proper frequency is affected by the physical properties of the soil and the applied water quality. Periodic resting will increase infiltration rates by allowing the accumulated organics to be biochemically oxidized and the soil to dry and reaggregate. Resting may be essential to long term performance of systems in aggregated soils.

Based on current knowledge of conventional system performance, the hydraulic performance of SWIS can be maximized by incorporating the following features:

- Pretreatment to remove organics and suspended solids to concentrations less than or equal to those of typical domestic septic tank effluent;
- Narrow trenches, 0.5 to 3 ft wide, excavated parallel to surface or groundwater surface contours;
- Shallow placement, less than 2 ft from final grade to the infiltrative surfaces;
- Wastewater application rates that account for soil characteristics and proposed system design (see Table 4.4.1);
- Dosing of the infiltrative surfaces one to four times daily;
- Multiple cells (two minimum) or reserve areas to allow periodic resting and standby capacity for emergency operation or reconstruction; and
- Devices for monitoring daily wastewater flow, infiltrative surface ponding, and, if necessary, groundwater surface elevation monitoring.

TABLE 4.4.1: RELATIVE ADJUSTMENTS TO CONVENTIONAL DESIGN HYDRAULIC LOADING RATES

SOIL OR DESIGN FACTOR (BASELINE)[a]	RELATIVE ADJUSTMENT TO LOADING RATE
SOIL STRUCTURE (granular or blocky)	
Increasing macropores	No adjustment
Decreasing macropores	Decrease 10 to 50%
SHRINK/SWELL POTENTIAL (<20% expandable clays)	
Greater than 20% 2:1 clays	Avoid
Less than 20% 2:1 clays	No adjustment
SOIL BULK DENSITY (low to moderate)	
Increasing	Decrease 10 to 50%
Decreasing	No adjustment
SOIL MOISTURE (moderately to well drained)	
Excessively drained	No adjustment
Poorly drained	Avoid or modify
SOIL TEMPERATURE (mean annual temperature 5 to 10°C)	
Mean annual temperature >15°C	Increase 10 to 30%
Mean annual temperature < 5°C	Decrease 10 to 50%
INFILTRATIVE SURFACE GEOMETRY (2 to 3 ft trenches)	
Infiltrative surface >4 ft.	Decrease
Infiltrative surface <2 ft.	No adjustment
INFILTRATIVE SURFACE DEPTH (3 ft below grade)	
Infiltrative surface <3 ft. below grade	No adjustment
Infiltrative surface >3 ft. below grade	Decrease 10 to 30%
WASTEWATER APPLICATION (dosing 1 to 4 times daily)	
Dosing/resting	No adjustment
Continuous ponding	Avoid
WASTEWATER STRENGTH (BOD_5: 150 mg/L; TSS: 80 mg/L; TKN: 55 mg/L)	
Increasing BOD_5, TSS, and TKN	Decrease in proportion to waste strength
Decreasing BOD_5, TSS, and TKN	Increase in proportion to waste strength
EVAPOTRANSPIRATION POTENTIAL (precipitation ≥ evaporation)	
Increasing	Increase 10 to 20%
Decreasing	No adjustment

a Baseline values in parentheses are those typical for conventional design hydraulic loading rates.

4.5: TREATMENT PERFORMANCE

Under proper site and operating conditions, conventional septic tank systems are capable of nearly complete removal of biodegradable organics, suspended solids, and fecal coliforms. These are parameters which have been traditionally monitored as a means of assessing wastewater treatment performance. However, environmental protection and public health agencies are increasingly concerned over potential ground and surface water contamination from other wastewater constituents. Potential impacts include toxicity, introduction of pathogenic agents, and excessive fertilization of surface waters.

Wastewater may contain toxic compounds including nitrogen, toxic organics, and metals which may be released into groundwater or surface waters. Although ubiquitous in the environment, nitrogen can be toxic under specific conditions. Nitrate nitrogen in drinking water has been linked to methemoglobinemia in infants, a disease which reduces the oxygen carrying capacity of blood. Infants fed formula reconstituted with high nitrate water are at risk, particularly in the first three months of age, whereas infants that are breast-fed, fed whole milk, or pre-prepared formula are unlikely to be at risk (National Academy of Sciences, 1974). To eliminate potential risk, the U.S. Public Health Service established a nitrate limitation of 10 mg-N/L in public water supplies. This level was established from data that showed no reported cases of methemoglobinemia at concentrations below 10 mg-N/L (Walton, 1951). This level remained the drinking water standard under the Safe Drinking Water Act (SDWA) promulgated by U.S. EPA in 1974. Ammonia nitrogen can also be toxic to fish if present in sufficiently high concentrations.

Toxic organics are in a variety of household chemicals and cleaning agents which can be ultimately discharged with wastewater (Hathaway, 1980). Many of the organic substances are persistent in the aqueous environment and are known to be carcinogenic.

Metals such as lead, tin, zinc, copper, iron, cadmium, and arsenic, are present in many household products and plumbing systems. Although metals are readily adsorbed or otherwise removed from solution, it is theoretically possible that cationic surfactants could displace metals associated with particulate matter via ion exchange, thus keeping metals in solution (Rapaport, 1991). In excessive concentrations in drinking water, they can be toxic. In the aquatic ecosystem, they can accumulate in fish which ultimately may be consumed by humans and cause metal toxicity.

Increasing the productivity of the water through nutrient addition may also generate trihalomethanes (THMs) in treated water. It has been shown that algae and aquatic weeds can contribute significant amounts of THM precursors to the water column (AWWA, 1989). Upon chlorination, THMs, known carcinogens, can be created. It has been demonstrated that both whole algal cells and algal components can form chloroform when chlorinated during water treatment (Morris and Baum, 1978; Briley, et al., 1984).

Endotoxins may also enter surface waters through the groundwater below land application systems (U.S. EPA, 1984a; Rail, 1989). Gram-positive bacteria are the predominant bacterial microflora of soil, while gram-negative bacteria normally predominate in water. When wastewater is irrigated onto land, antagonism between the two groups of bacteria eliminates the gram-negative bacteria. Endotoxins, consisting of lipopolysaccharides from the cell wall of gram-negative bacteria are released as a result. Endotoxins were found to occur in groundwater below infiltration ponds in Denmark at concentrations 10,000 times the minimum dose needed to produce clinically measurable effects by parenteral injection in humans. Once in groundwater, endotoxins may reach surface waters. Endotoxins are only toxic if they enter the bloodstream, so they pose problems where potable water is used for the production of intravenous solutions.

Pathogenic agents found in wastewater include bacteria and viruses. There are a number of genera and many species of bacteria, some enteric and some nonenteric, which may become pathogenic if ingested in sufficient numbers. Genera of potentially pathogenic enteric bacteria include Escherichia, Klebsiella, Proteus, Salmonella, Serratia, Shigella, Streptococcus, Vibrio, and Yersinia (University of Wisconsin, 1978; Morrison, 1983; Pekdeger, 1984). Other nonenteric, potentially pathogenic bacteria which may be present in wastewater include Aeromonas, Campylobacter, Clostridium, Francisella, Legionella, Leptospira, Mycobacterium, Pseudomonas, and Staphylococcus (University of Wisconsin, 1978; Morrison, 1983; Pekdeger, 1984; Tchobanoglous and Burton, 1991). In general, these organisms can cause intestinal illnesses to varying degrees, however, others, particularly the nonenteric bacteria, may cause infections in other organs of the body, e.g. respiratory (Klebsiella, Legionella, Mycobacterium, and Pseudomonas) and renal (Leptospira).

Viruses are obligate, intracellular parasites that are host specific. They differ fundamentally from bacteria. They are submicroscopic particles ranging in size from 0.025 to 0.100 microns in diameter (bacteria range in size from 0.5 to 2.0 microns) and are incapable of reproduction or other life functions outside a host cell.

Viruses are capable of causing various respiratory, enteric, and other diseases. Respiratory viruses are transmitted through the air while enteric viruses are shed in the feces of the host and transmitted to other hosts primarily through food or water. The number of virus particles that constitute an infectious dose varies, but it has been shown that one PFU is capable of producing human infection (Katz and Plotkin, 1967). Cells which are infected and producing virus tend to abandon their special functions in the host body and sometimes die. If enough cells die or become functionally diverted, disease results.

Over half of waterborne disease outbreaks in the United States are classified as acute gastroenteritis of unknown etiology (Lippy and Waltrip, 1984). Recent retrospective seriological studies, however, suggest that many of the gastroenteritis outbreaks are caused by Norwalk and Norwalk-like viruses and rotaviruses (Kaplan et al., 1982). It is estimated that 23 percent of the reported water-borne disease outbreaks in the U.S. are caused by Norwalk viruses (Keswick et al., 1985). Over a hundred different enteric viruses are known to be excreted in human feces (Berg, 1964; Buras, 1974). These include poliovirus, the causative agent in paralytic poliomyelitis; coxsackieviruses which cause herpangina, myocarditis, pleurodynia, meningitis, and diarrhea; echovirus, responsible for meningitis and diarrhea; and hepatitis A, which causes infectious hepatitis (Gerba et al., 1975; Gerba and Keswick, 1981; Kowal, 1982; Sobsey, 1983b).

The nutrients, nitrogen and phosphorus, in wastewater may contribute to fertilization of surface waters which can lead to excessive growth of algae and other aquatic plants. Fresh water bodies are typically phosphorus limited. Other necessary aquatic plant nutrients are usually present in abundance so that only very low concentrations of phosphorus can lead to a direct increase in aquatic plant growth. Lake studies have shown that when total phosphorus concentrations exceed 30 ug/L, lakes tend to be highly productive or eutrophic (Vollenweider, 1968). Growth can lower water quality and its decay at the reservoir bottom can alter the oxidation state of sediments to release more phosphorus stored there. Release of more phosphorus can trigger even larger algae blooms and a concomitant increase in available phosphorus from bottom sediments.

Other water bodies, such as estuaries, may be nitrogen limited such that additions of nitrogen will lead to increased growth of aquatic plants.

4.5.1: Fate and Transport of Wastewater Constituents in Soil and Groundwater

4.5.1.1: <u>Nitrogen</u>

Septic tanks remove approximately 30% of the nitrogen in raw domestic wastewater (University of Wisconsin, 1978). The nitrogen either is retained in the sludge which is periodically removed for off-site disposal or, if in the form of nitrate, denitrified in the anaerobic environment of the tank and vented as nitrogen gas. The remaining nitrogen leaves the tank in the form of dissolved ammonium (NH_4^+) or organic nitrogen. Approximately 75% of effluent nitrogen is ammonium (University of Wisconsin, 1978; Kristiansen, 1981a,b; Ronayne et al., 1982).

Nitrogen can undergo several transformations below a subsurface wastewater infiltration system including adsorption, volatilization, mineralization, nitrification, denitrification, and biological uptake. Nitrification, the conversion of organic and ammonium nitrogen to nitrate by microorganisms under aerobic conditions, is the predominant transformation that occurs below subsurface infiltration systems. The negatively charged nitrate ion is very soluble and moves readily with the percolating soil water. Biological denitrification, which converts nitrate to gaseous elemental nitrogen, can remove nitrogen from percolating wastewater. Denitrification occurs where anaerobic conditions prevail with an available carbon source. Denitrifying bacteria use nitrate as a substitute for oxygen as an electron acceptor. It has been generally thought that anaerobic conditions with organic matter seldom occur below SWIS. Therefore, it is usually assumed that all the nitrogen applied to SWIS ultimately leaches to groundwater (Walker et al., 1973a,b; Brown et al., 1978). However, several studies indicate that denitrification can be significant. Jenssen and Siegrist (1988) found in their review of several laboratory and field studies that approximately 20% of nitrogen is lost from wastewater percolating through soil. Factors that have been found to favor denitrification are fine grained soils (silts and clays) and layered soils (alternating fine grained and coarser grained soils with distinct boundaries between texturally different layers) particularly if the fine grained soil layers contain organic material (Jenssen and Siegrist, 1988). It was also concluded that nitrogen removal in conventional septic tank systems can be enhanced by placing the system high in the soil profile where organic matter in the soil is more likely to exist and by dosing septic tank effluent onto the infiltrative surface to create alternating wetting and drying cycles.

Nitrogen contamination of groundwater below SWIS has been documented by many investigators (Preul, 1966; Polta, 1969; Bouma et al., 1972; Ellis and Childs, 1973; Walker et al., 1973a,b; Viraraghavan and Warnock, 1976a,b,c; Gibbs, 1977a,b; Reneau, 1977, 1979; Peavy and Groves, 1978; Andreoli et al., 1979; Erickson and Bastian, 1979; Peavy and Brawner, 1979; Wolterink et al., 1979; Starr and Sawhney, 1980; Carlile et al., 1981; Cogger and Carlile, 1984; Uebler, 1984; Ayres Associates, 1989; Robertson et al., 1989, 1990; Tinker, 1991). Nitrate nitrogen concentrations were usually found to exceed the drinking water standard of 10 mg-N/L near the SWIS.

Nitrate moves freely in groundwater with little retardation. Denitrification has not been found to be significant in the saturated zone. Reduction of nitrate concentrations in groundwater occurs primarily through dilution. However, nitrogen loss has been documented to occur via denitrification as groundwater enters surface water bodies because it must pass through organic rich bottom sediments. Nitrogen concentrations in groundwater were shown to decrease to less than 0.5 mg-N/L after passage through sediments in one Canadian study (Robertson et al., 1989; 1990).

4.5.1.2: Phosphorus

The septic tank removes approximately 4 to 8% of phosphorus in raw wastewater through sedimentation. This estimate is based on annual raw wastewater loadings and concentrations of phosphorus in septage removed from the tank (Rezek and Cooper, 1980; U.S. EPA, 1984a, 1984b; Ayres Associates, 1985). Approximately 80% to 85% of the phosphorus that remains suspended or in solution passes from the tank, primarily in the form of orthophosphate. The other 15% to 20% of the phosphorus is organic (University of Wisconsin, 1978; Alhajjar et al., 1989).

The fate and transport of phosphorus in soils is controlled by sorption and precipitation reactions (Sikora and Corey, 1976). At low concentrations (<5 mg/L) the phosphate ion is chemisorbed on the surfaces of iron and aluminum minerals in strongly acid to neutral systems and on calcium minerals in neutral to alkaline systems. As phosphorus concentration increases, phosphate precipitates form. Some of the more important compounds formed include strengite ($FePO_4 \cdot 2H_2O$), variscite ($AlPO_4 \cdot 2H_2O$), dicalcium phosphate ($CaHPO_4 \cdot 2H_2O$), octacalcium phosphate ($Ca_4H(PO_4)_3 \cdot 3H_2O$), and hydroxyapatite ($Ca_{10}(PO_4)_6(OH_2)$). In acid soils, phosphate sorption probably involves the aluminum and iron compounds while in calcareous or alkaline soils, calcium compounds predominate.

Estimates of the capacity of the soil to retain phosphorus are often based on sorption isotherms, fitted to the Langmuir model (Ellis and Erickson, 1969; Sawhney and Hill, 1975; Sikora and Corey, 1976; Sawney, 1977; Tofflemire and Chen, 1977). This method significantly underestimates the total retention capacity of the soil (Sawney and Hill, 1975; Sikora and Corey, 1976; Tofflemire and Chen, 1977). This is because the test measures the chemisorption capacity, but does not take into account the slower precipitation reactions which regenerate the chemisorption sites. These slower reactions have been shown to increase the capacity of the soil to retain phosphorus by 1.5 to 3 times the measured capacity by the isotherm test (Sikora and Corey, 1976; Tofflemire and Chen, 1977). In some cases, the total capacity has been shown to be as much as 6 times greater (Tofflemire and Chen, 1977). These reactions may take place in either unsaturated or saturated soils (Ellis and Childs, 1973; Reneau and Pettry, 1976; Sikora and Corey, 1976; Jones and Lee, 1977a, 1977b; Robertson et al., 1990).

The capacity of the soil to retain phosphorus is finite, however. With continued loading, phosphorus movement deeper into the soil profile can be expected to occur. The ultimate capacity of the soil is dependent on several factors including its mineralogy, particle size distribution, oxidation-reduction potential and pH. Fine textured soils provide more sorption sites for phosphorus. Iron, aluminum, and calcium minerals in the soil allow precipitation reactions to occur. Sikora and Corey (1976) estimated that phosphorus penetration into the soil below a subsurface infiltration system would be 52 cm/yr in Wisconsin sands and 10 cm/yr in Wisconsin silt loams.

However, knowing the retention capacity of the soil is not enough to predict the travel of phosphorus from subsurface infiltration systems. Equally important is an estimation of the total volume of soil which the wastewater will contact as it percolates to and through the groundwater. Fine textured, unstructured soils can be expected to disperse the water and cause contact with a greater volume of soil than coarse, granular soils or strongly structured fine textured soils having large continuous pores. Also, the rate of water movement and the degree to which its elevation fluctuates are important factors.

Monitoring of groundwater below subsurface infiltration systems has shown that the amount of phosphorus leached to groundwater is dependent on the characteristics of the soil, the unsaturated thickness through which the wastewater percolates, the applied loading rate, and the age of the system (Bouma et al., 1972; Brandes, 1972; Dudley and Stephenson, 1973; Ellis and Childs, 1973; Childs et al., 1974; Harkin et al., 1979; Jones and Lee, 1979; Erickson and Bastian, 1980; Carlile et al., 1981, Gilliom and Patmont, 1983; Cogger and Carlile, 1984; Whelan and Barrow, 1984). The amount of phosphorus in groundwater has been found to vary from background concentrations to concentrations equal to that of septic tank effluent. However, removals were found to continue within groundwater aquifers (Ellis and Childs, 1973; Childs et al., 1974; Reneau and Pettry, 1976; Reneau, 1979; Rea and Upchurch, 1980; Carlile et al.; 1981; Gilliom and Patmont, 1983; Cogger and Carlile, 1984; Robertson et al., 1990). Therefore, retardation of phosphorus contamination of surface waters from subsurface infiltration systems is enhanced by construction in fine textured soils without continuous macropores that allow rapid percolation. Distance of the system to the surface water is also an important factor.

4.5.1.3: Pathogenic Bacteria

Due to the number of different pathogenic bacteria and the infrequency of their occurrence in raw wastewater, study of pathogenic bacteria occurrence in septic tank effluent has been limited to a few genera and to general tests that identify groups of coliform bacteria, i.e. total coliforms (Citrobacter, Enterobacter, Escherichia, and Klebsiella) and fecal coliforms (American Public Health Association, 1989; Tchobanoglous and Burton, 1991). Septic tanks have not been found to appreciably reduce the bacterial numbers present in raw wastewater (U.S. EPA, 1980).

Once enteric and nonenteric bacteria enter a soil, they are subjected to life process stresses not encountered in the host. Temperatures will be much lower, nutrients and energy sources will likely be appreciably less in quantity and availability, and pH, moisture, and oxygen contents will not likely be conducive to long term survival. Survival time of enteric bacteria in the soil is reduced by increasing temperatures, low nutrient and organic matter contents, acidic conditions (pH values of 3 to 5), low moisture content, and the presence of indigenous soil microflora (Gerba et al., 1975). Elimination of potentially pathogenic bacteria is faster at high temperatures (37°C), pH values of about 7, low oxygen content, and high dissolved organic substance content (Pekdeger, 1984). The rate of bacterial die-off approximately doubles with 10°C increases of temperature between 5 to 30°C (Tchobanoglous and Burton, 1991). Observed survival rates for various potential pathogenic bacteria have been found to be extremely variable. Longer survival times than six months may occur at greater depths within unsaturated soil where oligotrophic conditions exist (Pekdeger, 1984).

The main methods of bacterial retention in unsaturated soil are filtration, sedimentation, and adsorption (Gerba et al., 1975; Bicki et al., 1984; Cantor and Knox, 1985). Filtration accounts for the most retention. The size of bacteria range from 0.2 to 5 μm (Pekdeger, 1984; Tchobanoglous and Burton, 1991). Thus, physical straining will occur by soil micropores and surface water film thicknesses smaller than this. Sedimentation is supported by slow permeability rates of wastewater in soil which may be caused by fine textures and/or unsaturated conditions. Adsorption of bacteria onto clay and organic colloids will occur within a soil solution having high ionic strength and neutral to slightly acid pH values (Canter and Knox, 1985).

Normal operation characteristics of septic tank/subsurface infiltration systems result in retention and die-off of most, if not all, observed pathogenic bacterial indicators within 3 ft of the infiltrative surface (McGauhey and Krone, 1967; Bouma et al., 1972). With a mature biomat at the infiltrative surface, most are removed within the first vertical or horizontal foot of the trench/soil interface (University of Wisconsin, 1978). Hydraulic loading rates of less than or equal to 5 cm/day (2 inches/day) have also been found to promote better removal of bacteria introduced with septic tank effluent (Ziebell et al., 1975). Biomat formation and lower hydraulic loading rates promote unsaturated flow which is the key to removal of bacteria from wastewater discharged to soil for treatment. Retention behavior of actual pathogens in unsaturated soil may be different than that of the indicators, such as coliforms, that have been fairly well studied.

Failure to properly site, design, install, and/or operate and maintain subsurface infiltration systems can result in the introduction of bacteria, some of which may be pathogenic, into groundwater. Literature reviews prepared by Hagedorn (1982) and Bicki et al. (1984) identify a number of references which provide evidence that infiltrative surfaces improperly constructed below the groundwater surface or too near fractured bedrock correlate with such contamination. Once in groundwater, bacteria introduced with septic tank effluent have been observed to survive for sufficient lengths of time (7 hours to 63 days) to travel distances as much as 100 ft (Gerba et al., 1975).

4.5.1.4: Viruses

Viruses are not a normal part of the fecal flora. They occur in septic tank effluent intermittently, in varying numbers, and reflect the combined infection and carrier status of the house residents (Berg, 1973). It is estimated that less than 1 to 2 percent of the stools excreted in the United States contain viruses (University of Wisconsin, 1978). Therefore, they are difficult to monitor and, as a result, little is known about their frequency of occurrence and rate of survival in septic tank systems. However, once an infection (clinical or subclinical) has occurred, it is estimated that feces may contain 10^6 to 10^{10} viral particles per gram (Kowal, 1982). Consequently, if virus is present in septic tank effluent, it is likely to be present in high numbers. Vaughn and Landry (1977) sampled raw household wastewater and septic tank effluent over an eleven month period. Virus concentrations in raw wastewater that entered a septic tank ranged from 0 to 2,365 plaque forming units per liter (PFU/L). Viruses were detected at 2.6 PFU/L in only one septic tank effluent sample collected during the period. Yeager and O'Brien (1977) and Hain and O'Brien (1979) sampled five septic tanks in New Mexico and found virus in three of them ranging in concentrations from 1,600 to 3,700 PFU/L. Harkin et al. (1979) monitored 33 randomly selected septic tank systems over a nine month period. In 78 effluent samples collected, no evidence of virus was found.

Viruses may be both retained and inactivated in soil. However, viruses may also be retained but not inactivated. If not inactivated, viruses may accumulate in soil and subsequently released due to changing conditions. The result could be contamination of groundwater. Table 4.5.1 presents a summary of the factors which influence virus survival in soils and Table 4.5.2 presents a summary of the factors which influence virus retention. A thorough review of these factors is presented by Sobsey (1983a).

Most studies of the fate and transport of virus in soils has been with column studies using a specific serotype, typically poliovirus 1, or bacteriophages (Robeck et al., 1962; Drewry and Eliassen, 1968; Drewry, 1969, 1973; Hori et al., 1971; Nestor and Costin, 1971; Goldsmith et al., 1973; Lefler and Kott, 1973, 1974; Young and Burbank, 1973; Green and Cliver, 1975; Dubois et al., 1976; Lance et al., 1976; Schaub and Sorber, 1977; Burge and Enkiri, 1978; University of Wisconsin, 1978; Bitton et al., 1979; Lance and Gerba, 1980; Sobsey et al., 1980; Lance et al., 1982). The generalized results of these studies indicate that adsorption of virus is the principal mechanism of virus retention in soil. Once retained, inactivation rates range from 30% to 40% per day. Adsorption is enhanced by increasing ionic strength of the wastewater.

TABLE 4.5.1: RESPONSE TO FACTORS AFFECTING VIRUS SURVIVAL IN SOILS (AFTER SOBSEY, 1983a)

Factor	Virus Survival Response
Temperature	Decreases with increasing temperature
Microbial Activity	Decreases with increasing activity
Soil Moisture	Increases with increasing moisture
pH	Decreases at pH below 3 and above 9
Ionic Strength	Decreases with increasing ionic strength
Virus Association with Particulate Matter	May increase or decrease
Soil Organic Matter	Decreases with increasing organic matter

TABLE 4.5.2: RESPONSE TO FACTORS AFFECTING VIRUS RETENTION IN SOILS (AFTER SOBSEY, 1983a)

Factor	Virus Retention Response
Soil Texture	Increases with decreasing particle size
pH	Increases with decreasing pH
Ionic Strength	Increases with increasing ionic strength and increasing cation valencies
Organic Matter	Decreases with increasing organic content
Moisture Content	Decreases with increasing moisture content and hydraulic load
Virus Type	Different types and strains are not retained equally relating to chemical properties of encapsulation

The transport of viruses through unsaturated soil below wastewater infiltration systems has been monitored by various investigators (Wellings et al., 1975; Schaub and Sorber, 1977; Hain and O'Brien, 1979; Vaughn and Landry, 1980; Vaughn et al., 1981; Vaughn et al., 1982, 1983; Jansons et al., 1989a). Most of these studies focused on indigenous viruses in the wastewater. In most cases, viruses were found to penetrate more than 3 m of unsaturated soil to the groundwater. Some serotypes were found to move more freely than others.

Poliovirus Type 1, commonly used in soil column studies, has been found to be far less mobile than other enteroviruses suggesting it is a poor indicator of pollution potential. Studies also demonstrated that fecal coliforms are not effective indicators of human enterovirus contamination because they are effectively removed in soil where indigenous viruses are not.

Virus contamination of groundwater has been found to occur below land application sites. Rigorous studies are limited, however. Most viral monitoring studies have been of large municipal rapid or slow rate infiltration systems, but some recent studies have investigated septic tank systems (Hain and O'Brien, 1979; Scanjura and Sobsey, 1981; Stramer and Cliver, 1981; Vaughn et al., 1983; Yates, 1985). Results vary as to distance migrated, but all indicate that significant vertical and horizontal travel is possible. Temperature and dissolved oxygen concentration in the groundwater have been found to be significantly correlated with inactivation rate (Yates et al., 1985; Yates et al., 1986; Jansons et al., 1989). Based on estimated decay rates for virus in groundwater, several investigators have recommended a re-examination of permitted setback distances of septic tank systems from wells and surface waters (Gerba, 1982; Vaughn et al., 1983; Yates et al., 1986; Jansons et al., 1989a, 1989b). Yates et al. (1986) estimate that separation distances could range from 15 to more than 150 m based on decay rates of 0.15 log PFU/day in groundwater.

4.5.1.5: <u>Toxic Organic Compounds</u>

The investigation of toxic organic compounds (toxic organics) and their presence in domestic wastewater discharged to SWIS is relatively recent. Thus, the amount of information available on their occurrence in septic tank effluent is limited. Hathaway (1980) identified several toxic organics that would likely be found in domestic wastewater due to their frequent occurrence in common household products, but few data exist concerning their occurrence in septic tank systems.

A number of toxic organics have been found in septic tank effluent. Most data available are from large community septic tank systems. Toxic organics that have been found to be the most prevalent include 1,4-dichlorobenzene, methylbenzene (toluene), dimethylbenzenes (xylenes), 1,1-dichloroethane, 1,1,1-trichloroethane, and dimethylketone (acetone). These compounds may be found in household products such as solvents, cleaners, and perfumes.

No known studies have been done to determine toxic organic treatment efficiency of single home septic tanks. A study of toxic organics in domestic wastewater and effluent from a community septic tank found that removal of low molecular weight alkylated benzenes was significant where as virtually no removal was noted for higher molecular weight compounds (DeWalle et al., 1985). Removal efficiency was also observed to be positively related to tank detention time.

Studies of toxic organic behavior in unsaturated soil are relatively recent and have been conducted mostly with the attempt at understanding the behavior of petroleum product contamination and pesticides. No known studies have been published which focus on the behavior of toxic organics introduced with septic tank effluent into unsaturated soil.

The avenues of mobility available to toxic organics are gaseous or liquid phases. In the gaseous phase, toxic organics diffuse outward in any direction within unobstructed soil voids. In the liquid phase, they follow movement of the soil solution. In general, toxic organics, because of their mostly nonpolar nature, will not be electrochemically retained in unsaturated soil. Toxic organics may be transformed to less innocuous forms within the soil by indigenous or introduced microorganisms.

Rates of movement within both phases are dependent on soil and toxic organic type. Soils having fine textures, abrupt interfaces of distinctly different textural layers, a lack of fissures and other continuous macropores (interaggregate pores), and low moisture content retard toxic organic movement (Hillel, 1989). Loss of toluene, 1,1,1-trichloroethane, and trichloromethane, in the gaseous phase, from a model drainfield trench/soil system has been found to be less than 10% (Sauer, 1991). However, if gaseous exchange between soil and atmosphere is sufficient, appreciable losses of low molecular weight alkylated benzenes, such as toluene and dimethylbenzene (xylene), can be expected due to their relatively high vapor pressure (Bauman, 1989). Those toxic organics relatively miscible in water can be expected to readily move with soil water. Retention of heavier toxic organics that remain in liquid or solid phases may be tightly bound to soil particles (Preslo et al., 1989). Biodegradation appears to be an efficient removal mechanism. Vigorous biodegradation of halogenated and substituted benzenes was indicated in the unsaturated zone below a single family home septic tank system (Robertson, 1991). Dichlorobenzene was found to be reduced from 3,460 μg/L in the septic tank effluent to 13 μg/L in the groundwater 2 m below the SWIS. Toxic organics have been found to be completely removed from septic tank effluent after passing through a subsurface sand filter (Ganzel, 1991). The system received wastewater from 75 homes. For three years of monitoring, none of the toxic organics detected in septic tank effluent were found in the sand filter effluent.

Once toxic organics reach an aquifer, their movement will generally follow in the direction of groundwater movement. Behavior of each within an aquifer may be different, however. Some will stay near the surface and experience much lateral movement; others, such as aliphatic chlorinated hydrocarbons, will experience greater vertical movement because of their heavier molecular weight (Dagan and Bresler, 1984). Based on this, 1,4-dichlorobenzene, toluene, and xylenes in septic tank effluent would be expected to experience more lateral than vertical movement in an aquifer. 1,1-dichloroethane, 1,1,1-trichloroethane, dichloromethane, and trichloromethane would be expected to show more vertical movement. The degree of toxic organic movement is effected by their degree of solubility in water. Acetone, dichloromethane, trichloromethane, and 1,1-dichloroethane are quite soluble in water and are expected to be very highly mobile; 1,1,1-trichloroethane, toluene, and 1,2-dimethylbenzene (o-xylene) are expected to be moderately mobile; and 1,3-dimethylbenzene (m-xylene), 1,4-dimethylbenzene (p-xylene), and 1,4-dichlorobenzene are expected to have low mobility (Fetter, 1988).

Some investigations have documented toxic organic contamination of surficial aquifers by domestic wastewater discharged from community SWIS. Of VOCs detected in groundwater samples collected in the vicinity of subsurface infiltration systems, Kolega (1989) found trichloromethane, toluene, and 1,1,1-trichloroethane to be most frequently noted and in some of the highest concentrations. Xylenes, dichloroethane, and dichloromethane were also detected along with others. Tomson et al. (1984) also found toxic organic contaminated groundwater in the vicinity of small community and commercial subsurface infiltration systems.

4.5.1.6: Surfactants

Very little research has delineated the occurrence of surfactants within domestic septic tank effluent. Those that have, have focused on methylene blue active substances (MBAS), which are anionic surfactants and compose the bulk of surfactants in household laundry detergent. The most common anionic surfactant used in household laundry detergent is linear alkylbenzenesulfonate (LAS) (Sedlak, 1991; Westall, 1991). The sulfonate ligand imparts a negative charge to the compound and has nearly equal affinity for metal cations or a slight preference for Ca^{2+} and Mg^{2+} (Westall, 1991). Whelan and Titmanis (1982) found a range of LAS concentrations, from 1.2 to 6.5 mg/L, in septic tank effluent. Others have found no MBAS substances in septic tank effluent and concluded that MBAS was removed in the septic tank (Alhajjar et al., 1989).

Cationic and nonionic surfactants are also present in household laundry detergent and fabric softeners, but in negligible amounts (Westall, 1991). No data regarding concentration(s) of these in septic tank effluent have been found.

The behavior of surfactants in unsaturated soil is dependent on surfactant type. It is expected that minimal retention of anionic and nonionic surfactants will occur within unsaturated soil having low organic matter content. In fact, nonionic surfactants have been studied for remediation of soil contaminated with hazardous organic compounds (Vigon and Rubin, 1989). However, the degree of mobility will be subject to soil solution chemistry, organic matter content of the soil, and rate of degradation by soil microorganisms. Soils having high organic matter should favor retention of surfactants due to the lipophilic component of surfactants. Surfactants are readily biodegraded under aerobic conditions and are relatively stable under anaerobic conditions (Westall, 1991). Substantial attenuation of LAS in unsaturated soil beneath a subsurface infiltration system has been demonstrated (Robertson et al., 1989; Shimp et al., 1991). LAS concentrations in the soil solution were found to be 1.6 mg/L whereas system effluent ranged between 10 to 14 mg/L. In groundwater beneath the system, LAS was not detected above the detection limit of 5 μg/L. Cationic surfactants strongly sorb to cation exchange sites of soil particles and organic matter (McAvoy et al., 1991; Westall, 1991). Thus, in general, fine texture soils and soils having high organic matter content will favor retention of these surfactants.

Some investigations have identified the occurrence of MBAS in groundwater (Perlmutter, 1971; Thurman et al., 1986). The type of anionic surfactant was not specifically identified, however, it was surmised that the higher concentrations noted were probably of alkylbenzenesulfonate (ABS) which is degraded by microorganisms at a much slower rate than LAS. There has also been research which showed that all three types of surfactants may be degraded by microorganisms in saturated sediments (Federle and Pastwa, 1988). No investigations have been found that identify cationic or nonionic surfactants within groundwater which originated from subsurface wastewater infiltration systems.

4.5.1.7: Metals

Little information exists regarding metals in septic tank effluent. However, they are undoubtedly present in raw household wastewater. Many commonly used household products contain metals. Other potential sources of metals include vegetable matter and human excreta. Aging interior plumbing systems may also contribute lead, cadmium and copper (Canter and Knox, 1985). Several metals have been found in domestic septage confirming their presence in wastewater. These are listed in Table 4.5.3 with their potential sources.

A recent survey of domestic septic tank liquid constituents was conducted in Elkhart County, Indiana (Watkins, 1991). Several EPA priority pollutant metals were found. Table 4.5.4 presents results of this ongoing survey. Another study identified only copper and zinc within septic tank effluent from five separate residences (Whelan and Titmanis, 1982). Concentration means and ranges of the means were 6.8 and 5 to 12 μg/L (parts per billion or ppb) for copper and 7.6 and 4 to 18 μg/L for zinc, respectively. Other metals analyzed were cadmium, chromium, lead, and mercury. None of these metals were identified at their respective detection limits of 10, 20, 40, and 0.2 μg/L.

The existence of metals in soil is one of complex physical, chemical, and biochemical reactions and interactions. Matthess (1984) and Canter and Knox (1985) provide literature reviews on the soil properties of importance to the fate and transport of metals within unsaturated soil. The primary processes controlling the fixation/mobility potential of metals in subsurface infiltration systems are adsorption on soil particles and/or complexation with organic molecules. Since the amount of naturally occurring organic compounds in the soil below the infiltrative surface typically would be low due to the depth of the infiltrative surface, the cation exchange capacity of the soil and soil solution

TABLE 4.5.3: POTENTIAL SOURCES OF METALS IN DOMESTIC WASTEWATER AND THEIR OBSERVED MEAN CONCENTRATIONS IN SEPTAGE

Metal	Potential Source[a]	Feige et al. (1975)	Bennett et al. (1979)	Segall et al. (1979)
			mg/L	
Antimony	Drugs for care of parasitic diseases, paint pigments	-	-	-
Arsenic	Cotton plant defoliant, weed killer, wood preservative, cattle and sheep dip, aquatic weed control, electronic -semi-conductors, medicine-treatment for amoebic dysentery	-	0.16	-
Cadmium	Solder, lawn treatment, luminescent materials, photo chemicals, textile printing, batteries, ascaricide, paints, pigments	0.2	9.1	0.1
Chromium	Abrasives, tanning chemicals, water repellent textiles, pigments, photo chemicals, textile printing, paints, wood preservatives	-	1.1	0.6
Copper	Fungicides, pigments, textile preservatives, wood preservatives, varnish, paint, photo chemicals, plumbing fixtures	-	8.3	8.7
Lead	Batteries, pigments, paints, solder, glaze, stabilizers for plastic, matches	-	8.4	2.0
Mercury	Insecticides and rodenticides, weed killers, textile preservatives, batteries, antiseptic, pearlescent paint	0.02	0.4	-
Nickel	Coins, jewelry, zippers, plumbing fixtures, corrosion, coverings, dyes, pigments, PVC pigment, fungicide for vegetables, photographic chemicals, skin treatment, diuretics, ointments (skin, eyes), crabgrass control	<1.0	0.7	0.4
Selenium	Photographic chemicals, silver compound antiseptics	-	0.07	-
Silver	Photographic chemicals, silver compound antiseptics	-	-	-
Zinc	Luminescent materials, pigments, rubber compounding, ointments (antiseptic), deodorant, disinfectants, paint, wood preservative	62	30	9.7

a After Hathaway (1980).

TABLE 4.5.3: POTENTIAL SOURCES OF METALS IN DOMESTIC WASTEWATER AND THEIR OBSERVED MEAN CONCENTRATIONS IN SEPTAGE

Metal	Potential Source[a]	Feige et al. (1975)	Bennett et al. (1979)	Segall et al. (1979)
		mg/L		
Antimony	Drugs for care of parasitic diseases, paint pigments	-	-	-
Arsenic	Cotton plant defoliant, weed killer, wood preservative, cattle and sheep dip, aquatic weed control, electronic -semi-conductors, medicine-treatment for amoebic dysentery	-	0.16	-
Cadmium	Solder, lawn treatment, luminescent materials, photo chemicals, textile printing, batteries, ascaricide, paints, pigments	0.2	9.1	0.1
Chromium	Abrasives, tanning chemicals, water repellent textiles, pigments, photo chemicals, textile printing, paints, wood preservatives	-	1.1	0.6
Copper	Fungicides, pigments, textile preservatives, wood preservatives, varnish, paint, photo chemicals, plumbing fixtures	-	8.3	8.7
Lead	Batteries, pigments, paints, solder, glaze, stabilizers for plastic, matches	-	8.4	2.0
Mercury	Insecticides and rodenticides, weed killers, textile preservatives, batteries, antiseptic, pearlescent paint	0.02	0.4	-
Nickel	Coins, jewelry, zippers, plumbing fixtures, corrosion, coverings, dyes, pigments, PVC pigment, fungicide for vegetables, photographic chemicals, skin treatment, diuretics, ointments (skin, eyes), crabgrass control	<1.0	0.7	0.4
Selenium	Photographic chemicals, silver compound antiseptics	-	0.07	-
Silver	Photographic chemicals, silver compound antiseptics	-	-	-
Zinc	Luminescent materials, pigments, rubber compounding, ointments (antiseptic), deodorant, disinfectants, paint, wood preservative	62	30	9.7

a After Hathaway (1980).

TABLE 4.5.4: CONCENTRATION OF METALS IN SEPTIC TANK EFFLUENT (Watkins, 1991)[a]

Metal	Mean Concentration µg/L	Range µg/L
arsenic	37 (5)[b]	6 - 59
barium	890 (5)	400 - 1,310
admium	83 (7)	30 - 330
chromium	320 (7)	60 - 1,400
lead	2,700 (1)	-
mercury	2 (2)	1 - 3
nickel	4,000 (1)	-
selenium	15 (6)	3 - 39

a Samples collected from the outlet end of nine septic tanks.
b Number in parentheses indicates number of septic tanks in which the metals were detected.

pH become the controlling properties for metals mobility below the infiltrative surface. It is likely that movement of metals through the unsaturated zone, if it occurs at all, is accomplished by movement of organic ligand complexes formed at or near the infiltrative surface.

Information regarding the transport and fate of metals in groundwater is limited. One study attempted to link septic tank systems to metal contamination of rural potable water supplies, but only weak correlations were found (Sandhu et al., 1978).

4.5.2: Summary of Documented Conventional OSDS Performance

Septic tank systems are designed to provide wastewater treatment and disposal through soil percolation and groundwater recharge. Satisfactory performance is dependent on the properties of the soil underlying the infiltrative surface. The soil must have adequate pore characteristics, size distribution and continuity to accept the daily volume of wastewater that is applied and to provide sufficient soil/water contact and retention for achievement of acceptable treatment before the percolating wastewater enters groundwater.

Important soil properties include:

* texture (particle size)
* structure (arrangement/aggregation)
* pore size distribution and continuity
* bulk density
* mineralogy
* organic content
* cation exchange capacity
* pH
* moisture content
* redox potential

Satisfactory performance based on monitoring of traditional wastewater parameters (BOD_5, suspended solids, and fecal coliforms) has been shown to occur where an aerobic, unsaturated zone of medium to fine texture soils, 2 to 5 ft in thickness, is maintained below the infiltrative surface during operation. Soils with excessive permeabilities (coarse texture soil or soil with large and continuous pores), low organic matter contents, low pH, low cation exchange capacities and redox potentials, high moisture contents, and low temperatures have been shown to reduce treatment efficiencies.

Groundwater monitoring below properly sited, designed, constructed, and operated subsurface infiltration systems has shown BOD_5, suspended solids, fecal indicators, and surfactants to be effectively removed within 2 to 5 ft in unsaturated, aerobic soil. Phosphorus and metals can be removed through adsorption, ion exchange, and precipitation reactions, but the capacity of soil to retain these ions is finite and varies with soil mineralogy, organic content, pH, redox potential, and cation exchange capacity. The fate and transport of viruses is largely unknown, but evidence is growing that some types of virus are able to leach with wastewater from subsurface infiltration systems to groundwater. Fine texture soil, low hydraulic loadings, aerobic subsoils, and high temperatures favor virus destruction. Toxic organics appear to be removed in aerobic subsoils, but further study of the fate and transport of these compounds is needed. Public health and environmental risks from properly sited, designed, constructed, and operated septic tank systems appear to be low. However, use of conventional septic tank system technology in high density developments or environmentally sensitive areas could increase these risks to unacceptable levels.

Septic tank systems do impact groundwater quality and, therefore, have the potential to impact surface water quality. Studies have shown that after the treated percolate enters groundwater, it remains as a distinct plume for as much as several hundred feet. Solute concentrations can remain above ambient groundwater concentrations within the plume. Attenuation of solute concentrations is dependent on the quantity of natural recharge and travel distance from the source. Organic bottom sediments of surface waters appear to provide some retention or removal of wastewater contaminants. Groundwater must seep through bottom sediments to enter surface waters. Bottom sediments can be effective in removing trace organics, endotoxins, nitrate, and pathogenic agents through biochemical activity. However, few data regarding the effectiveness and significance of removals by bottom sediments are available.

4.6: PERFORMANCE BASED SYSTEMS

Onsite wastewater treatment systems must be designed for the site conditions encountered and the wastewater characteristics expected. Particular site limitations may prevent construction of a conventional onsite system which will perform acceptably. If onsite wastewater treatment is to be provided, an alternative system must be designed. Typically, alternative designs are treated as separate and discrete "black boxes" which are only acceptable for a specific suite of site and soil characteristics. However, alternative designs should be considered as a continuum of design features which allow systems to function acceptably under various site conditions. Appropriate features should be incorporated into the system design as needed to adapt to a particular site. In this manner, the site is not required to possess specific characteristics for a specific design, but rather, the design is adapted for the specific characteristics of the site to meet the hydraulic and treatment performance standards established. In some cases, the site can be modified to eliminate site limitations or the infiltration design changed so that the limitations cease to be a concern. In others, it may be necessary to provide additional pretreatment to meet the desired performance standards. Table 4.6.1 lists some common site limitations and appropriate site or design adaptations for each.

Although many "black boxes" have been developed and used, all are designed to enhance hydraulic or treatment performance. The design features used to enhance performance vary. For improved hydraulic performance, either the hydraulic loading rate is reduced (increased infiltrative surface area), the organic loading rate is reduced (aerobic pretreatment to remove biodegradable organics), or the system is constructed in more

TABLE 4.6.1: SUBSURFACE WASTEWATER INFILTRATION SYSTEM ADAPTATIONS FOR COMMON SITE LIMITATIONS (WPCF, 1990)

SITE LIMITATIONS	ADAPTATIONS
UNSATURATED THICKNESS	
Soil profiles with hydraulically restrictive horizons	Excavate to remove restrictive horizons and backfill with specified fill material OR Elevate infiltrative surface in soil profile or above natural grade in specified fill material
Shallow water tables or seasonally saturated zones within desired design unsaturated zone	Elevate infiltrative surface in soil profile or above natural grade in specified fill material AND/OR Drain subsoil
Shallow creviced or porous bedrock	Elevate infiltrative surface in soil profile or above natural grade in specified fill material
Water table with high "mounding" potential or shallow impervious bedrock within desired design unsaturated zone	Reduce hydraulic loading per unit area of infiltrative surface AND/OR Elevate infiltrative surface in soil profile or above natural grade in specified fill material
SUBSOIL AERATION	
Soils with moderate to high water holding capacity determined by texture, structure, and/or bulk density	Reduce hydraulic and/or organic loading to infiltrative surface AND/OR Reduce width and depth of infiltrative surface AND Reduce dosing frequency OR Elevate infiltrative surface above natural grade in specified fill material
High moisture content due to capillary fringe of shallow water table	Treat as shallow water table
TREATMENT	
Rapidly permeable soils	Construct infiltrative surfaces within specified fill material whether above or below natural grade
Soils with few fines or low cation exchange capacity	Pretreat to remove constituent of concern

permeable soil materials (mounds or fills). For improved treatment performance, either greater unsaturated depth below the infiltrative surface is provided (mounds, fills, subsurface drainage or removal of restrictive soil horizons), or additional pretreatment is provided before the treated water is discharged to the infiltration system (nitrogen removal). Descriptions and design criteria for alternative system designs may be found throughout the literature.

It is important to note that performance based systems may require more active management than conventional systems have typically received. Alternative designs, particularly those that include enhanced pretreatment, are more complex. Trained operators may be necessary to maintain the systems in proper working order. Therefore, provisions must be made for timely and effective third party management.

4.7: SUMMARY OF OSDS PERFORMANCE MONITORING IN FLORIDA

4.7.1: Florida OSDS Research Project

The Florida OSDS Research Project began in 1986 with the goal of determining "whether high density installation of systems, installation of systems under certain soil and water table conditions, and current methods of system installation are polluting state groundwater" (Chapter 381.273 (3), Florida Statutes). This section provides a brief review of the work conducted on this project.

4.7.1.1: Impact of Florida's Growth on the Use of Onsite Sewage Disposal Systems

As discussed previously, Florida has experienced tremendous growth over the past two decades, and serious concerns over the impact of this growth on the environment have resulted. The first efforts on the OSDS Research Project involved an analysis of Florida growth in relation to the use of OSDS.

To accomplish this, estimates of OSDS numbers by county were generated from U.S. Census data, state population projections, and building and OSDS permit data. A detailed survey of county environmental health directors was used to assess the location of future OSDS and to evaluate OSDS practices by county. Critical to the performance of OSDS is the capability of Florida soils to accept and treat septic tank effluent. Before evaluating the capability of Florida soil, the first task of the research project was to identify major soil types used for OSDS. Soil association maps which were available for all counties in 1986 (Florida Bureau of Comprehensive Planning, 1974) were used to estimate on which soils in each county future

OSDS would most likely be developed. Details of this phase of the research can be found in Ayres Associates (1987) and Sherman et al. (1987). Only a brief summary is provided here.

Results of this initial research phase suggested that over 75% of the existing and projected number of OSDS installations would be in 24 of 67 Florida counties. These counties encompassed or bordered Metropolitan Statistical Areas (MSA) suggesting that the majority and highest densities of OSDS were on the urban fringes and outlying areas of Florida cities. Growth data suggested this would continue to be the case in the future unless the rate of sewer construction was increased to serve recent and projected growth in these areas.

This assessment also showed that large areas of Florida have soil conditions unsuitable for conventional OSDS designs. Approximately 74% of state soils have severe or very severe limitations for conventional OSDS designs based on USDA Soil Conservation Service (SCS) criteria. The most common limiting condition is seasonal wetness or shallow groundwater. Slowly permeable soils, shallow bedrock, and periodic flooding are other limiting conditions frequently encountered.

The distribution of soils with limiting conditions varies across the state. Areas having soils best suited for OSDS use are concentrated in the central Florida ridge and upper panhandle regions of the state.

The county health unit survey revealed that severely limiting soils were routinely used for OSDS in "high use" counties. Among the soil series listed as most frequently used for unsewered development, eight were predominant. Five of these series, Paola, Candler, Astatula, Tavares, and Basinger are Quartzipsamments and Psammaquents. Of the remaining three, Myakka and Immokalee are Haplaquods and Apopka is a Paleudult. Nearly all of these are very sandy, highly or excessively permeable, and many are very wet or have seasonally shallow groundwater. Only the Apopka is well drained with moderate permeability.

The survey also indicated that since 1984, conventional trench and bed systems accounted for less than 55% of the OSDS installed in Florida. Mounds and systems constructed in fill accounted for 44%. Many counties reported that mound and fill systems were becoming more common because new developments were frequently located on soils with shallow water tables.

Overall, the results of the survey of OSDS practices in Florida suggested that, increasingly, areas of the state having severely limiting soils were being developed without central sewerage. If OSDS are to function properly on such soils, systems which are designed to overcome the specific limitations must be used. While some counties indicated that such designs were being installed, it was not possible to determine from the survey whether the designs were appropriate for the site conditions encountered. This was particularly true for systems installed prior to the significant revisions to Chapter 10D-6, FAC, made in 1983. It was surmised that if inappropriate designs were used, contamination of groundwaters in Florida may be occurring.

Since this initial phase of the OSDS research project was completed, the 1990 census data have become available (U.S. Department of Commerce, 1993). Table 4.7.1 shows a comparison of Florida housing units from the 1970, 1980, and 1990 census data. These data reveal some interesting trends. Florida housing stock increased tremendously from 1970 to 1990, with an increase of over 3.5 million homes. An increase of 614,000 unsewered homes were estimated to utilize OSDS. However, as the table shows, the percentage of homes estimated on OSDS actually decreased over the 20 year period, from approximately 39 percent in 1970 to 26 percent in 1990.

Comparison of the 1990 census data with the ranking of counties developed during the research effort provides revealing demographic information (Ayres Associates, 1987). Table 4.7.2 shows the 15 counties which were predicted to experience the most new OSDS installations based on 1985 data, and the 15 counties which actually experienced the most increase in OSDS numbers between the 1980 and 1990 census. While the same counties for the most part appear in both lists, there are several significant differences that should be noted. Hillsborough, Dade, Duval, and Seminole counties are each within one of the five largest MSAs in Florida. These counties were projected to be among the top 15 "high use" OSDS counties based on 1985 data, but did not experience sufficient increase in unsewered home numbers from 1980 to 1990 to be included. The reason for this appears to be infrastructure capital improvements programs in which large sewering projects in the mid to late 1980's were completed. In fact, the estimated numbers of existing OSDS significantly decreased in Dade County between 1980 and 1990. Hillsborough County also had fewer OSDS in 1990 than were estimated in 1985. St. Lucie, Charlotte, Lake and Highlands Counties experienced sufficient growth in OSDS use to move into the 1980-1990 list of top 15 counties.

TABLE 4.7.1: ESTIMATED RESIDENTIAL OSDS NUMBERS IN FLORIDA, 1970 TO 1990 (U.S. Census Bureau, 1990; U.S. EPA, 1987) [a]

Year	Total Housing Units	Estimated OSDS Housing Units (number)	(percent of total)
1970	2,526,500	985,800	39%
1980	4,383,100	1,237,400	28%
1990	6,100,262	1,600,469	26%

a Based on the number of unsewered housing units reported in the U.S. Census database.

TABLE 4.7.2: PROJECTED AND ACTUAL RANKINGS OF FLORIDA COUNTIES EXPERIENCING THE HIGHEST INCREASE OF OSDS INSTALLATIONS

Rank	1985 Projected Ranking [a]	Actual Ranking (1980-1990) [b]
1	Marion	Marion
2	Hillsborough [c]	Brevard
3	Polk	Polk
4	Orange	Orange
5	Pasco	Lee
6	Lee	Volusia
7	Volusia	Hernando
8	Brevard	St. Lucie [c]
9	Palm Beach	Charlotte[c]
10	Citrus	Citrus
11	Dade [c]	Pasco
12	Duval [c]	Lake [c]
13	Seminole [c]	Highlands [c]
14	Hernando	Palm Beach
15	Sarasota	Sarasota

a From Table 2, Ayres Associates (1987).
b Based on unsewered housing unit estimates (1990 minus 1980 U.S. Census figures).
c Counties not on both lists.

These results suggest that significant numbers of OSDS are being replaced with public sewers on the urban fringes of Florida cities, and supports the data in Table 4.7.1 showing a decreasing percentage of Florida housing units on OSDS. Nevertheless, large numbers of OSDS continue to be installed in Florida.

4.7.1.2: Computer Modeling Assessment of Groundwater Contamination Potential from OSDS in Selected Hydrogeologic Regions

This phase of the Florida OSDS Research Project was initiated as a screening tool to evaluate the potential impact of OSDS use on groundwater, particularly in locations of high OSDS densities. Evaluation was accomplished through computer simulations of contaminant transport under varying hydrogeologic regimes in Florida. Specifically, the objectives of the modeling effort were to assess the relative potential of various Florida surficial aquifer conditions for contamination from high density use of OSDS. It was hoped that, when combined with data on Florida soils and OSDS use, the results would allow field monitoring sites to be chosen with priority given to regions of high density OSDS use most susceptible to contamination. This section describes the key results obtained from the modeling effort. Further details of this phase of the project can be found in Anderson et al. (1988) and Kirkner and Associates (1987).

Results of this phase of the research showed that the surficial hydrogeology in Florida is quite varied despite the large areas of sandy surface soils and general lack of topographical relief. Surficial aquifers in Florida range from the very shallow unconfined limestone of the Biscayne Aquifer in the southeast; to medium to fine sand and clayey sand water tables in the central peninsula, northern panhandle, and coastal areas; to the highly productive sand and gravel aquifer of the western panhandle.

Contaminant Source: To compare the contamination potential of various hydrogeologies from OSDS use, a "model subdivision" was assumed as the sole contaminant source. The characteristics of the subdivision are listed in Table 4.7.3.

TABLE 4.7.3: CHARACTERISTICS OF MODEL SUBDIVISION

- 50 acre subdivision size
- 200 homes maximum on OSDS (0.25 acre lots)
- 3 persons per home
- 45 gal/cap/day flow of septic tank effluent
- 50 mg/L NO_3-nitrogen enters groundwater

The homes in the subdivision were each assumed to be contributing typical quantities and quality of septic tank effluent at point discharges spread evenly across the subdivision. Although the transport of other parameters were modeled, the results for nitrate nitrogen yielded the most useful data for comparison of hydrogeologic regions and are the only results discussed in detail here. For the purposes of this study, it was assumed that all organic and ammonia nitrogen from the septic tank were nitrified by soil bacteria during wastewater infiltration through the vadose zone. A value of 50 mg/L $N0_3$-N was used as a conservatively high but realistic value of nitrogen input from each OSDS to groundwater below the subdivision.

Transport Model: An analytical solution to the advection-dispersion equation was used to model the transport of pollutants through the water table aquifer below and down-gradient of the subdivision. For the purpose of comparing various hydrogeologic regions, it was decided to model transport in a steady-state, one-dimensional flow field with three-dimensional dispersion, retardation, and first order decay. For conservative parameters such as nitrate, retardation and decay were both assumed to be zero. The equation used describes the transport of a solute through porous aquifer material, and yields a typical groundwater contamination "plume" downgradient from the source.

The primary means of transport for nitrate is advection as measured by the seepage velocity determined from the hydraulic conductivity, hydraulic gradient, and effective porosity of the aquifer. Hydraulic conductivity is a measure of the resistance to flow through a porous material while hydraulic gradient is the slope of the water table surface. Effective porosity is the fraction of interconnected void space.

Results: The modeling methods discussed above were used to assess the contamination potential of eight different surficial aquifer conditions characteristic of Florida. For brevity, portions of the results for two of these conditions are presented in a subjective fashion. A more comprehensive discussion of results is available elsewhere (Kirkner and Associates, 1987).

Contaminant concentration diagrams presented in this section are based on the model subdivision as the sole contaminant source and a time interval of 5 years since onset of effluent percolation to groundwater. While other sources of contamination would surely be present, the assumption used here allows a comparison of OSDS effects between groundwater regions. The vertical concentrations of contaminants are averaged over the upper 10 ft of the surficial aquifer which provides a mixing zone for dilution of contaminant concentrations. Nitrate nitrogen was used as the contaminant for comparative purposes. These assumptions yield conservatively high estimates of contaminant concentrations but simplify discussion of the modeling results. Maximum housing densities of 4 homes/acre were compared to results of 2 homes/acre to examine the theoretical effect of housing density.

Based on the methods and assumptions reported above, Figure 4.7.1 shows nitrate-nitrogen concentration contours generated by the transport model for example groundwater region 1. This region was characterized as having relatively high dispersivities and low seepage velocities which are typical of the shallow, flat water tables of the flatwood areas of central and south Florida. Mean dispersivity values used were 60 ft, 15 ft, 1.2 ft for longitudinal, transverse, and vertical dispersivity, respectively. A uniform seepage velocity of 0.12 ft/day resulted from the mean input values used for hydraulic conductivity, gradient, and porosity.

As Figure 4.7.1 shows, the transport model estimated groundwater concentrations in excess of the drinking water standard of 10 mg/L NO_3-N under almost all of the subdivision area. Concentrations of 20 mg/L NO_3-N were estimated over a somewhat smaller area at housing densities of 4 homes/acre. The effect of reducing housing density to 2 homes/acre eliminated the 20 mg/L NO_3-N contour all together and somewhat reduced the area of the 10 mg/L zone. The low seepage velocity used in this example limits downgradient contaminant movement and dispersion, which are velocity dependent.

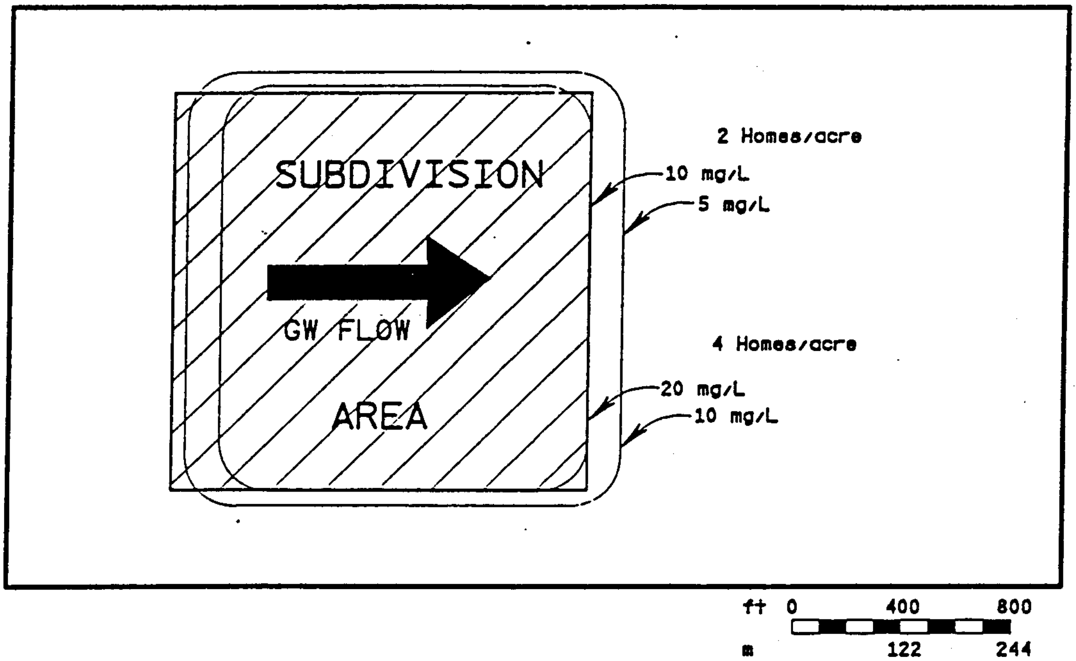

Figure 4.7.1: NO_3-N Contours at 5 Years Based on Model Results for Mean Input Parameters, Region 1 (High Dispersivity, Low Seepage Velocity)

In this example, the time interval of 5 years gave a misleading estimate of ultimate NO_3-N concentrations. At the mean seepage velocity of only 0.12 ft/day, groundwater under the subdivision only traveled approximately 215 ft, thus limiting the number of homes which could have impacted it. Therefore, significantly higher NO_3-N concentrations were estimated at longer time intervals as more homes discharged nitrate to groundwater.

Figure 4.7.2 shows similarly generated concentration contours for example groundwater region 2. This region was characterized as having relatively low dispersivities and high seepage velocities, which are typical of flat, very porous limestone water tables such as the Biscayne aquifer in southeast Florida. Mean dispersivity inputs were 30 ft, 3 ft, and 0.3 ft for longitudinal, transverse, and vertical dispersivity, respectively. A seepage velocity of 1.2 ft/day resulted from the input values of hydraulic conductivity, gradient, and porosity which characterized this hydrogeologic region.

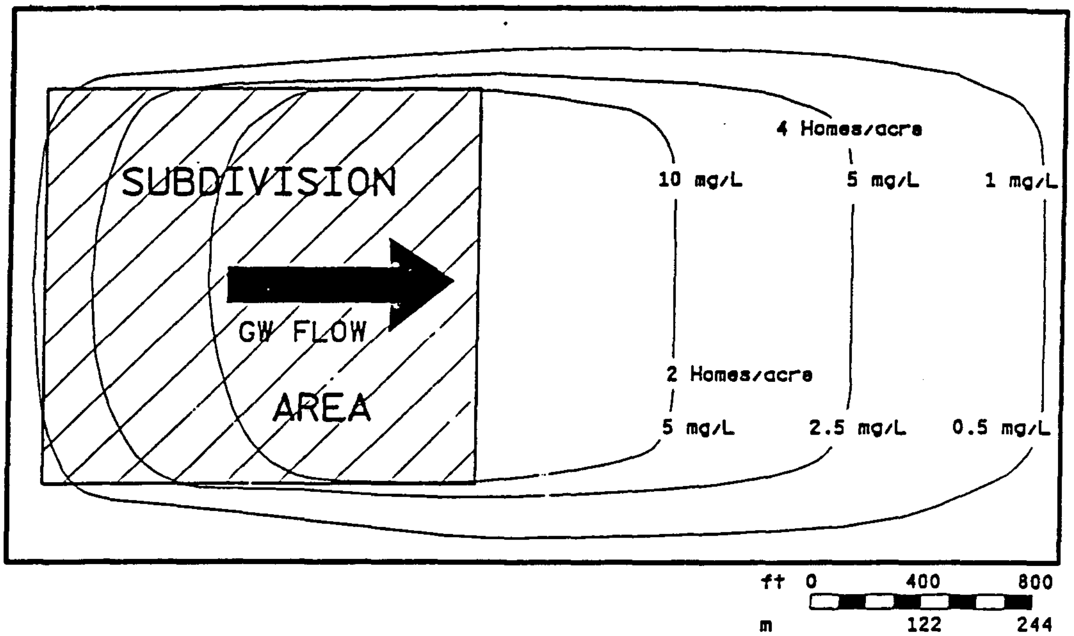

Figure 4.7.2: NO_3-N Contours at 5 Years Based on Model Results for Mean Input Parameters, Region 2 (Low Dispersivity, High Seepage Velocity)

The differences between regions 1 and 2 are obvious when comparing concentration contour maps (Figures 4.7.1 and 4.7.2). The 10 mg/L NO_3-N contour in region 2 moved downgradient much farther than in region 1 due to the higher mean seepage velocity. Although dispersivities were significantly lower in region 2 than region 1, lateral dispersion from the subdivision was approximately the same as evidenced by the width of the 10 mg/L contour at the 4 homes/acre housing density. This is due to the fact that dispersion is directly related to velocity for a given aquifer media. Figure 4.7.2 does not include a 20 mg/L NO_3-N contour at 4 homes/acre as was present in the results of the region 1 analysis. Since porosity of the two regions was similar, the higher seepage velocity and increased dilution caused by greater groundwater flow passing beneath the subdivision prevented concentrations from reaching 20 mg/L. A reduction in housing density to 2 homes/acre was estimated by the model to yield NO_3-N concentrations below the 10 mg/L drinking water standard in Region 2.

The time interval of 5 years was considered a more suitable modeling period for region 2, in contrast to the region 1 results. At a mean seepage velocity of 1.2 ft/day, groundwater from below the subdivision traveled approximately 2,137 ft over this time interval. This is larger than the longitudinal dimension of the model subdivision by about 650 ft. Thus, the NO_3-N concentrations estimated in this case should be much closer to potential maximum concentrations for the conditions simulated.

Results from this phase of the study show that the effect of seepage velocity on concentrations and contaminant spread are important for the assessment of contamination potential of surficial aquifers in Florida. Although there are conditions of very high seepage velocities in several Florida locations (most notably the Biscayne aquifer in southeast Florida and the sand and gravel aquifer in northwest Florida), the majority of water table conditions in the state are at very low hydraulic gradients and resulting low seepage velocities. At very low seepage velocities, the model indicated that considerable time must pass before the true effects of an OSDS development on groundwater quality will be realized. For the region 1 conditions for example, it would perhaps be 30 years or more before the maximum contamination potential of a conservative parameter such as nitrate was realized downgradient of the subdivision, based on the modeling results.

The results in Figures 4.7.1 and 4.7.2 suggest that concerns over nitrate contamination of groundwaters from large, densely developed OSDS subdivisions in Florida are not unfounded. Even in example region 2, where high seepage velocities were present for dilution, model results indicated that typical housing densities used in Florida may cause exceedance of the nitrate standard below and downgradient of larger subdivisions. The modeling results suggest that subsequent field monitoring should include relatively large subdivisions with densities of 4 houses/acre located in several areas of the state, and include a range of conditions from a shallow, flatwoods area to a porous limestone area in southeast Florida.

4.7.1.3: Groundwater Monitoring of Selected Florida Subdivisions Utilizing OSDS

The general assessment of Florida soils and surficial hydrogeology completed in the first two phases of the research project formed the framework to select subdivisions for field monitoring of OSDS impacts. Based on results of the assessments, it was desired to monitor four subdivisions which provided a cross section of soil and groundwater conditions most likely to be utilized for future OSDS installations in the state. Soils representative of the Florida Flatwoods, Central Florida Ridge, South Florida/Everglades and the Southern Coastal Plain (upper panhandle) were all desired, as well as groundwater conditions which ranged from very flat, shallow, slow moving sandy water table aquifers to very porous, fast moving limestone aquifers such as the Biscayne. In addition, it was desired to utilize subdivisions which were relatively isolated from other developments and were developed after the 1983 revisions to Chapter 10D-6.

Numerous subdivisions were visited across the state in an attempt to locate sites which met the above criteria. It became clear soon after the selection process began, however, that locating the four subdivisions would be very difficult. Isolated subdivisions built after 1983 with all lots developed were extremely difficult to find. It was finally decided to look for older subdivisions that were relatively isolated and likely had OSDS installations that met the 2-foot separation from groundwater as required by Chapter 10D-6.

Four subdivisions on varied soil and groundwater conditions were eventually located. The locations of the subdivisions are shown on Figure 4.7.3. Subdivision sites chosen included a high, well drained sand ridge setting in Polk County; a low, moderate to somewhat poorly drained flatwoods area in St. Johns County; a low, poorly drained flatwoods area farther south in Brevard County, and a shallow limestone aquifer (Biscayne) in Dade County. These subdivisions were chosen since they were thought to have been developed under the requirements of the 1983 revisions to Chapter 10D-6, even though several were developed before that date. This section provides a brief summary of the subdivision groundwater monitoring study. Further details can be found elsewhere (Ayres Associates, 1989, 1993).

Fieldwork in the subdivisions began in mid-1987 and included the following activities:

- Soil and shallow geologic investigations via soil borings.

- Temporary piezometer well installation and collection of water level and field water quality data.

- Electromagnetic (EM) terrain conductivity surveys to aid in the placement of permanent monitoring wells.

- Ground penetrating radar (GPR) surveys at two sites to enhance the understanding of the shallow geology.

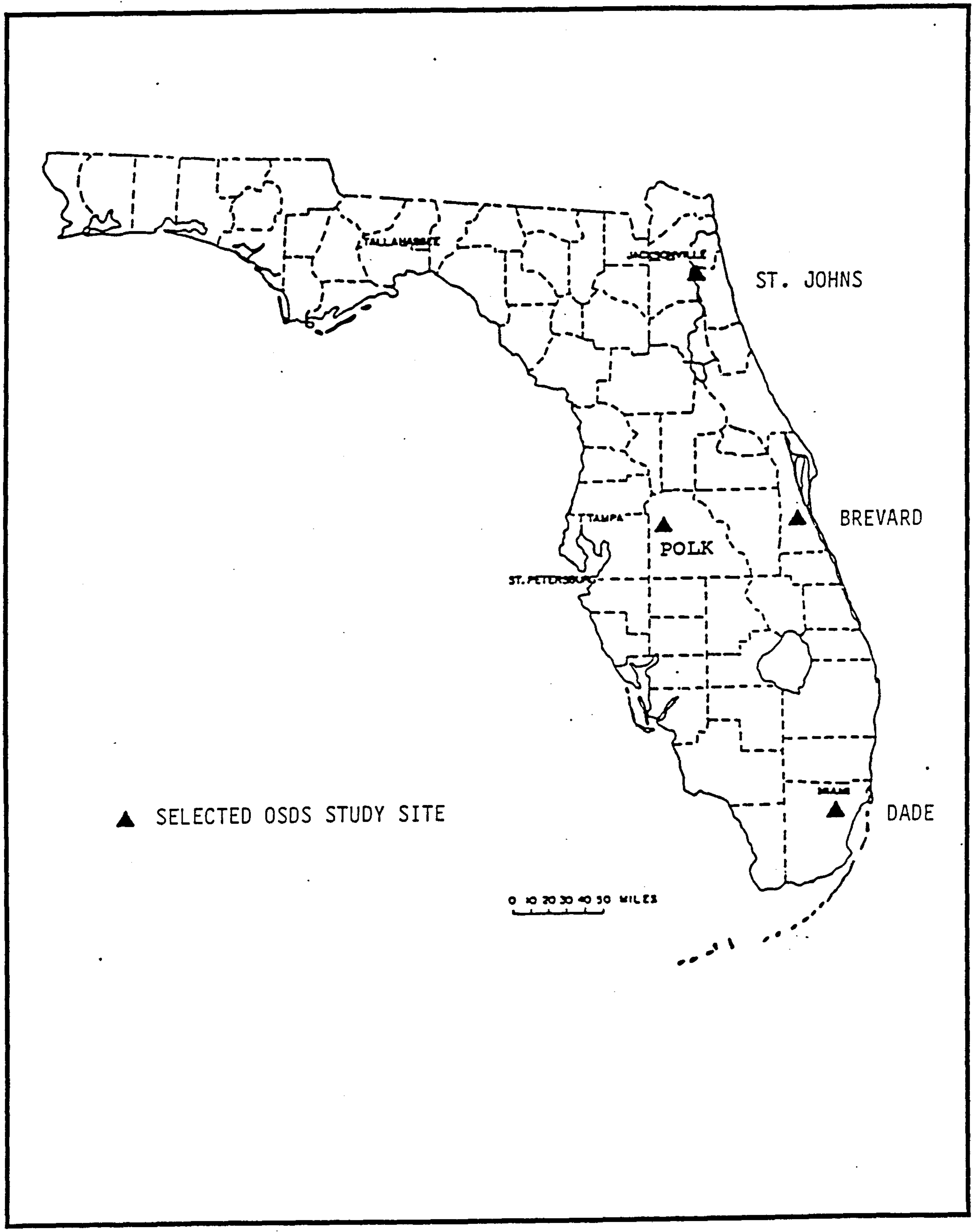

Figure 4.7.3: Location of Subdivisions Selected for Study.

- Preliminary data evaluation and site selection for more extensive groundwater sampling and analysis.

- Monitoring well installation, development and sampling.

- Groundwater monitoring data evaluation and interpretation.

At least 10 test borings were made at each site. Temporary piezometers were constructed in several of the test borings. Based on soil boring data, surface geophysical surveys and piezometer measurements, permanent groundwater monitoring wells were sited and installed. Seven wells were placed in each subdivision with the exception of the Polk County subdivision where only 5 were installed. Subdivision groundwater monitoring activities took place between May 1987 and November 1990.

The water table aquifers studied exist in fine sands beneath the subdivisions in Polk, St. Johns, and Brevard Counties, while in Dade County, the aquifer exists in shallow limestone. The depth to groundwater varied widely among the four subdivision sites: 1 to 4 ft below ground surface in Brevard County, 3 to 5 ft in Dade County, 2 to 7 ft in St. Johns County, and 9 to 18 ft in Polk County. Several OSDS drainfields in both the St. Johns and Brevard County subdivisions were determined to be less than 2 ft from groundwater during high seasonal water tables. It was suspected that this applied to numerous other drainfields in those subdivisions as well.

The observed horizontal groundwater gradient in all of the subdivisions was normally quite low, typically less than 0.4%. Estimates of groundwater seepage velocities yielded rates typically well below 25 ft/yr in the shallow groundwater below subdivisions in Polk, St. Johns, and Brevard Counties. Seepage rate in the groundwater below the subdivision in Dade County was considerably higher (670 ft/yr) due to the cavernous and vugular limestone aquifer.

Groundwater monitoring included the measurement of water table elevations and collection of samples for water quality analysis. Analyses were made onsite for temperature, pH, and specific conductance. Laboratory analyses were made for total dissolved solids (TDS), chlorides, 5-day biochemical oxygen demand (BOD_5), total Kjeldahl nitrogen (TKN), nitrate nitrogen (NO_3-N), sulfate (SO_4), total phosphorus (P), surfactants (MBAS), fecal coliform bacteria, and volatile organic compounds (VOCs).

The groundwater concentrations of many constituents varied widely between different wells on a given date and at a given well on different dates. Variations of an order of magnitude or more for pH, chlorides, nitrogen and phosphorus were not uncommon. The fluctuations were not consistent across all parameters ruling out simple dilution from precipitation events as a probable cause.

Paired wells located in close proximity to OSDS revealed notable concentrations of constituents commonly associated with septic tank effluent (STE), e.g. BOD5, TKN, P. These constituents are not exclusively derived from STE, and have other anthropogenic and natural sources in the environment. Nevertheless, concentrations were high enough in samples from certain wells in all four subdivisions to suggest that STE was the source. For example, in the subdivisions in Brevard and Dade Counties, concentrations of BOD_5 and TKN were routinely in the several parts per million (ppm) range. In the Polk and St. Johns County subdivisions, NO_3-N concentrations were above the 10 mg/L water quality standard in several wells downgradient from individual OSDS on several occasions.

Fecal coliform bacteria were detected on one or more occasions in samples from at least one well in each of the four subdivisions. In St. Johns County, a single sample from a single well revealed very low bacteria concentrations of 4 organisms/100 mL. In Polk County, one sample from each of three downgradient monitoring wells yielded fecal coliforms from 10 to 360 organisms/100 mL. In the subdivisions in Brevard and Dade Counties, samples from three and four wells, respectively, revealed concentrations of fecal coliforms, with numbers in Dade County as high as 17,000 organisms/100 mL.

VOCs were not detected in any of the groundwater samples collected (method detection limits typically 5 μg/L or less), with the exception of one sample in St. Johns County (1.8 μg/L of chloroform).

The groundwater monitoring data suggest that the impacts measured were from individual OSDS and/or small groups of OSDS, rather than the subdivision as a whole. Based on the low seepage velocities calculated in the Polk, St. Johns, and Brevard County subdivisions and the previous modeling results, migration of even mobile contaminants (e.g. chlorides, nitrates) would be expected to be limited since the subdivisions monitored were relatively young in age (i.e. < 20 years old). In these settings, the downgradient, horizontal distances that contaminants theoretically could travel were correspondingly low. As a result, these younger subdivisions may not exhibit single plumes, but rather many individual plumes, possibly from each household.

Thus, the basic conclusion drawn from the monitoring of groundwater beneath the subdivisions was that localized areas of groundwater impacted by OSDS were measured but no widespread downgradient impacts from whole subdivisions were noted. Therefore, additional study of individual OSDS was recommended to better define the impacts to groundwater.

4.7.1.4: Performance Monitoring of Individual OSDS in Florida Subdivisions

Results of the subdivision groundwater monitoring suggest that a better estimate of OSDS impact could be obtained by more detailed monitoring of individual systems and projecting the results to obtain cumulative estimates of subdivision-wide impacts. The St. Johns and Polk County subdivisions were chosen for this effort based on several reasons. First, both subdivisions are located on fine, sandy soils. However, the Polk County subdivision soils are very well drained with groundwater deeper than 9 ft while the St. Johns County soils are somewhat poorly drained with groundwater between 2 and 7 ft below ground surface. This would allow a comparison of OSDS performance in sandy soils with a thick and minimal unsaturated zones.

The monitoring of individual OSDS occurred at four homes in each of the two subdivisions. At each home the research effort included the following:

- Household and OSDS characterization.
- Soils characterization.
- STE characterization.
- Septage characterization.
- OSDS infiltration operation monitoring.

In addition, at two of the homes in each subdivision, soil sampling at the infiltrative surface and at several depths beneath it was conducted to evaluate transport of various parameters. This section presents a brief summary of the monitoring results. Further details can be found in Ayres Associates (1989, 1993) and Sherman et al. (1991).

Wastewater and septage characterization included analyses for a suite of constituents including: temperature, pH, specific conductance, chlorides, five day biochemical oxygen demand (BOD_5), total dissolved solids (TDS), total suspended solids (TSS), fats, oils and greases (FOG), total kjeldahl nitrogen (TKN), nitrate nitrogen (NO_3-N), total phosphorus (TP), surfactants (MBAS), and fecal coliform bacteria. OSDS infiltration monitoring included periodic measurement of ponding occurrences, and depth and the range of unsaturated soil beneath the infiltrative surface. Soil samples collected beneath infiltrative surfaces were analyzed for soil moisture content, total organic carbon (TOC), TKN, NO_3, TP, leachable ortho-phosphorus, VOCs, and fecal coliform bacteria. Figure 4.7.4 shows a diagram of sampling locations.

Table 4.7.3 shows results of STE monitoring for conventional parameters at eight homes in the two study subdivisions. STE concentrations of organics, solids, nutrients, and bacteria were found to be within the range of those reported from other locations in the USA and generally agreed well with the values in Table 4.3.5. A notable exception was STE temperature which would be expected to be higher in Florida.

Relatively low concentrations of VOCs were measured in most STE samples. The average total VOC concentration at each home ranged from 9 to 75 µg/L. Toluene was found in almost every sample, while chloroform and methylene chloride were often detected. At one home, 1,4-dichlorobenzene was detected.

Analysis of septage samples at each of the eight homes revealed characteristics consistently lower than those reported in the literature. This may have been due in part to the relatively high septage temperatures observed (27 to 32°C) which could result in increased digestion of solids. Concentrations of most constituents in septage (e.g. BOD5, TKN, TP, VOCs) were about five to ten times higher than those in the STE. Concentrations of TSS and fats, oils, and greases were approximately twenty times higher.

OSDS infiltrative surfaces in the St. Johns County subdivision were commonly closer than 2 ft to groundwater during periods of the year. Monitoring well data at one of the four study homes indicated that the infiltration system was within the saturated zone at various times during this study. Since most homes in the subdivision were at similar elevations, these data indicate that the impacts noted during groundwater monitoring could have been due to upgradient systems which did not meet the 2-foot separation to groundwater required by code.

Several constituents of STE were measured in soil samples collected at the infiltrative surface, and 2 and 4 ft below the infiltrative surface. Concentrations decreased a considerable amount as depth below the infiltrative surface increased within unsaturated soils. Results indicated that several parameters, including phosphorus and nitrogen, could enter groundwater if they were present 2 ft below the infiltrative surface. Significantly greater concentrations of all parameters were observed in soil samples collected from beneath the infiltrative surfaces close to the septic tank, compared to samples collected 10 to 15 ft farther away. Fecal coliform bacteria were found

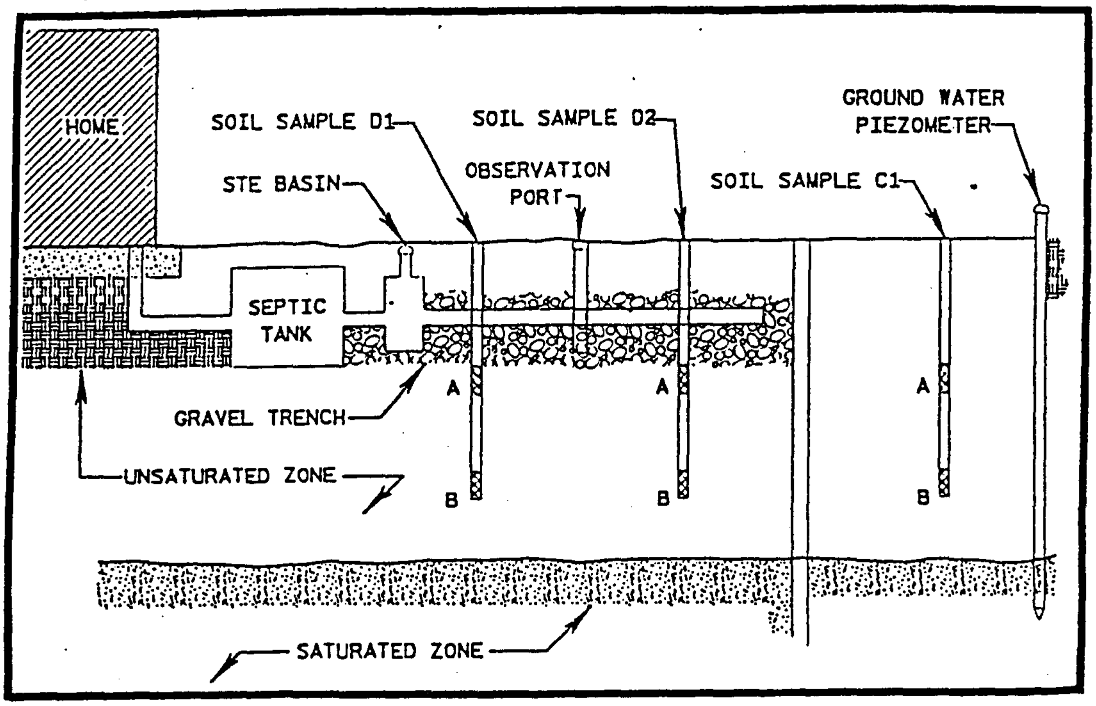

Figure 4.7.4: Profile Schematic of OSDS Sampling Locations

at the 2-foot depth below one of the sampled systems (St. Johns County). TOC and MBAS were substantially reduced within 2 ft, and VOCs were not measured above detection limits in samples 2 ft or more below infiltrative surfaces of the OSDS studied.

The results of the individual OSDS monitoring indicate that the presence of at least 2 ft of unsaturated, fine sandy soil provides a relatively high degree of treatment of most STE constituents. Additions of nitrogen to groundwater occurs as expected, and it appears at systems such as those studied that phosphorus would impact groundwater in sandy soils with only 2 ft of unsaturated soil below the infiltrative surface several years after system startup. Poor distribution of STE over the infiltrative surface substantially increases the impact. Significantly higher concentrations were observed in soil samples collected from near the septic tank where most effluent volume infiltrated.

It appears that in the hydrogeologic settings examined, high OSDS densities in relatively new subdivisions have not resulted in higher degrees of groundwater contamination than might be found immediately adjacent to individual systems in similar, but less densely populated areas. In contrast to the earlier groundwater modeling results, it appears that if subdivision-wide impacts are to occur, it may take decades to manifest the impacts due to low groundwater seepage velocities.

TABLE 4.7.4: CONCENTRATIONS OF SELECTED CONSTITUENTS IN STE AS MEASURED IN THIS STUDY

PARAMETERS (Units)	CONCENTRATIONS
Temperature (C°)	
Avg.	-
Range	18.0 - 33.8
No. Samples	44
BOD_5 (mg/L)	
Ave.	141
Range	111 - 181
No. Samples	36
TSS (mg/L)	
Ave.	161
Range	64 - 594
No. Samples	36
TKN (mg-N/L)	
Ave.	39
Range	36 - 54
No. Samples	36
NO_2+NO_3 (mg-N/L)	
Ave.	0.08
Range	0.6 - 0.14
No. Samples	36
P (mg-P/L)	
Ave.	11
Range	7 - 15
no. Samples	36
FOG (mg/L)	
Ave.	36
Range	8 - 111
No. Samples	19
MBAS (mg/L)	
Ave.	3.1
Range	1.3 - 6.8
No. Samples	34
F. coliforms (log#/L)	
Ave.	-
Range	5.1 - 8.2
No. Samples	36

4.7.1.5: <u>Human Enterovirus Monitoring of Subdivisions and Individual Homes</u>
This phase of the Florida OSDS Research Project focused on viral monitoring of OSDS in two of the subdivisions discussed previously, and was conducted in cooperation with the HRS Tampa Branch Laboratory (formerly HRS Epidemiology Research Center). At the Polk and St. Johns County subdivisions, groundwater monitoring for human enterovirus and an evaluation of potential viral movement through the septic tanks and below the infiltration systems of several individual OSDS were conducted. This section presents a brief review of the findings of this research effort. A more detailed discussion of the methods and results can be found elsewhere (Anderson et al., 1991; Lewis and Stark, 1993).

Subdivision Groundwater Monitoring: The objective of this monitoring phase was to determine if significant transport of enterovirus was occurring in the groundwater downgradient from the subdivisions previously studied. The same monitoring wells were used for this study although samples were collected on different dates.

Four down gradient monitoring wells in the Polk County subdivision were sampled monthly over a 16 month period, from February 1988 to June 1989. Four 378 liter (100 gallon) replicate samples were processed for each sampling event by pumping groundwater through a mobile laboratory for preprocessing as described in Anderson et al. (1991). A total of 24,192 liters (6,400 gallons) of groundwater was analyzed for each of the four wells during the study, 96,768 liters (25,600 gallons) in total. No human enteroviruses were detected in groundwater samples.

In the St. Johns County subdivision, four down gradient monitoring wells were sampled, twelve times each between March 1988 and July 1989. Three to six 378 liter replicate samples were processed during each sampling event. A total of 19,656 liters (5,200 gallons) of groundwater was analyzed for each of the wells during the study, 78,624 liters (20,800 gallons) in total. As in the Polk County subdivision, no human enteroviruses were detected in groundwater samples down gradient of the subdivision.

To evaluate the significance of these results, one must examine the results of groundwater monitoring for other parameters. One of the key conclusions of the earlier subdivision groundwater monitoring was that the impacts of OSDS subdivisions as a whole are difficult to measure under typical water table conditions in Florida (Ayres Associates, 1989; Sherman and Anderson, 1991). The reasons for this are the flat water table gradients typical in much of Florida and the resulting low groundwater seepage velocities.

Analysis of groundwater monitoring data generated from the two subdivisions indicates that any impacts measured from the OSDS are likely to have been from individual systems within several hundred feet of a given monitoring well, rather than from the subdivision as a whole because of the low seepage velocities. Nevertheless, it was suspected that several of the wells in the two subdivisions were impacted by nearby OSDS. In Polk County, two of the four down gradient wells monitored showed elevated levels of parameters commonly associated with STE, such as TDS, chloride, nitrogen species, phosphorus, and SO_4. Similarly, three of the four down gradient wells monitored in the St. Johns subdivision showed elevated levels of the same parameters. In both subdivisions, fecal coliform bacteria were detected on only one occasion, and in only one of the wells in the St. Johns subdivision and three of the wells in the Polk County subdivision.

It was difficult to make firm conclusions from the data generated about the occurrence of viruses in groundwater from OSDS. However, the number and volume of groundwater samples analyzed for enterovirus in this phase of study appear to indicate that groundwater transport of enterovirus down gradient from OSDS is more retarded than the transport of more conventional wastewater constituents.

Individual OSDS Monitoring: The objectives of this aspect of the viral study were to investigate the presence of human enterovirus in fecal stool specimens, STE samples, soil samples, and groundwater samples at single family residences in the two previously studied subdivisions. Eight septic systems at homes in the two subdivisions were studied with varying degrees of detail. The methods and results of fecal viral analyses can be found in Lewis and Stark (1993).

STE samples were collected from the eight study homes as outlined in Anderson et al.(1991) from January 1988 to April 1991. Results of viral analyses of these samples are summarized in Table 4.7.5.

Two hundred seventy (270) STE samples were collected and analyzed over the study period. Of these, 55 yielded a positive identification of a viral agent. Viruses serotyped from the STE samples included Coxsackievirus, Poliovirus, Echovirus, and Reovirus. Viral quantities exiting the septic tanks expressed as most probable number of infectious units per liter (MPN-IU/L), ranged from 0.06 to greater than 43.7.

TABLE 4.7.5 RESULTS OF SEPTIC TANK EFFLUENT VIRUS MONITORING

Home ID	Number Samples	Number Positive	Virus Serotyped[a]	Virus Quantity Range (MPN•IU/L)[b]
1-1	33	1	Coxsackie B1	0.09
		1	Polio 3	0.07
		1	ECHO 6	0.38
1-2	39	3	ECHO 14	0.80 - > 5.03
		3	ECHO 12	0.21 - > 5.68
		3	Coxsackie B2	0.27 - >43.7
		5	REO	5.5 - >30.0
		1	ECHO 6	0.48
1-3	35	2	ECHO 14	0.14 - 0.29
		1	ECHO 12	0.14
		1	Polio 1 & 2	0.61
		4	Coxsackie A9	0.3 - 2.9
		1	Coxsackie B3	0.50
		1	ECHO 22/23	0.14
		1	Coxsackie B4	2.9
1-4	43	1	ECHO 14	0.19
		2	Polio 1 & 2	> 5.30 - 21.5
		1	Polio 3	5.62
		1	Polio 1, 2 & 3	0.49
		7	Coxsackie B4	0.09 - >12.4
		1	ECHO 6	> 3.79
2-1	36	0	N/A	N/A
2-2	40	1	Coxsackie B4	0.22
		3	Coxsackie B5	0.11 - > 2.00
		4	Coxsackie B3	0.10 - 1.71
		3	Coxsackie A9	0.19 - >20.7
		1	REO	> 1.51
		1	ECHO 9	0.68
2-3	36	0	N/A	N/A
2-4	8	0	N/A	N/A
TOTALS	270	55	See Table	0.06 - >43.70

a Coxsackievirus, Poliovirus, ECHOvirus, REOvirus.

b Most Probable Number of Infectious Units per liter.

On several occasions, the same human enteroviral serotype found in fecal specimens from a resident child was subsequently isolated in septic tank effluent from the same household. Table 4.7.6 gives an example of the sequence of events at Home 1-4 in the Polk County subdivision. Coxsackievirus B4 was detected in stool samples from the home from February 19 to March 19, 1990. The same viral serotype was detected in STE initially on February 19 and in each of the six sampling events through April 30, a period of approximately 10 weeks. During the time when the household residents were excreting Coxsackievirus B4, the agent in STE was consistently present in numbers too numerous to estimate. The disappearance of detectable virus from the feces of the residents was followed by a measurable progressive decline in detectable viruses exiting the septic tank. Coxsackievirus B4 was no longer detectable in the septic tank effluent six weeks after the last positive stool specimen. Similar results were obtained at several other households in the study.

Soil samples were collected from below the infiltration trenches of the OSDS at Home 1-4 in Polk County after viruses were detected in feces and septic tank effluent at the home. At the Polk County home, soil cores were collected on March 28, 1990 after Coxsackievirus B4 was detected in feces and STE. Two soil cores were collected to a depth of 30 inches below the infiltrative surface, one from each of two infiltration trenches. Viral analyses were conducted on samples collected from various depths of each soil core. Coxsackievirus B4 was isolated from both trenches at the four inch depth below the infiltrative surface at 0.015 and 0.014 MPN-IU per gram of soil, but not at any greater depth. At the time soil cores were collected, the same viral serotype in the STE was greater than 12 MPN-IU per liter. These results suggest that the viral particles were attenuated by the sandy soil immediately below the infiltrative surface. At this home, depth to groundwater was greater than 15 ft. Thus, soil below the system was relatively dry.

Similar soil cores were collected at Home 2-2 in the St. Johns County on February 26, 1990, after Coxsackievirus B3 was isolated in feces and STE. No virus was isolated from either of two soil cores collected to a depth of 70 inches below the infiltrative surface. However, virus was last detected in STE at this home 21 days prior to the soil core sampling.

Groundwater monitoring for viruses was conducted in close proximity to the wastewater infiltration area at the OSDS serving Home 2-2. The infiltration system consisted of three 2-foot wide gravel filled trenches. Two groundwater monitoring wells were installed by hand auger, one well (W1) directly below the infiltration area between two infiltration trenches and the

TABLE 4.7.6: COMPARISON OF STOOL SAMPLE AND STE SAMPLE RESULTS, HOME 1-4.

Sample Date	Stool Sample Results	Serotype	STE Results (MPN•IU/L)
1/23/90	-----[a]	Neg.[b]	0
1/25/90	Neg.	-----	-----
2/19/90	Coxsackie B4	Coxsackie B4	> 7.21
3/ 5/90	-----	Coxsackie B4	> 9.76
3/ 7/90	Coxsackie B4	-----	-----
3/ 9/90	Coxsackie B4	-----	-----
3/12/90	Coxsackie B4	-----	-----
3/19/90	Coxsackie B4	-----	-----
3/28/90	-----	Coxsackie B4	>12.30
4/ 2/90	Neg.	----	-----
4/16/90	-----	Coxsackie B4	1.45
4/17/90	-----	Coxsackie B4	0.32
4/30/90	-----	Coxsackie B4	0.09
5/ 1/90	-----	Neg.	0

a No sample.
b Negative result, no viral agent detected.

second (W2) approximately 10 ft down gradient of the first, within 2 ft of the most downgradient infiltration trench. Figure 4.7.5 shows a plan view of the OSDS at Home 2-2 with locations of the monitoring wells and groundwater flow direction indicated.

Groundwater monitoring was initiated after viral agents were successfully isolated from both feces and STE samples and was continued for over one year. Home 2-2 was selected for this research due to its location in fine sandy soils with a water table varying from 4 to 5 ft below grade, 2 to 3 ft below the infiltrative surface. Groundwater from wells W1 and W2 were sampled for enteroviruses from February 1990 to April 1991. Routinely, six 378 liter replicates were pumped from each well for each sampling event as previously described,

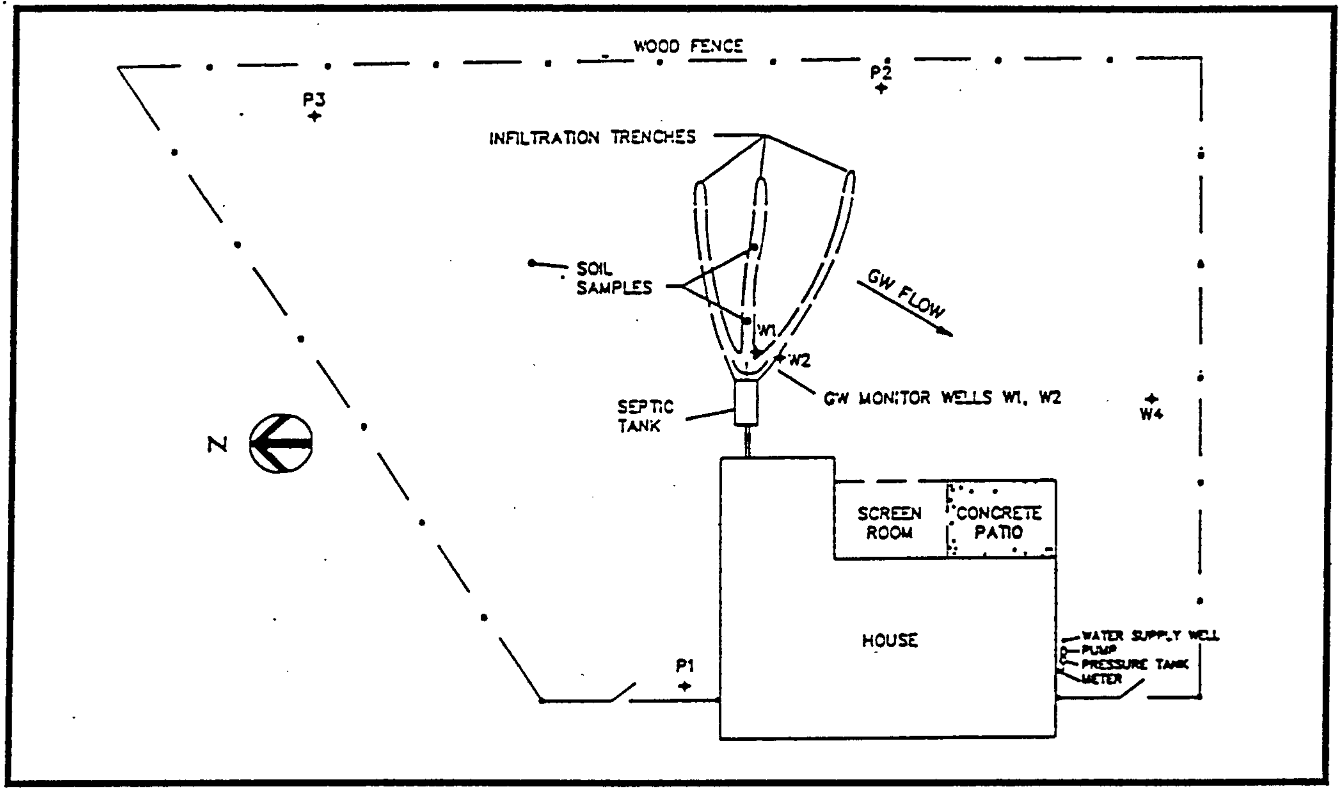

Figure 4.7.5: Plan View of OSDS Monitored at Home 2-2, Subdivision 2.

although some sampling events varied slightly. Each well was sampled on fourteen different occasions. This resulted in a total of 9,330 gallons of groundwater being analyzed from well W1 and 8,600 gallons from well W2.

Viral agent was detected at low levels in well W1 on two occasions. The viral agent in W1 was serotyped as Coxsackievirus A9 and was the same virus which had been detected in both feces and STE samples (see Table 4.7.7). This virus was discharged from the septic tank for a period of approximately sixty (60) days. Additionally, the same serotype was detected in groundwater immediately below the infiltration area on two occasions, three weeks apart. No virus was detected in groundwater samples collected from well W2, approximately 2 ft from an infiltration trench and 10 ft downgradient from W1. Analysis of samples from both wells identified impacts from other constituents of septic tank effluent (N, P, Cl, Fecal coliform).

TABLE 4.7.7 COMPARISON OF SELECTED STOOL, STE, AND GROUNDWATER SAMPLES, HOME 2-2.

Sample Date	Stool Results (Resident)	STE Results Serotype	(MPN•IU/L)	Groundwater Results Serotype	W1 (MPN•IU/L)[a]	W2
2/26/90	----- [b]	Neg. [c]	0	Neg.	0	Neg.
3/12/90	Neg. (RF)	Cox A9	1.72	Cox A9	0.003	Neg.
3/15/90	Cox A9 (JF)	-----	-----	-----	-----	-----
4/ 2/90	-----	Cox A9	>7.28	Cox A9	0.0005	Neg.
4/15/90	Neg. (JF)	-----	-----	-----	-----	-----
5/ 3/90	Neg. (JF)	-----	-----	-----	-----	-----
5/ 9/90	-----	Cox A9	0.19	Neg.	0	Neg.

a Most probable number of infectious units per liter.
b No sample.
c Negative result, no virus detected.

Unlike the samples collected for analysis of conventional parameters which were obtained by bailers, viral sampling was conducted by pumping groundwater from the monitoring wells. Typically 10 hours of pumping at a rate of 1 gallon per minute was conducted for each sampling event. This was required to obtain the 600 gallon sample volumes for viral adsorption and subsequent analysis. This method significantly altered normal flow patterns near the wells and resulted in groundwater being "pulled" from a radial area around the monitoring well. The radial distance around each well from which groundwater was pulled during monitoring at Home 2-2 was at least 4 to 6 ft based on a simple calculation estimating effective pore volume of the surrounding fine sandy soil. At the natural seepage velocities encountered in the study (0.02 to 0.06 ft/day), this increased flow due to pumping of groundwater represented 60 to 200 days of "natural flow" past each well. Since both monitoring wells were sampled via pumping on an approximately monthly basis for 14 months which included a rainy season, a continuous sampling of groundwater was obtained which should have adequately represented impacted groundwater in the vicinity of the infiltration system.

These results corroborate those of the subdivision groundwater monitoring study. They suggest that groundwater transport of enteroviruses down gradient from OSDS is more retarded than transport of conventional parameters. These results were obtained at Home 2-2 where groundwater elevations were observed within 2 ft of the infiltrative surface on several occasions.

4.7.1.6: Impact From OSDS on Water Quality in the Turkey Creek Sub-Basin of the Indian River Lagoon

This phase of the OSDS research was conducted to determine the potential impact from OSDS on water quality in the Turkey Creek Sub-Basin of the Indian River Lagoon. Groundwater and surface water samples were collected in a residential subdivision in Palm Bay, Florida, in a site-specific study of individual OSDS. The subdivision is typical of many OSDS subdivisions in the area in that groundwater and surface water drainage from the site flows via canals to the Indian River Lagoon. This study was funded as part of the Surface Water Improvement and Management (SWIM) program of the St. Johns River Water Management District.

The primary objective of the Indian River Lagoon Study was to assess the impact of several existing OSDS on water quality, particularly nutrient and bacteriological concentrations, in adjacent canals. Secondary objectives of the study were to add to the database on migration of pollutants from individual OSDS and to evaluate pollutant attenuation in the subsurface environment below and downgradient from such systems. To accomplish these objectives, two different residential OSDS and an undeveloped control site were investigated over a two year period to determine impacts on a drainage canal which empties into the Indian River Lagoon.

Summary of Methods: Septic tank effluent samples were collected and analyzed to characterize the quality of wastewater discharged to the OSDS infiltration areas. Water meter readings were collected to estimate average wastewater flow. Twenty five (25) monitoring wells and 12 piezometers were clustered at the two homes and a control site in the study area. Groundwater and surface water samples were collected on 14 different sampling dates over a study period from February 1990 through March 1992. These samples were analyzed for key water quality parameters indicative of OSDS impacts. Seepage meters and canal piezometers were installed in the canal bottom to determine seepage rates for estimating nutrient loading to the canal. Canal surface and seepage water samples were collected and analyzed for the same parameters as the groundwater samples. Depth to the water table was measured during each sampling event. The measurements were used in conjunction with survey data to determine groundwater flow direction. Aquifer tests and a bromide tracer test were conducted to determine travel time through the unsaturated zone and groundwater seepage velocity.

Site Characteristics: The residences studied were typical of those in the Port Malabar subdivision and utilized separate black and greywater OSDS. Water use monitoring indicated that wastewater loading rates to the systems were below design loading rates per Chapter 10D-6, FAC. Soils of the study area were typical of the South Florida Flatwoods land resource area and consisted of Myakka sand at the Jones site, Oldsmar sand at the Groseclose site, and Eau Gallie sand at the control site. A sandy clay loam layer was encountered at depths of 5 to 7 ft at the Groseclose site.

Summary of Results: Based on the data collected in the study, the following results were obtained on groundwater quality and the potential impact to surface water quality from OSDS in the area.

Groundwater flow direction at both residences and the control site was in the general direction of the drainage canal to the north. Groundwater elevation monitoring indicated an unsaturated soil thickness below infiltrative surfaces which varied during the study from 3.3 to 5.2 ft at the Jones site to 1.2 to 2.9 ft at the Groseclose site. Bromide tracer testing at the Groseclose site indicated an average travel time of 5 days to move through the 1.75-foot unsaturated zone below the infiltrative surface and an average groundwater seepage velocity of 0.24 ft/day towards the canal.

Analysis of ground and surface water samples from wells located at different distances from the OSDS drainfields indicated that the concentration of nitrate and nitrite-nitrogen (NO_3&NO_2-N), TKN, TP, and conductivity were generally significantly higher in the vicinity of the OSDS when compared to the upgradient wells. However, contaminant concentrations in wells located 20 to 40 feet from the OSDS were at or below background concentrations.

Fecal coliform (FC) counts in samples from the groundwater monitoring wells were generally below 10 colonies/100 mL on two-thirds of the sampling dates. Fecal streptococcus (FS) levels were high in samples from all wells, generally ranging from 100 to 2,000 colonies/100 mL (geometric means). Bacterial data did not statistically ($p \leq 0.05$) indicate significant reductions in number with increasing distance from the OSDS. The high levels of fecal streptococcus encountered in the groundwater at the Groseclose site were thought to be attributable to the utilization of canal water for lawn irrigation and the nearby presence of ducks, geese, and chickens. Analysis of bacterial data suggests a wildlife rather than human source of contamination. The FC/FS ratios of collected groundwater samples were very low. The average FC/FS ratio was 0.04 which is indicative of a non-human source of fecal contamination.

Bacterial counts were high and variable in the surface water obtained from the canal. Fecal coliform and fecal streptococcus levels peaked at a canal station located near the Groseclose site. As previously mentioned, the peak levels of bacteria appeared to be related to the presence of numerous ducks, geese, and chickens in the vicinity of this sampling station. This was supported by FC/FS ratios at other canal monitoring stations which averaged 0.17. The FC/FS ratios also suggest that stormwater run-off may be a source of bacterial loading to the canals. Based on sample analysis of canal water, OSDS impacts on the water quality of receiving canals were not evident. There were no statistically significant ($p \leq 0.05$) relationships between nutrient concentrations in the canal surface water and sampling locations relative to OSDS.

Considerable increases in concentrations of several parameters were measured in August and September of 1991, near the end of the study period. Nitrate-nitrogen concentrations increased in groundwater obtained from monitoring wells located within 25 ft of the blackwater OSDS. Phosphorous and TKN concentrations also increased in some of the wells. At the Jones site, peak nitrate-nitrogen concentrations exceeded 50 mg/L at several wells located within 20 ft of the blackwater drainfield. Total phosphorous and TKN concentrations were also elevated. Fecal coliform counts increased during the August and September 1991 sampling events. It was speculated that these increases were due to higher water table elevations or a shift in groundwater flow direction, but further monitoring would be necessary to determine the specific cause. Based on data collected after five rainfall events, no conclusive cause and effect relationships on either ground or surface water quality could be determined.

Water quality data of seepage meter fluid may not be directly comparable to monitoring well or even canal piezometer data and, in turn, may not be useful for the determination of nutrient loading to the canal. Based on parameter concentrations encountered in seepage meter fluid, water quality was probably affected by conditions within the seepage meter itself, such as anaerobic conditions.

Data collected from bromide tracer tests at the Groseclose site indicated that conservative parameters such as nitrate and chloride should reach the canal from the OSDS in approximately 270 days, yet concentrations of these compounds were found to be at background levels within 20 to 40 ft of the OSDS infiltration areas. Although some dilution may be responsible for these results, calculation of "dilution factors" indicated that denitrification may have been contributing to substantial nitrogen removal and phosphorous was significantly attenuated by onsite soils. Additional monitoring of the bromide tracer should be conducted to estimate dilution more accurately.

The results of the study indicate that OSDS were impacting groundwater in their immediate vicinity, but they were not impacting canal water quality significantly at the time of the study. This may not continue indefinitely, however, and it is estimated that total phosphorous loading to the canal may eventually reach a maximum of 1 to 2 kg/home/year for homes bordering the canal. Although nitrogen was significantly reduced at the study sites (especially Groseclose), it was estimated that under less favorable conditions, total nitrogen loading from homes bordering the canal could be as high as 4 to 7 kg/home/year. Fecal bacterial impacts to the canal could not be assessed from the variability of the data collected. It is concluded that a better indicator of bacterial impacts than the fecal coliform testing is needed.

Based on the results obtained, it was recommended to HRS and the St. Johns River Water Management District that a preliminary nutrient budget for the Indian River Lagoon from all sources be completed utilizing the estimated loadings for OSDS inputs. If the OSDS nutrient loading is determined to be a significant part of the overall nutrient budget of the lagoon, it is recommended that additional study to refine OSDS nutrient loading estimates be conducted. If the preliminary estimates prove to be accurate, it is recommended that an investigation of nutrient reduction techniques for OSDS be initiated.

4.7.1.7: <u>Investigation of the Treatment Capability of Fine Sand for Septic Tank Effluent Renovation at an In-situ Lysimeter Facility</u>

The final phase of the Florida OSDS Research Project was the design, construction, and monitoring of an in-situ field lysimeter station to evaluate the STE treatment capability of fine sandy soils. Details of the design, construction, and operation of this facility can be found elsewhere (Ayres Associates, 1993b). Only a brief summary of the preliminary results of this research effort are presented here.

The lysimeter station was designed to evaluate the treatment capability of the soil over time under controlled conditions in the field. Controlled conditions are necessary because determination of OSDS treatment capabilities and impacts to groundwater is difficult from field studies of existing systems due to variable flows, wastewater and groundwater characteristics. Most previous controlled experimental work has been done with laboratory soil columns, but field conditions cannot typically be duplicated in the laboratory limiting the transferability of results. Laboratory simulation of field conditions is particularly difficult for the climate of Florida where heavy thunderstorms may be affecting treatment performance within the very porous, sandy soils of the state. For these reasons, a field study on undisturbed natural soils with control over selected variables important to evaluation of treatment capability was desired. The field lysimeter research station was designed and constructed to accomplish this.

Experimental Design: The experimental design consisted of evaluating the effect of unsaturated soil thickness and wastewater hydraulic loading rate on the capability of fine sandy soil to treat septic tank effluent. Table 4.7.8 summarizes the selected variables which were controlled and studied.

The main response variable measured was the quality of soil water below the infiltrative surface. Parameters measured included total organic carbon (TOC), total kjeldahl nitrogen (TKN), nitrate nitrogen (NO_3-N), total phosphorus (TP), chloride (Cl), surfactants, as methylene blue active substances (MBAS), fecal coliform (FC), fecal streptococcus (FS) bacteria, and several conventional water quality indicators. In addition, soil moisture content, oxidation-reduction potential (ORP), and temperature were measured to evaluate soil characteristics. Climatic conditions, such as temperature and rainfall were also monitored to better evaluate treatment efficiency of the soil.

TABLE 4.7.8: SUMMARY OF LYSIMETER STATION STUDY VARIABLES.

Controlled Variables	Variable Values
Soil Type	fine sand (Quartzipsamment)
Unsaturated Thickness	2 ft and 4 ft
Hydraulic Loading Rate	0.75 gpd/ft^2 and 1.50 gpd/ft^2

Lysimeter Station Design and Construction: The research lysimeter station was constructed on the University of South Florida (USF) Campus in Tampa. Figures 4.7.6 and 4.7.7 show plan and sectional views, respectively, of the facility. Septic tank effluent is obtained from a campus ministry building which houses three to four students and is currently served by an existing OSDS. Soil at the research station site is Candler fine sand, an excessively drained, uniform fine sand commonly found on the uplands of central Florida. Groundwater at the site is greater than 20 ft below ground surface.

To evaluate the effect of unsaturated thickness on treatment, a subsurface sampling gallery was constructed to a depth of approximately 8 ft below grade (see Figure 4.7.7). To preserve undisturbed, native soil conditions next to the gallery for wastewater infiltration areas, 70-foot long sheet-pilings were vibrated into place isolating a 4-foot wide by 70-foot long soil area on

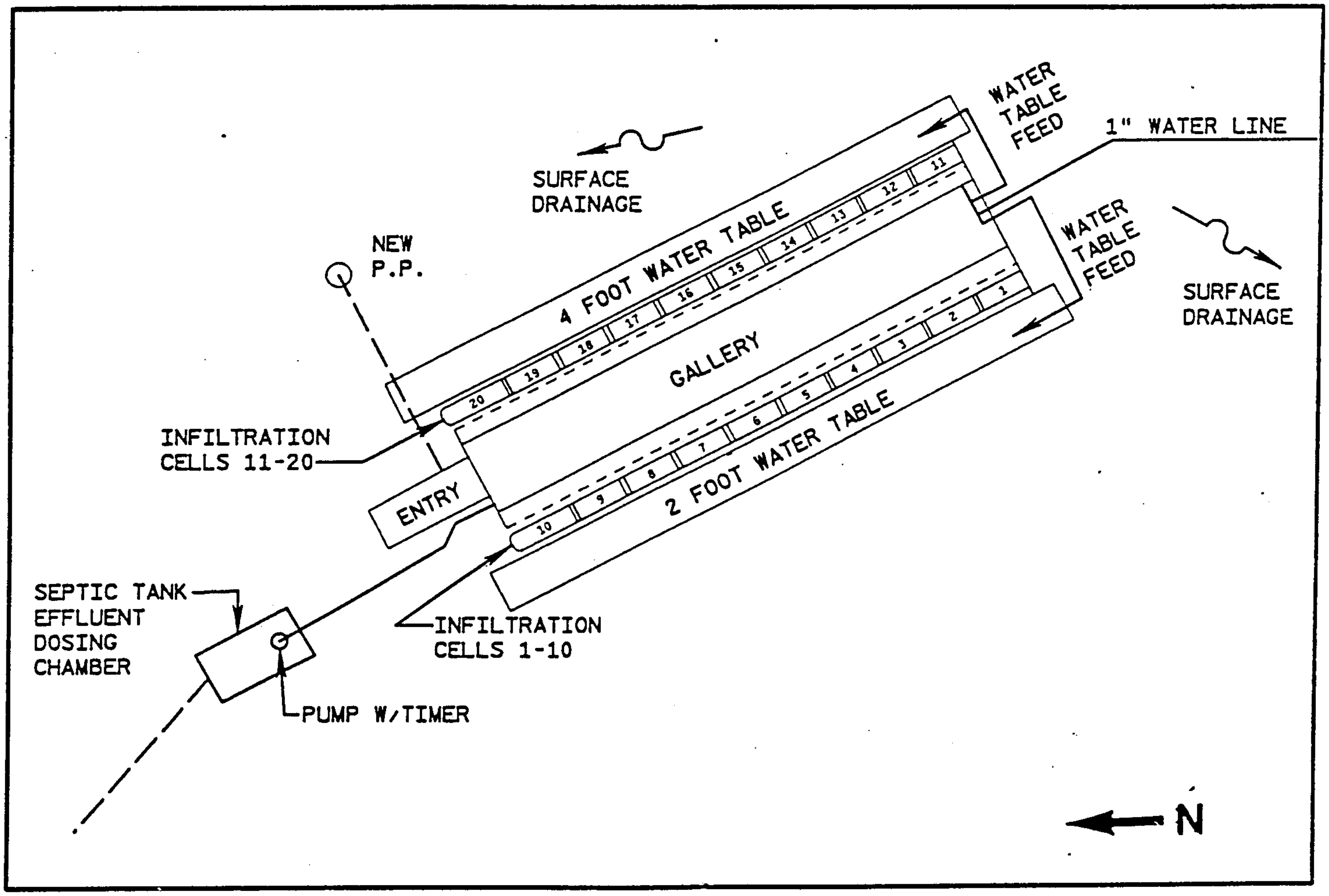

Figure 4.7.6: Plan View of the Lysimeter Station Facility

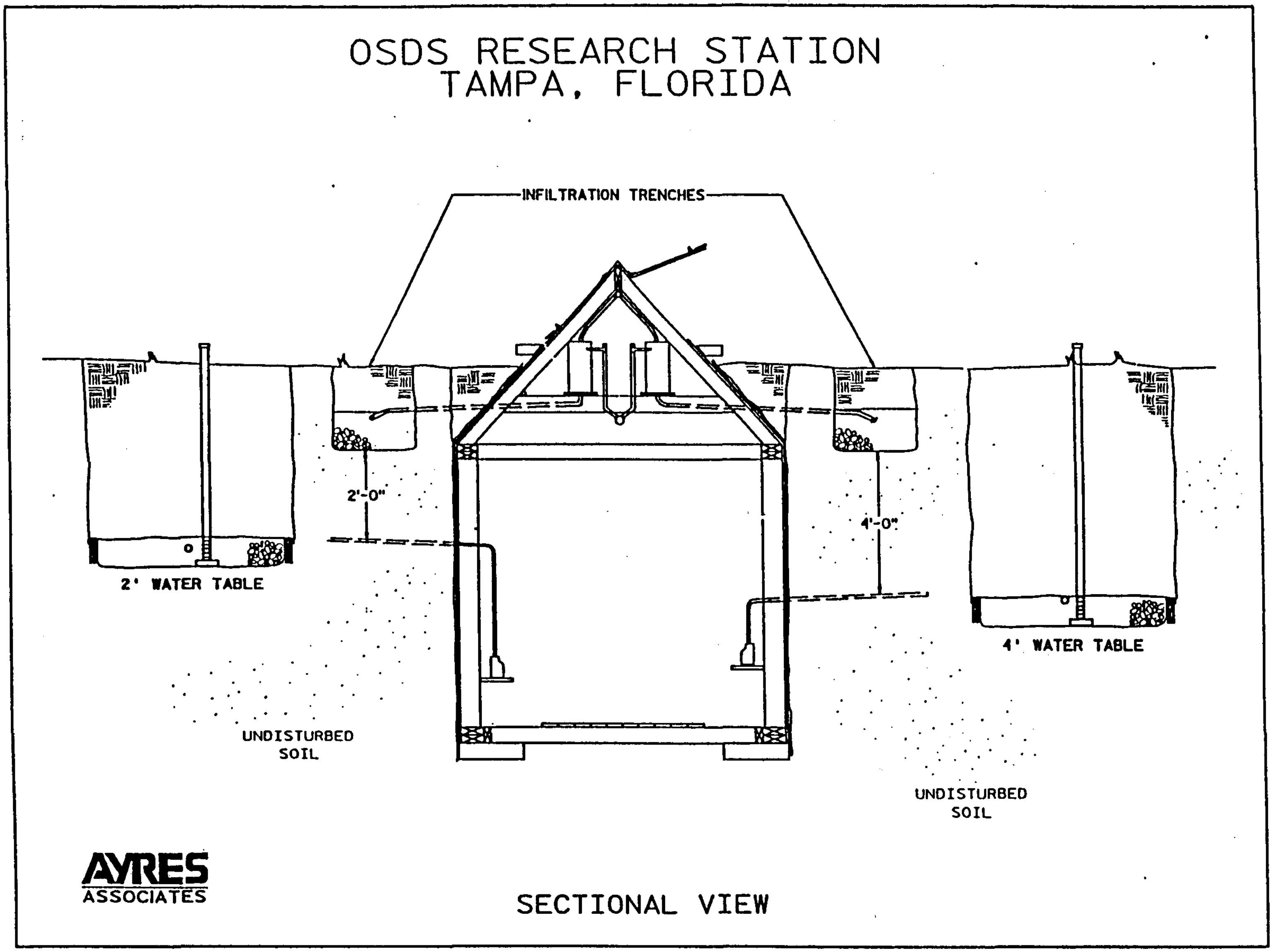

Figure 4.7.7: Sectional View of the Lysimeter Station Facility

each side of the proposed gallery structure. The gallery was constructed between the sheet-piled areas. Outside of each sheet-piled area, artificial water tables were constructed to simulate the soil moisture conditions for the two unsaturated thicknesses selected (See Figure 4.7.7). The artificial water tables were constructed with the water surface at a depth of approximately 2 ft below the proposed infiltrative surface elevation on one side of the gallery and 4 ft below on the other side. These were constructed by excavating to the desired depth and lining the excavation with 10 mil plastic. A 4-inch layer of pea gravel was placed on the liner, and 1-inch PVC water distribution line placed on the gravel. These areas were backfilled with native soil to original grade. Saturation is maintained for 6 to 12 inches above the liner to simulate the effects of a water table.

Once construction of the gallery and water tables was completed, the sheet-pilings were removed, leaving undisturbed soil next to the gallery walls. Two-foot wide, gravel-filled, wastewater infiltration trenches were constructed on either side of the gallery, approximately 18 inches from the gallery walls. Each trench was divided into eight individual wastewater infiltration cells, separated by a divider wall which prevents wastewater short circuiting from one cell to another, and two end cells. Divider walls were placed to a depth of 2 inches below the infiltrative surface and extended 2 inches above the top of the 12-inch thick gravel layer. Each cell has a separate wastewater distribution system to which loading can be controlled. Thus, 16 separate infiltration systems can be operated simultaneously, eight subject to a 4-foot water table and eight subject to a 2-foot water table. In addition, two control cells were established on each side of the gallery at the ends of each row of infiltration cells. These were established to examine the quality of water naturally percolating through the soil, and consist only of a column of natural soil and do not contain gravel or distribution piping.

Lysimeter Monitoring Equipment: To obtain soil water samples from the two unsaturated depths, stainless steel sampling pans were fabricated and pushed from within the gallery with a hydraulic jack into the unsaturated soil below the infiltration trenches. The pans were inserted below each individual infiltration cell, at 2 or 4 ft below the infiltrative surfaces. The pans have a slight descending gradient toward the gallery for collection of samples. The intent of the pans is to intercept septic tank effluent percolating through the unsaturated soil. Soil water should saturate at the pan surface and flow through a sampling tube into the gallery for collection in sample bottles. The pans extend to within approximately 12 inches of the artificial water tables. Horizontal and vertical separation prevents the water tables from flowing onto the pans, causing sample dilution. The purpose of the water tables is to maintain adequate moisture content in the soil to minimize lateral movement of percolating soil water. Without the water tables, dry soil next to the lysimeter pans would likely pull much of the percolating STE away from the pans preventing sample collection. The resulting effect of the water table-lysimeter pan complex is to simulate the physical soil conditions above 2 and 4-foot water tables.

In addition to the stainless pans, porous ceramic suction lysimeters were placed in the soil at selected locations. The suction lysimeters were placed in the soil immediately over the pans, and at one-foot intervals in several locations. This allows sample collection from a desired depth in the absence of complete saturation of the soil above the pans. Soil moisture tensiometers, soil thermometers, and oxidation-reduction probes were also placed in the unsaturated zone at selected locations.

Lysimeter Operation: Wastewater was dosed to the infiltration cells by pumping into dose pots which can be calibrated based on a desired cell loading rate. Wastewater was distributed to the cells six times per day on a schedule designed to approximate the loading from a single family home. Doses were applied at 6:00, 7:00, and 8:00 a.m., and noon, 6:00 and 7:00 p.m. each day. On each side of the gallery, three cells received 0.75 gpd/ft^2 of STE and three cells received 1.5 gpd/ft^2 of STE. The remaining two cells received tap water, one at each loading rate, on the same loading schedule as the cells that received STE. Tap water cells were used as additional controls to examine effects which were not related to wastewater loading. The experimental conditions consisted of the following: triplicate STE cells at 0.75 gpd/ft^2 and 1.50 gpd/ft^2 loading with a 2-foot unsaturated zone, triplicate STE cells at 0.75 gpd/ft^2 and 1.50 gpd/ft^2 loading with a 4-foot unsaturated zone, one tap water cell at each loading rate and unsaturated thickness, and two end cells of native soil at each unsaturated thickness. These conditions are summarized in Table 4.7.9.

Operation of the lysimeter facility was started in June 1992, with tap water to correct any preliminary operational problems. In August 1992, wastewater dosing was initiated with STE from the campus ministry. Monitoring of STE quality was begun after several days and soil water sample collection after two weeks.

It was realized during the initial operation with tap water that soil water samples may be difficult to obtain by way of the lysimeter pans. Apparently soil saturation at the pan surface has not occurred under the loading rates and soil conditions of the experiment. The facility design was based on the theory that a steady state moisture condition would develop below the infiltrative surfaces with time resulting in saturation on the pan surfaces. This did not occur. Examination of

TABLE 4.7.9: SUMMARY OF LYSIMETER STATION EXPERIMENTAL CONDITIONS

Number of Infiltration Cells	Unsaturated Thickness (ft)	Loading Rate (gpd/ft^2)	Loading Type
3	2	0.75	STE
3	2	1.50	STE
1	2	0.75	Water
1	2	1.50	Water
2	2	0.00	None
3	4	0.75	STE
3	4	1.50	STE
1	4	0.75	Water
1	4	1.50	Water
2	4	0.00	None

soil moisture characteristics at the site explain the reason for this. Figure 4.7.8 presents soil moisture retention curves for cores collected from the site. Four curves are shown for sample depths of 1 to 4 ft below the infiltrative surface. These data show the uniformity of the soils with depth. They also show the abrut drop in volumetric water content that occurs as soil moisture tension rises. This is typical of a sand with very uniform grain and pore sizes, as all pores tend to drain at the same tension. Soil moisture tension immediately above the pans under operating conditions of the 2-foot unsaturated zone was observed to be 35 to 40 millibars (mb), and 40 to 45 mb below the 4-foot unsaturated zone. These soil moisture tensions, although near saturation, are enough to retain water in the soil pores. Lateral movement of the soil water, away from the gallery was the likely cause for failure to achieve zero potential at the soil-pan surface interface. Distribution of greater volumes of water to the artificial water tables may sufficiently modify these conditions and allow saturated flow on the pan surface.

Because the pans did not yield samples by gravity, the ceramic suction lysimeters were used for soil water sample collection during the first five sampling events. By applying a vacuum to the porous ceramic cups, soil suction was overcome and soil water pulled into the sampler. The ceramic suction lysimeters were of limited use, however, as they cannot be used to sample for

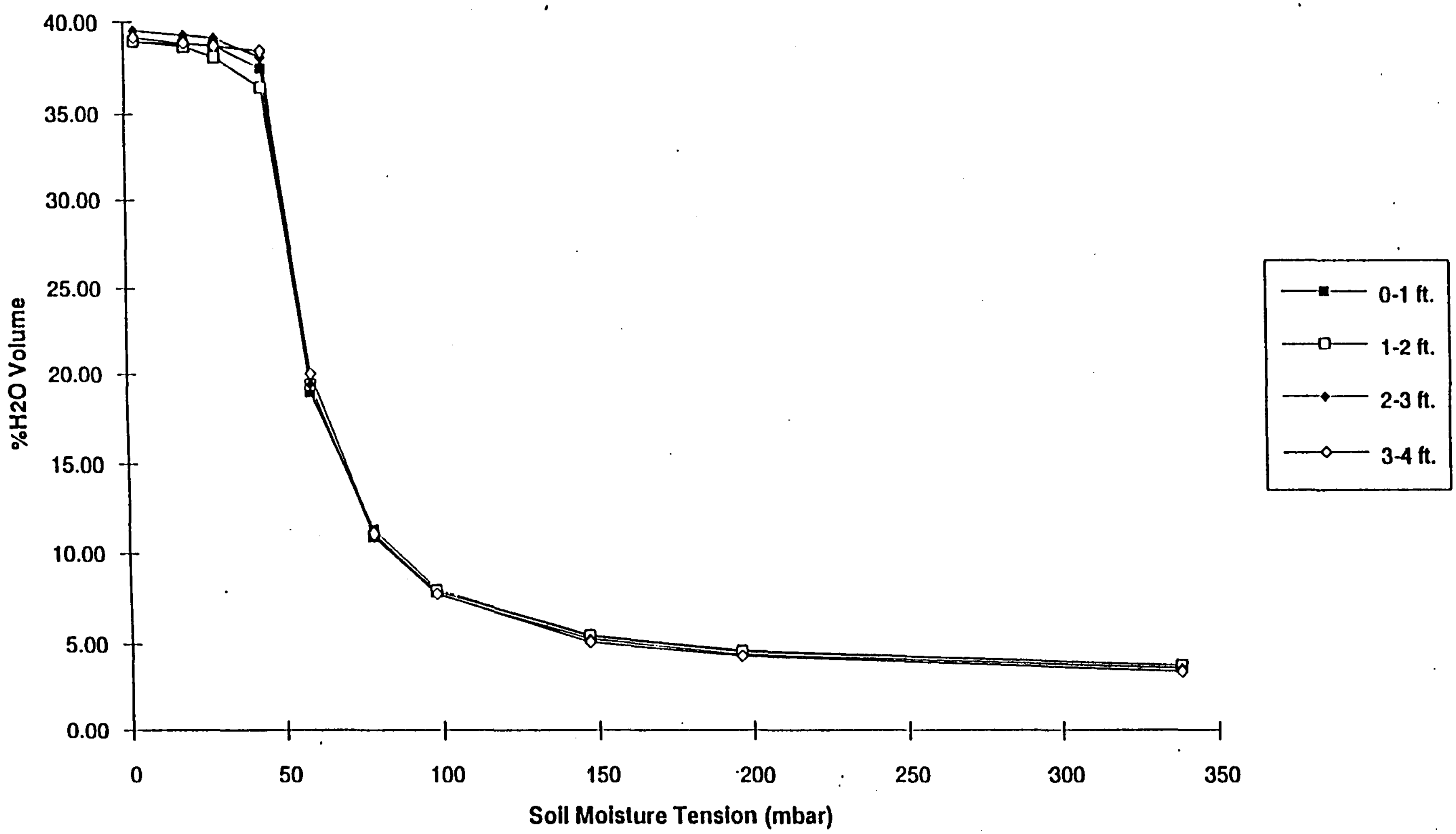

Figure 4.7.8: Soil Moisture Retention Curves

bacteria because of attenuation by the porous ceramic. Therefore, a very low vacuum (50 mb) was applied to the sampling tubes of the pan lysimeters in order to obtain a sample. The attempt was successful in releasing soil water for sample collection. Initial samples contained small quantities of soil particles and were turbid, but samples progressively cleared with each sample event. Subsequent samples were collected from the pans by this method.

Preliminary Lysimeter Results: Table 4.7.10 shows preliminary results for several key parameters of this research effort based on sample analyses through the first six months of facility operation. The parameters shown on this table were unaffected by the ceramic suction samplers so the data were combined with that from the stainless steel pan samplers. Differences among treatment replicates were not observed so values from all similar cells were utilized for this analysis. Septic tank effluent sample analysis results are also presented in Table 4.7.10.

Results of monitoring soil water below the tap water control cells indicated that contributions from soil of the parameters measured would not occur, except for TOC. Tap water increased in TOC concentrations by approximately 2 to 4 mg/L as it percolated through the native soil at the site, regardless of loading rate or unsaturated zone thickness. All other parameters appeared to remain relatively unchanged.

The average soil water quality results are interesting when compared to the average quality of the applied STE. These preliminary results indicate significant attenuation of TOC, MBAS, TP, and TKN in both the 2 and 4-foot cells. In fact, little difference in attenuation of these parameters was observed between results of the two unsaturated zones. TOC reductions were on the order of 80 percent, while MBAS was reduced by over 99 percent. TKN reductions were in excess of 97 percent indicating almost complete nitrification of STE nitrogen. Total phosphorus was reduced by over 96 percent for all conditions except the high loading of the 2-foot cells. Since capacity of the soil to retain phosphorus is limited, it will eventually "break through" and enter the water table. It appears that this may be about to occur in the high loaded, 2-foot cells, as several of the last samples collected showed a trend of increasing TP. Continued sampling of the cells should occur so that time to phosphorus breakthrough can be measured and phosphorus capacity of the soil estimated.

TABLE 4.7.10: PRELIMINARY DATA SUMMARY OF LYSIMETER STATION STE AND SOIL WATER QUALITY

			2 feet below infiltrative surface		4 feet below infiltrative surface	
Parameter	Statistics	STE Value	0.75 gpd/ft2	1.5 gpd/ft2	0.75 gpd/ft2	1.5 gpd/ft2
	n[a]	9	28	30	26	25
	x[b]	499.33	470.14	480.33	375.38	500.88
TDS	s[c]	72.30	130.82	128.94	108.31	99.08
	min	400.00	184.00	192.00	200.00	334.00
	max	610.00	620.00	654.00	592.00	656.00
	n	9	28	28	25	25
	x	75.67	42.54	43.96	29.87	40.84
Chloride	s	20.16	15.05	14.20	11.53	9.31
	min	44.00	9.20	14.00	8.80	22.00
	max	110.00	65.00	65.00	49.00	58.00
	n	9	28	30	26	25
	x	49.22	8.21	8.77	8.70	9.26
TOC	s	11.83	3.51	2.92	4.61	3.30
	min	33.00	3.70	3.60	4.40	5.60
	max	68.00	17.00	18.00	25.00	18.00
	n	9	27	29	25	25
	x	9.49	0.05	0.05	0.05	0.05
MBAS	s	4.24	0.01	0.01	0.01	0.00
	min	4.60	0.05	0.05	0.05	0.05
	max	17.00	0.08	0.09	0.08	0.06
	n	9	29	32	26	28
	x	9.56	0.38	1.23	0.20	0.37
TP	s	3.13	0.76	1.78	0.36	0.46
	min	7.20	0.01	0.02	0.02	0.02
	max	17.00	3.80	8.80	1.80	2.00
	n	9	29	31	26	25
	x	47.22	0.78	0.88	0.78	1.19
TKN	s	5.19	0.23	0.29	0.35	0.61
	min	40.00	0.40	0.35	0.25	0.57
	max	53.00	1.40	1.60	2.10	3.50
	n	9	29	31	25	25
	x	0.04	24.41	22.90	14.71	23.52
NO3	s	0.05	10.12	10.41	7.72	6.90
	min	0.01	1.70	0.01	2.00	11.00
	max	0.16	39.00	38.00	29.00	35.00

[a] Number of samples.
[b] Sample mean.
[c] Standard deviation of the mean.

The reductions in parameter concentrations apparently are due in part to dilution of the effluent with natural soil moisture. Assuming chloride is a conservative parameter, it appears that 40 to 45 percent of the noted reductions in parameter concentrations may be due to dilution. It should be noted that the reduction in chloride in the low loaded, 4-foot cells is unusually high and was not included in determination of this estimate. The reason for this disparity is unclear.

Nitrification of STE nitrogen resulted in NO_3 nitrogen concentrations in excess of 20 mg/L at both the 2-foot and 4-foot depths. Although this suggests a reduction in total nitrogen from STE concentrations, it appears that most of this reduction is due to dilution with natural soil moisture as previously indicated. Therefore, substantial concentrations of NO_3 nitrogen are expected to enter groundwater from SWIS in these fine sandy soils.

No detection of fecal coliform or fecal streptococcus has occurred after six sampling events. It appears that these fecal indicator bacteria are effectively attenuated in the unsaturated fine sandy soil below the infiltration trenches.

The preliminary results described here for the lysimeter facility show substantial attenuation of key parameters related to STE treatability in soils. Effective removal of TOC, MBAS, TP, TKN, and fecal indicator bacteria were observed at both the 2-foot and 4-foot depths. Little difference was noted due to the hydraulic loading. Nitrate nitrogen, as expected, was generated from nitrification of STE nitrogen and was transported to both the 2-foot and 4-foot depths at relatively high concentrations.

While the results above show excellent treatment of STE by fine sandy soil, these results need to be evaluated in light of operational status of the lysimeter station. As discussed previously, saturated conditions did not form at the pan samplers. This indicates that soil below the infiltration trenches may be somewhat drier than would be the case if an actual water table were present. Measurements of soil moisture near the pan surface indicated very wet, but not quite saturated conditions. Based on the soil moisture retention data collected, it appears that present lysimeter conditions more closely simulated 2-foot and 4-foot distances to a capillary fringe, rather than a true water table. Thus, the observed reductions of some parameters may be somewhat higher than would occur with the existence of a true water table condition. It is felt that such a condition can be simulated at the lysimeter station with minor modifications.

4.7.2: Summary of Florida Research Project Results

The OSDS Research Project has resulted in advancements in knowledge related to the use, function, and performance of onsite wastewater systems in Florida. Major findings of this research include:

- It appears that most of the future OSDS in Florida will be installed on the fringes of urban areas.

- Many of these areas have sandy soils with relatively high water tables, where the use of conventional OSDS will be prohibited.

- Groundwater modeling of OSDS impacts from high density subdivisions indicated a potential for nitrate contamination to exceed the groundwater standard down gradient of the subdivisions, but only after a considerable period of continued OSDS use.

- Groundwater monitoring of subdivisions using OSDS in four differing areas of the state did not confirm the modeling results. Suspected impacts from OSDS were measured in groundwater samples in every subdivision, but the impacts appeared to be from individual systems and not the subdivision as a whole. Widespread groundwater contamination down gradient from OSDS subdivisions was not found to occur, but the subdivisions studied were relatively young and not completely developed.

- Volatile organic compounds (VOCs) were routinely measured at concentrations typically less than 100 ppb in septic tank effluent samples from household septic tanks. However, VOCs were not found in soil or groundwater 2 ft below the infiltrative surface.

- Soil core sampling below wastewater infiltration systems of conventional OSDS installed in sandy soils indicated that several STE parameters, including nitrogen and phosphorus, could enter groundwater if present at 2 ft or less below the infiltrative surface. Biodegradable organics, surfactants, and fecal indicator bacteria were effectively attenuated in most systems by 2 ft of unsaturated soil. It appears that attenuation of almost every parameter except nitrogen could have been increased had more even distribution of effluent to the infiltration system been achieved.

- Viral monitoring of individual OSDS documented the occurrence of human enterovirus in infected homeowner feces, in septic tank effluent, in sandy soils below an infiltration system, and in shallow groundwater directly below the system. Virus was not found in the contaminant plume immediately down gradient of the infiltration system, however, indicating that groundwater transport of enterovirus may be more retarded than conventional parameters.

- An investigation of several OSDS along drainage canals in the Indian River Lagoon Basin indicated that OSDS were not significantly impacting surface water quality in the canals after eight years of operation, even though significant impacts from both nitrogen and phosphorus were measured in groundwater in the immediate vicinity (within 20 ft) of the wastewater infiltration systems.

- Preliminary results of the USF lysimeter facility indicate that 2 ft of unsaturated, fine sandy soil provides substantial attenuation of total organic carbon, surfactants (MBAS), fecal indicator bacteria, and total kjeldahl nitrogen. Total phosphorus has been effectively removed during the first six months of operation. Nitrate-nitrogen moved to depths of 2 and 4 ft in substantial concentrations and would likely impact shallow groundwater as expected.

- Prior to the studies performed as part of this project, very little onsite wastewater treatment research had been conducted in Florida. This research project represents a beginning of what should be an on-going onsite wastewater treatment system research program in Florida. Continuing the research will help to provide answers to many remaining questions regarding the performance and impacts of onsite system practices. Future research should focus on long-term, detailed studies of individual systems or of controlled experiments in the field and laboratory.

SECTION 5.0
PROGRAM STANDARDS

5.1: PERFORMANCE VERSUS PRESCRIPTIVE STANDARDS

Codes regulating onsite wastewater treatment systems are typically prescriptive, defining in detail the system design for specific site conditions and anticipated system use. Approvals for system use are limited to regulatory assessment based on strict compliance with the code rather than actual system performance. Implicit in prescriptive standards is a performance standard based solely on public health protection. It assumes that by keeping the wastewater below the ground surface and distant from water supply wells and surface water public health will be protected. This standard was developed for scattered rural homes and did not consider the consequences of onsite system use in high density subdivisions. Today, this standard is no longer adequate since it does not address potential environmental impacts. If onsite system practices are to improve, systems should be judged based on their performance relative to a rational standard rather than conformance to a design specification for which the performance standard on which it is based is not articulated and may not be understood.

Performance standards define the acceptable environmental impacts of onsite wastewater treatment systems by specifying measurable performance requirements. They do not require that site characteristics or treatment methods be specified. For example, a site with very rapidly permeable soils and a shallow water table would not be acceptable for development under a prescriptive code because of concerns for contamination of the groundwater by pathogenic organisms. However, under a performance based code, the regulating agency may accept an onsite treatment system which can demonstrate reliable and consistent pathogen removal to measurable performance levels specified by the agency before release to the groundwater. Thus, potential building sites are not deemed unsuitable if a treatment system can be designed and operated reliably and consistently to meet performance standards established for that site.

Regulatory programs for onsite treatment systems based on performance standards place greater responsibilities on the regulating agency, site evaluator and design professional, construction contractor, system operator, and system owner. The regulating agency must establish the performance standards and develop the competency to review and approve system designs which may be submitted to meet the established standards. The service provider (site evaluator, designer, contractor, or operator) must develop the knowledge and skills to design, build, and or operate treatment facilities capable of meeting the performance standards. The owner must accept the responsibility for meeting the performance standards and the costs associated with proper management.

The benefits of this approach to onsite wastewater treatment include:

- Greater protection of public health and the environment because each onsite system would be designed to meet specific public health and/or environmental goals,

- Preservation of groundwater resources because groundwater recharge would be increased by retaining wastewater in the basin rather than discharging it downstream through a regional treatment plant,

- Responsible and rational land use planning because it would be possible to provide onsite wastewater treatment facilities on sites where new developments are best suited rather than only where sites are suitable for conventional septic tank systems, and

- Availability of affordable housing because lower cost building sites and treatment facilities could be provided.

5.2: REQUIREMENTS FOR EFFECTIVE STANDARDS

A basic criterion for any standard is that it be measurable. The most simple performance standards are to require onsite treatment systems to perform without wastewater backups or surface breakouts and to achieve a specific water quality objective in the groundwater at some boundary such as the property line. The water quality objective should be based on a risk versus benefit analysis of achieving a particular standard under the given site conditions. Although such a standard is measurable, providing the necessary monitoring can be costly and difficult. Wastewater backups and surface breakouts are easily observed, but monitoring groundwater is expensive and often ineffective because locating the plume is difficult and travel times from the system to the point of monitoring can be long. As an alternative, a performance standard can be set before the pretreated wastewater is released to the environment. However, for many site conditions, it is difficult to predict what level of treatment is required before the wastewater is released to the infiltration system since the soil will provide significant additional treatment as the wastewater percolates to groundwater. A standard for the pretreated effluent could be overly conservative, increasing the cost of the onsite treatment system unnecessarily. For routine sites, therefore, providing an optional prescriptive alternative such as is currently used may be more practical. This would give the designer an optional standard on which to base the design.

If prescriptive designs are allowed, they must be capable of meeting the same performance standards as an engineered system would be required to meet. If used on an acceptable site, it would be assumed that the system would meet the performance standard without having to monitor water quality as long as the system is operated within its permit limits. In other words, the prescriptive design for a prescribed site should have a "defined performance" which is acceptable to the regulating agency. This performance could be that of a conventional septic tank system. However, the elimination of water quality monitoring would not preclude other monitoring such as hydraulic performance.

5.3: STANDARDS PROMULGATED BY CHAPTER 10D-6

Chapter 10D-6 is principally a prescriptive code similar to those used in most other onsite wastewater treatment regulatory programs. It establishes minimum requirements to be used by County Public Health Units throughout the state. More strict requirements can be set by county ordinance.

The chapter is divided into two parts. Part I applies to all areas of the state except where specific provisions of Part II exempt compliance with Part I requirements. Part II applies to those counties where more than 60% of the surface and subsurface soils consist of Key Largo Limestone or Miami Limestone (10D-6.041(1)). Part II was promulgated because of concerns for excessive fertilization of near shore marine waters from onsite wastewater treatment systems.

Part I is prescriptive for routine sites but provides nonspecific performance criteria for design review and approval of alternative systems on sites that are not suitable for construction of the prescribed septic tank system design. Approval of alternative system designs may be granted "...where evidence exists that use of such systems *will not create sanitary nuisance conditions, health hazards or pollute receiving waters*." Also,

"...procedures for routine maintenance, operational surveillance, and environmental monitoring *to assure the system continues to function properly...*" may be required (10D-6.049(5)).

Non-specific performance criteria are also stated for defining when failures of onsite systems have occurred and must be repaired. Failures are defined as "...a condition existing within an onsite sewage treatment system which *prohibits the system from functioning in a sanitary manner* and which may result in the *discharge of untreated or partially treated wastewater* onto ground surface, into surface water, into ground water, or which may result in the *failure of building plumbing to discharge properly...*" (10D-6.042(23)).

These standards are not measurable and do not provide effective criteria for initiating an enforcement action. "Nuisance conditions", "health hazards", "pollute", "function properly", "sanitary manner", and "partially treated" are not quantitatively defined and, therefore, their meaning can be argued. This makes the code difficult to enforce when performance problems occur.

Part II of Chapter 10D-6 provides special requirements for unique site conditions in the Florida Keys portion of Monroe County and those areas of Dade County where the Key Largo Limestone or Miami Oolite exist within 18 inches of the surface. This part of the chapter is strictly prescriptive. No performance standards are provided so that it is not at all clear what the treatment objective is for the onsite systems. Apparently, this portion of the code was promulgated because of concerns for potential impacts of wastewater constituents on the environment. However, quantitative performance criteria are not presented so that effective alternative designs cannot be developed nor approved, and treatment failures on operating systems cannot be identified. Therefore, except for obvious hydraulic failures, the regulating agency has no enforcement authority after the system is constructed.

SECTION 6.0
TECHNICAL GUIDELINES

6.1: SITE EVALUATION

6.1.1: Minimum Program Requirements

The most critical element in onsite wastewater treatment system design is the evaluation of the site on which the system is to be constructed. Most onsite systems rely on the site to provide final treatment and disposal of wastewater. In this context, the site becomes a biological, physical, and chemical treatment facility for wastewater as well as a porous medium through which to dispose treated effluent. Therefore, the site evaluator should regard the site as a wastewater treatment facility to be evaluated for its capability to renovate and hydraulically accept the expected wastewater. The site must be evaluated with regard to its capability for providing satisfactory environmental conditions to support active biomass, provide chemi-adsorption sites and physical filtration, and maintain adequate hydraulic conductivity.

The site evaluation needs to provide sufficient information to determine if the site can support an onsite wastewater treatment system, what system design concept to use, and what design parameters to follow. Site information collected must:

- Be collected using credible and accepted procedures, and
- Be presented clearly and precisely in a consistent manner acceptable for review.

The necessary components of a thorough site evaluation include assessments of:

- Permeability of the soil to water and air (O_2 and CO_2),
- Depth of unsaturated soil above seasonally saturated zones, high water table, and/or capillary fringe,
- Anion and cation exchange capacities, and
- Horizontal setback distances from surface features.

6.1.1.1: Site Evaluation Procedures

Site evaluations can be performed in a variety of ways. However, the information collected and the manner in which it is collected by the site evaluator should be in a form that can be easily reviewed and interpreted by the reviewing agency. Standard procedures should be established for use in all site evaluations which are consistent with acceptable practices. A common nomenclature for providing site descriptions should be required.

Soil profile descriptions usually are the most critical component of a site evaluation. Observation and assessment of soil properties can be done with adequate precision in the field. They can be best observed in undisturbed profiles exposed on faces of backhoe excavated pits dug to a depth sufficient for evaluation 2 to 4 ft below the proposed infiltrative surface elevation. A sufficient number of pits should be dug to adequately describe the variability of soil on the proposed site. Hand augers may be used to confirm soil conditions on the site, but they are not acceptable for performing detailed descriptions because the auger disturbs the soil making it difficult to observe structure and identify other soil features. An accurate plot plan should be drawn to scale showing the location of all soil borings. The plot plan also should show slope direction, grade or surface contours at 2-foot intervals, horizontal setbacks, and site evaluator recommendations. It should include a north arrow and identification and elevation of a permanent bench mark. Soil boring descriptions should be presented in table format. USDA Soil Conservation Service nomenclature should be used to provide a common and known descriptive standard for soil properties which is consistent with published soil surveys.

6.1.1.2: Site Evaluation Components

Soil Permeability: The permeability of the soil is not only important for movement of water, but also for transmission of oxygen to meet the oxygen demand created by microorganisms degrading wastewater constituents. Permeability is determined by the size, shape and continuity of the soil pores, and their availability to fluid flow. Several soil factors directly affect permeability. Soil texture is a measure of the relative proportion of sand, silt, and clay in the soil. Texture and structure both influence pore size and continuity. Soil structure refers to the organization of the soil particles into aggregates. The aggregates are separated by surfaces of weakness that provide a network of voids in the soil. Soil bulk density is the ratio of the mass of the soil to its bulk volume. Higher bulk densities are more dense with less pore volume and lower permeability. Clay mineralogy will impact permeability if expandable clays are present in significant amounts. Permeability can vary with depth due to horizon changes and stratification within the soil profile. Soil drainage as represented by matrix colors, mottles, and gleying provides information regarding the frequency and duration of saturated periods. Each of these individual soil properties can be important in assessing the permeability of soil and the availability of soil pores to transmit water and air. The relative importance of each will vary from site to site and observation of all may not be needed. However, texture, structure, horizonation, and soil color should be observed and recorded as part of all site evaluations.

Unsaturated Thickness: Depth to saturation from the infiltrative surface affects treatment as well as hydraulic performance. A minimum depth of 2 ft is needed between the infiltrative surface and a seasonally saturated zone to provide adequate aeration of the subsoil, to maximize wastewater contact with soil particle surfaces, and to maximize travel times for achievement of typical infiltration system treatment levels. Soil indicators that should be observed are soil saturation, soil color and mottling, and horizonation. Where test pits intersect the water table, the elevation of water in the pit after equilibration establishes the water table elevation. To establish the seasonally high water table elevation, soil color and mottling may be used. Perched water tables may occur during wet periods due to differences in permeabilities between soil horizons or stratified materials. Landscape position is also important in assessing soil drainage by providing information regarding flooding hazard and potential for subsurface drainage from higher landscape positions. Where seasonally high water table elevations cannot be established, monitoring of groundwater elevations in piezometers during wet periods may be necessary. Where the seasonally high water table is established to be near the 2-foot separation distance in granular soils, the thickness of the capillary fringe should be estimated from the effective size and uniformity of the soil particles.

Soil Sorptive Capacity: The movement of some wastewater constituents through soil is retarded by ion exchange capacity of the soil. Metals, orthophosphate, toxic organics, and virus can be retained at sorption sites in the soil until treatment, mineralization, or precipitation can occur to remove them from the waste flow. To assess the capacity of the soil to retain such constituents, it would be necessary to measure soil properties such as organic matter content and type, mineralogy, texture, oxidation-reduction potential, and pH. Special instruments or laboratory tests are needed to make these measurements. Typically, such factors are not included in site evaluations, but where onsite systems may be installed near sensitive ground and surface water resources, these factors should be considered to determine whether advanced pretreatment should be employed.

Horizontal Setback Distances: Spatial relationships of site features are important to determine the available unencumbered area for construction of the onsite treatment system. Appropriate setback distances must be maintained from wells, surface waters, property lines, etc., and enough suitable reserve area should exist for installation of a replacement system. This information is obtained through horizontal and vertical physical measurements, which are correlated to a permanent benchmark. Features on adjacent properties that might impact site suitability such as steep slopes, surficial drainage, and existing wells or onsite systems should be identified and their location shown.

6.1.2: Current Florida Program

Generally, the provisions within Chapter 10D-6 address most of the relevant requirements for an adequate site evaluation. However, emphasis is placed on evaluation of the site to accept the hydraulic load rather than on the capability of the site to treat wastewater.

6.1.2.1: Site Evaluation Procedures

Chapter 10D-6 establishes requirements for site evaluation. A minimum of two soil profile descriptions per site are required. They are to be dug to a minimum depth of 6 ft or to refusal (10D-6.044(3)(c)). USDA soil classification methodology must be followed in describing the profiles. Rather than rely solely on hand texturing of soil materials, laboratory sieve analyses can be required to determine a specific soil texture classification. However, the method by which profiles must be observed is not established.

Site data are entered on HRS-H Form 4015, "Onsite Sewage Disposal System Site Evaluation and System Specifications". This form provides spaces for most relevant information needed to assess site suitability for onsite system application.

6.1.2.2: Site Evaluation Components

Soil Permeability: Soil texture, soil color, horizonation, and soil series identification must be presented on HRS-H Form 4015 to apply for a construction permit. Description and evaluation of soil structure, relative bulk density, or clay mineralogy are not required. The soil texture limitation grouping and its correlated soil loading rate (10D-6.048 Table IV) mentions soil structure only as a factor which may impact soil permeability. A soil classified as "moderately limited soil materials" (10D-6.047(1)(b)) could be severely limited for conventional onsite system application if the soil structure were platy or massive or if the bulk density were high, even if all other site features were favorable. Where restrictive soil horizons exist within the soil profile, they may be replaced with slightly limited soil provided 54 inches of suitable soil is maintained below the infiltrative surface and all other site factors are favorable (10D-6.048 Table IV, Footnote 3).

Unsaturated Thickness: A minimum separation of 2 ft between the seasonally high water table and the infiltrative surface of an onsite system is required (10D-6.047(2)). Estimation of the seasonally high water table elevation must be based on review of available soil surveys, soil color, soil stratification, vegetation, and a site evaluation. Identification of the landscape position is not required. There is no requirement for wet season monitoring of water table elevations on sites where estimating the seasonal high water table is difficult. Where a 2-foot separation cannot be maintained, suitable fill material is permitted to be imported to provide the necessary separation from saturated zones when all other site factors are favorable (10D-6.047(2)).

Soil Sorptive Capacity: Sorptive capacity of the soil is not addressed by Chapter 10D-6.

Horizontal Setback Distances: Horizontal setback distances are defined for critical site features(10D-6.046). The setback distances required are similar to most onsite system codes.

6.2: DESIGN

6.2.1: Minimum Program Requirements

Site and soil conditions must be carefully integrated with the onsite wastewater treatment system design and operation to achieve the performance required. The design concept must be compatible with the characteristics of the wastewater to be treated and the site on which it is to be constructed. The system must be sized adequately to treat the wastewater characteristics and flows projected over the life of the system and provide means for effective management. An onsite treatment system regulatory program must ensure that both the design concept and system design are appropriate.

Whether prescriptive or performance based codes are used, the objectives of the regulatory agency in design review are the same. Prescriptive codes are more simple to implement, requiring only that the regulatory agency confirm that the system concept and design conform to the prescribed requirements. Unfortunately, prescriptive codes must be written for "typical" conditions, and do not address atypical site conditions which may be encountered. As a result, the most appropriate system concept and design may not be allowed because it does not comply with the code or, if the site is not judged "suitable" for the prescribed design, development may not be permitted. Performance codes can overcome these shortcomings by providing maximum flexibility. Flexibility allows the most appropriate system to be implemented for the site. However, because system designs that are submitted may not be familiar to the reviewer, the reviewer must be more knowledgeable to be able to technically judge a design. Therefore, reviewers must have more technical training than is necessary to review prescriptive designs. Judgment will become a significant factor in the design approvals under a performance based code. Thus, more frequent disputes between the designer and reviewer are likely. Therefore, for routine situations, prescriptive codes are more practical.

Prescriptive and performance codes can be effectively combined in a single program. The performance standard establishes the criteria that must be met by an onsite system under the given site conditions. Prescriptive designs which are accepted as providing adequate treatment to meet the performance standards under defined conditions can be provided by the code. The designer has to option to either "engineer" a system or use the prescriptive design as required by site and soil conditions.

6.2.1.1: <u>Prescriptive Design Codes</u>

Prescriptive codes cannot be written for all potential situations. They are generally based on a conventional design consisting of a septic tank and subsurface infiltration system. With such a concept, the performance criteria that are implicitly assumed are that wastewater backups or surface seepage (hydraulic criteria) do not occur and that wastewater organics and pathogenic agents (water quality criteria) do not reach the groundwater over the life of the building to be served (design life criterion). Other water quality criteria are not addressed by this design concept.

The criterion that the system function satisfactorily over the life of the building to be served (or until sewers are available) is necessary with prescriptive codes where renewable operating permits are not employed, because active regulatory agency involvement usually ceases after the system is constructed. Therefore, the wastewater characteristics and flows must be projected over the life of the building to be served. This, of course, is difficult, particularly for commercial establishments. Typically, prescriptive codes provide guidelines for maximum flow estimates based on the size of the building, but they are rarely accurate. As a result, systems are frequently over- or under-sized. While guidelines for estimating wastewater characteristics and flows can be useful, designs of systems should be based on reasonable characteristics and flows expected over a specified design life and an operating permit written based on these parameters. Review of the permit at renewal can identify any need to modify the system if wastewater characteristics have changed or the design flow has been exceeded.

6.2.1.2: <u>Performance Based Design Codes</u>

"Engineered" systems, designed to meet performance standards, must be submitted for design review. While it is the responsibility of the designer to select and size the appropriate system design, the reviewing agency must still satisfy itself that the proposed design is reasonable for the site. Therefore, minimum documentation should be submitted with the design for support.

Submitted designs may be either "alternative" or "innovative" ("experimental"). Alternative designs, in this context, are designs that are proven, but not included in the prescriptive design code. Innovative or experimental designs, on the other hand, are designs which do not have documented performance data for the given operating conditions. Documentation that should be submitted with the proposed design includes:

- Description of the system, its unit processes and method of operation,
- Previous experience and performance data,
- Operation and maintenance procedures and operator qualifications, and
- Contingency plans for correcting performance failures.

If the designer can satisfactorily demonstrate that the proposed system is an "alternative" system, a conventional operating permit would be issued with appropriate limitations addressing normal and emergency operation procedures. "Innovative" systems are not commonly approved for new construction unless a conventional system can be constructed on the site if the proposed system fails to perform. Permits for innovative systems should include specific and detailed monitoring protocols and reporting procedures. A signed statement should be included with the design package that the owner is aware of the experimental nature of the system and that replacement with a conventional or alternative system may be required if an uncorrectable performance failure occurs.

6.2.2: Current Florida Program

6.2.2.1: <u>Design Requirements</u>
Chapter 10D-6 uses a prescriptive approach to design. Design approval by the county health unit includes a review of the proposed system type, its horizontal and vertical placement on the lot, and sizing criteria used. Design details and drawings are not required. Construction inspections by the county health unit are used to confirm that the system is actually designed and constructed according to code requirements.

To obtain design approval, the owner or owner's representative must file an "Application for System Construction Permit" (HRS-H Form 4015). Completion of this application includes providing a site evaluation report and description of the proposed building and its intended use. The site evaluation report should provide estimates of the daily wastewater flow, the unobstructed area needed for the estimated flow, and the available unobstructed area that meets applicable soil and setback requirements. The site evaluation report should also identify the type of subsurface infiltration system that is to be used, the appropriate wastewater hydraulic loading rate for the soils encountered, an established benchmark, and the elevation in the soil profile that the infiltrative surface should be constructed. Information should be given in the remainder of the application which provides a description of the proposed building, including type of use, and size and type of water use fixtures. Business establishments must provide additional detailed information regarding waste characteristics (HRS-H Form 4018A). A plot plan which is drawn to scale showing property boundaries, slopes and drainage features, all necessary setback distances including those from adjacent lots, and the location of all system components must be attached to the application.

If the design concept and sizing criteria are accepted by the county health unit, a construction permit is issued. Construction must follow the construction requirements in Chapter 10D-6. Compliance with the code is verified through construction inspections conducted by personnel of the county health unit. Through this process of construction permit application and construction inspection, treatment system concept and the system design review requirements are satisfied.

This approach to design requires that the site evaluator determine the type, size and placement of the onsite treatment system. If the determinations of the site evaluator are acceptable, the construction contractor must follow the construction details prescribed by the code for the type of system selected. Prescribed construction requirements are provided for infiltration trenches and beds constructed in natural soil (10D-6.056) and mounds and filled systems constructed in imported fill materials (10D-6.049).

Daily wastewater flow rates may be estimated either by using guidelines provided in the code (Table II, 10D-6.048) or by providing metered water use data from six similar establishments over the most recent 12 month period. The guidelines provided are typical maximum day estimates for various types of establishments and uses based on an assumed maximum capacity. Single family homes are sized according to the number of bedrooms or total floor area in the home. If ultra low volume flush toilets are used exclusively, a 15% reduction is granted for residential applications and a 25% reduction is allowed for non-residential applications where food is not prepared (10D-

6.048(1)(b)). The reduced flows can only be applied to the infiltration system sizing. Pretreatment unit sizing must use flow estimates based on conventional water fixture use.

Pretreatment unit sizing is based on the daily wastewater flow estimates. Wastewater strength is not specifically addressed by the sizing criteria presented. However, for estimated flows exceeding 1500 gpd or where commercial wastewater is to be treated, chambered septic tanks or tanks in series are required (10D-6.048 (2)). Restaurants must also install grease interceptors for the kitchen wastes.

Sizing of the infiltration system is based on the estimated daily wastewater flow and the texture of the soil at the site (10D-6.048(5)). Acceptable infiltration rates vary from 0.25 to 0.5 gpd/ft^2 in clayey soils and 1.25 to 1.75 gpd/ft^2 in coarse sands for bed and trench systems, respectively. The strength of the wastewater discharged to the system is not considered except where extended aeration package plants are used for pretreatment. In such instances, the size of the infiltration system may be reduced by 25% because of the additional treatment that is provided by the package plant (10D-6.0541(2)(f)). If ultra low volume flush toilets are used, system sizing can be reduced by 15 to 25% because of the reduced hydraulic loading (10D-6.048(1)(b)), but this allowance has the impact of increasing the mass organic loading to the infiltrative surface. Other soil and infiltration system design factors which can affect design infiltration rates are not considered, nor is subsurface drainage of the renovated wastewater from the site evaluated.

Chapter 10D-6 (10D-6.056(4)) does promote shallow trench designs, but does not prohibit deep or bed designs. Trench widths must be 1.5 to 3 ft wide with a minimum separation distance between trenches of 2 ft. However, beds can be constructed with no restriction on width, but the maximum design wastewater application rate is reduced 20 to 50% from that allowed for trenches. The maximum length of the trenches or beds is limited to 100 ft. Depth of the distribution pipe below final grade is limited to 2 ft, but no maximum depth of the infiltrative surface is specified. The minimum depth from final grade to the infiltrative surface allowed is 18 inches.

Gravity distribution with 4-inch diameter perforated pipe is used to distribute wastewater to the infiltration system (10D-6.056(1)). Distribution boxes or header pipes are used to split wastewater flows between multiple distribution lines. However, dosing, using pumps or siphons, is required where the total infiltrative surface area exceeds 1000 ft^2 (10D-6.056(3)).

Design features that facilitate active management of the infiltration systems are not addressed in the code. However, an unobstructed area, suitable for construction of a replacement subsurface infiltration system must be reserved on each site for system repair or expansion (10D-6.046(4)).

6.2.2.2: Requirements for Non-Conforming Designs

On sites where the prescriptive requirements cannot be met or where alternative systems designs are needed, non-conforming designs are allowed by Chapter 10D-6. Variances may be granted where minor deviations from code requirements are needed to relieve or prevent excessive hardship (10D-6.045). In requesting the variance, the applicant must show that the intent of the code, to protect public health and groundwater and surface water quality, will still be met by the change.

Alternative systems designed by a licensed professional engineer may be approved "...where evidence exists that use of such systems will not create sanitary nuisance conditions, health hazards or pollute receiving waters..." (10D-6.048(5)). Procedures for routine maintenance, operational surveillance and environmental monitoring to assure the system continues to function properly also may be required as part of the submittal. No other requirements are established for alternative system review.

Experimental systems are defined as systems which are not specifically addressed by the code and are approved by the State Health Office for limited testing and evaluation (10D-6.042(22)). The design submittal must include detailed design and construction plans, results of previous testing, and monitoring plans. The manufacturer or other legally authorized person must agree to replace the experimental system with a code compliant system if failure occurs, and provide a performance bond in an amount sufficient to cover the costs of replacement. No specific performance criteria or monitoring procedures are specified by the code.

6.3: CONSTRUCTION

6.3.1: Minimum Program Requirements

Proper construction of onsite wastewater treatment systems is critical to successful performance. Since most onsite systems are designed to utilize soil in its native state, native conditions must be preserved. A frequent cause of system failure is poor construction practices which damage the soil (U.S. EPA, 1985). Once damaged, native conditions can seldom be restored. Therefore, it is imperative that an onsite treatment system program ensure proper system construction.

The role of the regulatory agency in system construction is to confirm that:

- Construction conforms to the approved plan,
- Appropriate construction methods are employed, and
- Acceptable construction materials and equipment are used.

Confirmation of these aspects of construction must be accomplished through on-site inspections by the agency. If the on-site inspection reveals any non-conformance, effective enforcement actions which can be taken against the owner and contractor must be available. To allow enforcement actions to be taken, however, standards must be established against which construction practices can be measured.

Standards by which construction inspections are made must be established by the program. The standards may be established in the code (prescription) or established as part of the design review through approval of construction documents containing construction drawings and specifications.

6.3.1.1: On-Site Inspections

On-site inspections by the regulatory agency are necessary to confirm that system construction conforms to the approved plan and procedures. To be effective, inspections must occur at appropriate points during construction. Inspection for conformance to the approved plan usually can be done near completion of construction, but before covering of the components. This time also can be used to inspect materials and equipment used. Inspection for confirming appropriate construction procedures should be done during the period that the infiltrative surface is exposed. Damage to this surface can significantly impact performance. If inspections cannot be performed as construction proceeds, then the exposed surface should be inspected for damage before the surface is covered with aggregate or other materials. Once covered, damage to the infiltrative surface cannot be observed and corrected.

6.3.1.2: Conformance to Design

Whether the program is prescriptive or performance based, a system design plan must be approved before construction can proceed. This plan is used during construction to see that the approved system design is followed. With a prescriptive design code, design drawings may not be required for approval. Instead, the design must conform to design requirements presented in the code. It is against this prescribed design that conformance by the construction is measured. On the other hand, with performance based programs, design drawings and construction specifications must be submitted for approval. The approved plans must be followed in this case and should be on site during construction for reference by both the contractor and inspector. Upon completion of construction, accurate record drawings of the system should be provided for the regulatory authority and the owner. Record drawings are necessary for operation and maintenance of the system, and for review and approval of additions and repairs to the system.

6.3.1.3: Appropriate Construction Methods

Because of the significant impact that construction methods can have on system performance, acceptable minimum construction standards should be made part of a regulatory program. This is true whether the program is prescriptive or performance based. Wastewater infiltration and percolation through the soil require that soil pores remain open. If soil pores are damaged during construction, system failure is likely to result. Unfortunately, any damage that does occur is difficult or impossible to correct. Resulting changes in siting or design might have to be made. Therefore, the purpose of the construction guidelines should be to establish the minimum criteria within which the contractors must operate.

Procedures which may impact the permeability of soil must be carefully planned so damage is kept to minimum. Excavation, placement of gravel or sand fill, and backfilling operations must be done with the proper equipment and with care (U.S. EPA, 1980). Only low load-bearing construction equipment should be used in the design area. Construction should not proceed if the soil is near its plastic limit. If mound construction or other filling operations are anticipated, the site should be carefully cleared of trees and brush by cutting trees at ground level and mowing and raking the site before chisel plowing the area to a depth of 8 to 12 inches along the slope contour. Once exposed, the infiltrative surface should be covered within 12 hours to prevent desiccation or before periods of precipitation to prevent puddling.

6.3.1.4: Acceptable Materials and Equipment

Materials and equipment used in the construction of the onsite system such as pipe and fittings, joints and sealants, aggregate, bedding materials, pumps and controls, etc., should conform to generally accepted standards. Such standards include the American Society of Testing and Materials (ASTM), National Sanitation Foundation (NSF), Underwriters Laboratory (UL), etc. References to the appropriate standards should be made in the code for those items commonly used. For items that do not have established universal standards, it may be necessary to develop specific standards in the code. This is commonly done for prefabricated treatment tanks. If done, the standards must include product review procedures and testing requirements. Periodic quality control tests should also be included in the standard.

6.3.1.5: New Products

The program should also address new products that may be submitted by manufacturers for approval. Product approval procedures should be clearly defined. These procedures should include pilot or full scale testing over specific or agreed upon time periods for comparison against specific, pre-determined performance standards.

6.3.2: Current Florida Program

6.3.2.1: On-Site Inspections

The Florida onsite wastewater treatment system program requires that construction inspections be performed by the regulatory agency prior to covering the system with backfill (10D-6.043(2)). One inspection is required. No attempt is made or required to inspect the exposed infiltrative surface of the system. The contractor must notify the county health unit when the inspection can be performed. The county health unit must make a reasonable attempt to inspect the uncovered system within two working days of notification. If the inspection is for a repaired system, the department must make a reasonable effort to inspect within one working day of notification.

6.3.2.2: Conformance to Design

Approval of construction is required before the system can be put into operation and the building occupied (10D-6.043(2)). The principal purpose of the inspection is to confirm that the construction conforms to the design. Because Chapter 10D-6 uses a prescriptive approach to design, construction drawings are not required for conventional onsite treatment systems. Construction inspection, therefore, is used to observe system layout, sizing, and materials of construction, and to compare it to the prescriptive requirements. No record drawings of the constructed system are required as part of the construction approval.

6.3.2.3: Appropriate Construction Methods

Chapter 10D-6 is silent on acceptable construction procedures. However, only registered contractors may install onsite systems. To maintain registration, the contractors must successfully complete six classroom hours of instruction regarding the public health and environmental impacts of onsite systems (10D-6.073(3)). This continuing education requirement is to raise the awareness of the contractors regarding the need for care and proper procedures in construction. However, this provision does not apply to property owners who choose to construct their own systems. Owner constructed systems are permitted without the owners having to demonstrate knowledge of onsite system construction (10D-6.070(4)). Without specific construction procedure guidelines or a requirement for a pre-approved plan of construction, the regulatory agency has no enforcement powers until the system fails to meet the established performance standards.

6.3.2.4: Acceptable Materials and Equipment

Material specifications are included in Chapter 10D-6 for various materials and equipment. Specifications using ASTM and Florida Department of Transportation standards are included for distribution piping and aggregate (10D-6.056(2) & (4)). Similar standards are followed for materials used in construction of treatment tanks. However, for many other system components no specifications are established, nor are the designer or contractor required to submit material or equipment specifications for approval. Only when a package extended aeration treatment unit is proposed may equipment specifications be required as part of the application for a construction permit (10D-6.0541).

All prefabricated treatment tanks must be approved before they are accepted for use (10D-6.054). To obtain approval, the manufacturer must submit engineered drawings for review. If the drawings meet the requirements of the code, the manufacturer is issued an approval number after the county public health unit confirms that the manufacturer constructs the tanks as indicated in the submitted drawings. No provisions are provided in the code for random testing to confirm quality control after the approval number is issued.

6.3.2.5: <u>New Products</u>

Provisions are made in Chapter 10D-6 for the approval of new or experimental products. Such products may be installed for testing, but the total number approved for installation is limited to a specific number. The systems must be monitored by the department against specific performance criteria agreed upon between the manufacturer and the department. A performance bond is required by the manufacturer to cover the costs of replacing the experimental system or component with a conventional design if performance does not meet the established standards. It is not stated in the code, however, if the product will be approved for general use following a successful testing period.

6.4: OPERATION AND MAINTENANCE

6.4.1: Minimum Program Requirements

Continuous attention to operation and maintenance of onsite wastewater treatment systems should be required in all programs. Operation and maintenance requirements should be established so that performance is monitored and corrections or modifications to the system or its operation can be made to maintain performance results within established standards. Such operator intervention is necessary to prevent performance failures and extend the service life of the system.

The objective of operation and maintenance requirements is to ensure that the system performs satisfactorily over its service life. Traditionally, the service life has been the life of the building the system serves. Therefore, the regulatory agency has had little control over the system once it was installed. Recently, some programs have required system inspections for compliance with current code at the point of sale of the property. If noncompliant, the system must be brought up to code at the time of the sale. This provision has helped to eliminate many failing systems, but it still fails to give sufficient control of the system to the regulatory agency.

To provide adequate control of system performance, an operating permit which must be renewed on a periodic basis is necessary. Over the permit period, specific requirements for performance monitoring and reporting to the regulatory agency should be established. Where monitoring and reporting of performance are not practical, continuing operation and maintenance by a qualified operator should be demonstrated and a certified statement that the system is performing to requirements should be provided as a condition of permit renewal.

6.4.2: Current Florida Program

The Florida onsite wastewater treatment system program has few requirements for active maintenance of systems. Chapter 10D-6 requires that the owner assume responsibility for maintenance of the system, but no authority is given the department to enforce the requirement (10D-6.050). Only if a system demonstrates an obvious performance failure can the department take enforcement action. Typically, for an enforcement action to occur, the failure must first be brought to the attention of the department by a third party. There are no provisions for regular inspection of system operation or performance.

Demonstrated operation and maintenance may be required for non-conventional systems. The use of package aerobic treatment units requires that a maintenance contract be executed with a qualified maintenance entity (10D-6.0541(2)). The contract must be renewed annually for the life of the treatment unit. Minimum requirements for unit inspection are established and authority given the department to select a representative number of systems for effluent monitoring. Other alternative system designs may require routine maintenance be performed as a condition of design approval (10D-6.049(5)).

SECTION 7.0
QUALIFICATIONS OF SERVICE PROVIDERS

7.1: MINIMUM PROGRAM REQUIREMENTS

All service providers should have a basic understanding of onsite wastewater treatment system siting, design, construction, and operation. This is particularly true where performance based or a combination of prescriptive and performance based codes are used. There must be a higher degree of responsibility placed on onsite treatment system professionals for proper application of onsite technology. Agencies should integrate a qualifications program into the onsite treatment practice.

A qualifications program has two objectives. First, it must ensure that only qualified individuals perform services related to onsite wastewater treatment systems. Second, the program must ensure that onsite treatment system work is performed according to appropriate standards. Such a qualifications program should include;

- Certification or licensing procedures for all service providers that require periodic renewal,
- Examinations to qualify for licensure that demonstrate that the applicant has the basic knowledge, skills and experience necessary to perform onsite treatment system services,
- Requirements for continuing education for knowledge and skills maintenance,
- A definition of "standards of practice" that must be followed,
- Disciplinary guidelines to maintain program quality.

Professional certification should encompass five separate divisions to include site evaluator, designer, contractor, operator, and program regulator. Professionals could hold certifications for any or all of the five divisions provided they meet the necessary requirements. Cross training should be encouraged as part of the continuing education programs.

Certified professionals must possess an understanding of onsite treatment system function and operation. This includes functional performance from both a hydraulic and treatment standpoint. It also requires an understanding of how an onsite treatment system operates and impacts the environment within which it is installed. Finally, they must have a working knowledge of applicable codes and practices necessary to operate in a particular regulatory jurisdiction. A variety of specialized skills in the areas of soil science, engineering, hydrology, construction, public health and maintenance are also necessary.

Site Evaluator: The primary skill requirement for a site evaluator is a knowledge of soils. This knowledge should be based on soil science training versus soils engineering training. Due to its focus on the physical, chemical, and biological aspects of surficial unconsolidated geologic materials, soil science training is more appropriate for application of onsite system technology. In addition, such persons are trained in the USDA classification system, on which nearly all onsite treatment system siting programs are based. Site evaluators must also possess basic skills in the fields of engineering, surveying, geology, hydrology and botany.

Designers: Designers generally are either engineers or installing contractors, however, this does not preclude those involved in other aspects of onsite treatment. The designer must have the ability to interpret and review information presented on the site evaluation form, especially the soils data. A thorough understanding of onsite treatment systems function and operation is necessary for selecting the most appropriate system with relation to the site features. Finally, the designer must be capable of preparation of plans and specifications for approval and construction of the system.

Contractors: Installing contractors must possess a thorough understanding of onsite treatment system function and operation. In addition, a contractor must be aware of construction methods and materials to insure successful installation and resultant operation, and to facilitate inspection and maintenance needs.

Operators: An operator must be knowledgeable concerning system performance from a treatment and hydraulic standpoint. The operator must be familiar with the mechanical and non-mechanical components, sampling and monitoring requirements, and emergency operation.

Regulatory Personnel: At a minimum regulatory personnel should be certified as site evaluators or designers since their primary task is to evaluate proposed sites and designs for onsite treatment system construction. However, they must also be knowledgeable in all other aspects of onsite technology in order to make inspections and judgments regarding site conditions, designs, construction practices, evaluation of O&M programs, and failure conditions when they arise.

7.2: CURRENT FLORIDA PROGRAM

Florida has in place a program for certifying Septic Tank Contractors (10D-6 Part III). This program certifies individuals or firms that install OSDS and provide repair and maintenance services, including septic tank pumping. This program adequately addresses the type of program necessary, although it is not all encompassing for other phases of onsite technology.

Septic tank contractors are defined (10D-6.072(a-e)) as those persons performing onsite treatment system installation, repair, modification, maintenance, and septic tank pumping. Property owners are allowed to perform installation, alteration, maintenance, and repair (10D-6.070(4)) for the owner-occupied residence with no requirement for certification.

Applicants for certification as septic tank contractors must meet minimum requirements (10D-6.072(a)(b)(c)), demonstrate experience (10D-6.072(4)(a)), provide references (10D-6.072(4)(b)), and pass an examination 10D-6.072(7)). Annual recertification requires successful completion of six hours of approved classroom instruction (10D-6.073(3)). Standards of practice are set forth in 10D-6.075. Suspension, revocation, or denial of registration is covered under 10D-6.074 and disciplinary guidelines are covered under 10D-6.0751. 10D-6.077 covers the fee structures for application, initial registration, and renewal of registrations.

Site evaluators are required to be "qualified" persons with soils training (10D-6.044(3)) and (10D-6.0571(2)). There is no specification as to the type of soils training or requirement for their demonstration of knowledge.

There is no specific certification for onsite treatment system designers in Chapter 10D-6. However, there are references which state designs by an engineer registered in the State of Florida who is "qualified" in the field of onsite wastewater system design may be required (10D-6.044(4), 10D-6.049). There are also references where designs or other certifications are required by an engineer (10D-6.047(6), 10D-6.049(5)).

The Florida program does not acknowledge onsite treatment system operators. Currently, those individuals certified as Septic Tank Contractors are permitted to provide maintenance and septic tank pumping services. Aerobic treatment units do require an operator/maintenance entity be licensed as a Class D state certified operator under the provisions of Chapter 17-602, FAC (10D-6.0541(3),(b)).

Certification of regulatory personnel as part of the OSDS program is not addressed in Chapter 10D-6.

SECTION 8.0
REGULATORY REVIEW AND ENFORCEMENT

8.1: MINIMUM PROGRAM REQUIREMENTS

With careful siting, design, construction and operation, onsite systems provide an effective and environmentally sound method of wastewater treatment and disposal. Failures of systems to perform are not due to inherent flaws in system concepts, but to their inappropriate application or operation. This occurs primarily because regulatory control is absent or not enforced. However, the lack of adequate regulatory control is seldom recognized as the problem and, as a result, most efforts towards improving onsite system performance have been directed toward finding alternatives to the conventional septic tank system. Various alternatives have been developed or proposed, but they usually fail to gain acceptance because regulatory agencies still insist that onsite systems be designed for neglect, be simple enough for anyone unfamiliar with these systems to design and install, and be inexpensive. The beliefs that it is not possible to ensure that onsite systems will be maintained, that it is not practical to control those that design and install them, and that homeowners will not pay for any system that costs more than the conventional system are almost universally accepted. If onsite treatment systems are to be effective, a strong regulatory framework is necessary.

The objective of a regulatory program is to ensure that practices meet expectations. Functions of an onsite wastewater treatment system regulatory program include:

- Establishment of rules,
- Verification of rule compliance,
- Enforcement, and
- Record keeping.

8.1.1: Rules

Rules define expectations and the consequences of not meeting those expectations. The expectations must be clearly stated. They may either be prescriptive or performance based, but must include a description of the standards that are expected to be met and the necessary procedures to be followed to gain regulatory approval. An appeals or variance process is also necessary for allowing deviations from the established rules where conditions exist which preclude compliance, but do not significantly compromise the intent of the rules.

8.1.2: Verification of Compliance

Mechanisms to verify compliance with the rules must be established. Typical mechanisms are reviews, inspections, and reporting. Reviews are usually used to see that site evaluation procedures and interpretation of results meet regulatory standards. In some cases, follow-up on-site inspections may be made before approval for onsite treatment system use is granted. Reviews are also used to approve system designs before construction can begin. Compliance with approved construction documents and procedures is usually checked by on-site inspections. Inspections are also used to confirm that the system is operating properly. Ensuring that systems continually meet performance standards requires reporting of on-going system operation monitoring. Review and follow-up of monitoring reports must occur to maintain operation effectiveness. While procedures for review of submitted site or system information can be straight forward, special authority must be granted the regulatory agency to enter private property to perform inspections.

8.1.3: Enforcement

Without effective enforcement mechanisms, compliance with the rules will be lax. Frequently, onsite treatment system programs suffer because taking enforcement actions are cumbersome and invite court actions. It is imperative, therefore, that program expectations are clear, consistent, and specific.

Enforcement mechanisms include permits, bonding, and licensing/certification or registration. Permits are written warrants granted by a regulatory agency which convey the right to conduct a specific activity, usually at a specific location for a given fixed period of time. This technique places the burden of obtaining and supplying all necessary data and information on the permit applicant. It can be used to administer and enforce program standards simply by making compliance with these standards a necessary prerequisite for issuance of the permit. The regulations which establish the permitting process must also impose the requirement of making it unlawful to conduct the regulated activity without first obtaining a valid permit. Permits are typically required for construction activities, whether new system construction or existing system repairs, and for system operation. Construction permits are usually voided within one or two years if system construction has not commenced during that period. Operating permits are also limited in term, but are renewable if proof of compliance with performance standards can be shown.

Bonding or other surety may also be used to ensure compliance. Posting of a bond is used to assure proper construction or operation of a system. Such bonds are generally returned after a specific period of time.

Controls on the service providers may be used to indirectly ensure compliance. Licensing or certification usually requires that the requisite degree of skill be demonstrated by examination, or inferred from proven experience or training. The license is required to provide services and may be suspended or revoked if the services are shown not to conform to established standards. Often pre-existing licensing programs are relied upon such as those for professional engineers, soil scientists, geologists or trades (*e.g.* plumbing). However, specific licensing programs for onsite wastewater treatment system providers have been found to be more effective. Regulators may be reluctant to suspend or revoke more general licenses which may prevent a provider from performing other, unrelated services thereby removing the provider's livelihood. Another pitfall of licensing is that excessive "grandfathering" privileges may be given when the program is first initiated. Compromising of program credibility is the result. Registration is an alternative to licensing/certification, but registration is usually only a bookkeeping, non-discretionary recording of providers. While a registration can be revoked, possession of a registration does not provide assurance of competency. Therefore, registration places a greater surveillance burden on the regulatory agency.

8.2: CURRENT FLORIDA PROGRAM

8.2.1: Rules

The authority to regulate all onsite wastewater treatment systems in Florida was given to the Florida State Department of Health and Rehabilitative Services by the state legislature (Section 381.0065, Florida Statues). Chapter 10D-6 (Florida Administrative Code), "Standards for Onsite Sewage Disposal Systems", was promulgated by HRS to provide rules for onsite system regulation by the department and HRS county public health units.

An interagency agreement with the Department of Environmental Regulation (DER) allows HRS county public health units to supervise the construction of facilities designed to treat up to 5,000 gallons per day. All industrial wastewaters are referred to DER.

Chapter 10D-6 is a prescription based code which establishes minimum standards for practices in Florida. Counties may set more stringent standards by ordinance, however. The standards established are generally clear and provide specific requirements for system siting, design and construction, but lack specificity for treatment performance failures (see Section 5.0).

Rules which establish performance expectations for approval of new products are not specific (10D-6.049(5)). Apparently, the performance standards are established on a case by case basis.

8.2.2: Verification of Compliance

Both reviews and inspections are used to verify compliance with the rules. Review of site evaluation data and system design concept is a prerequisite of construction permit approval (10D-6.043 & 6.044). Designs are not reviewed since they are prescribed. "Pre-cover-up" inspections are used to confirm the system has been constructed according to the approved design standards (10D-6.043(2)). A comprehensive construction inspection checklist is used to complete the inspection (HRS-H Form 4016).

Routine inspections of system operation are not included in the program. It is not clear that the department has the authority to enter private property for this surveillance function. Reporting of operation performance is also not a requirement of the program.

Inspections of products used in onsite systems is limited to review of design drawings and construction specifications followed by a pre-approval inspection at the manufacturing facility to confirm that the product is constructed according to the construction specifications and code requirements (10D-6.054; 6.0541). Once approval of new products is granted for use in Florida , no provisions for random quality control checks are provided.

8.2.3: Enforcement

Chapter 10D-6 relies on permits and licensing for program enforcement. Permits apply to new construction and repairs (10D-6.043(1)), and operation of systems which serve a building located in an area zoned for industry or manufacturing or one that will generate commercial sewage, or where an aerobic treatment unit is installed (10D-6.043(4)).

All septic tank contractors in Florida must be licensed by the department. Septic tank contractors are defined as those persons who construct repair or maintain onsite treatment systems. To obtain a license, the applicant must have at least three years active experience with a licensed septic tank contractor and pass an examination administered by the department. This license must be renewed annually. Six hours of approved classroom instruction is a condition of renewal. Specific guidelines for disciplinary actions including suspension or revocation of the license are established in the code (10D-6.074, 6.075, 6.0751). No licensing, registration, or continuing education of other service providers is required except where an alternative system is proposed. In such cases, a professional engineer, registered in Florida must design the system (10D-6.049(5)).

There do not appear to be strong enforcement mechanisms regarding "approvals" which are granted. Products which must obtain approval for use in onsite systems, including treatment tanks, have no surveillance program for quality control. Once product approval is granted no enforcement mechanism seems to be available other than to deny individual system construction approvals.

SECTION 9.0
RECOMMENDATIONS

This section presents recommendations for improving onsite wastewater treatment system practices in Florida based on the results of the Florida OSDS Research Project, published literature, and the extensive research and practical applications experience of the project team.

9.1: PROGRAM STANDARDS

Like most state programs, Florida uses a prescriptive based approach to siting and design of onsite wastewater treatment systems. This approach defines in detail system design and required site characteristics for system construction. In other words, the site must fit the prescribed system design. Onsite system programs need to place greater emphasis on the impacts that site characteristics have on system performance. The following program improvements are recommended.

- Florida should develop and implement a performance based program for siting, design, construction, and management of onsite wastewater treatment systems. Performance based programs define the performance goals which must be met by an onsite wastewater treatment system for a particular site. This approach is significantly different from a prescriptive based approach because it requires that the system design fit the site to meet not only established public health but also environmental goals.

- Measurable performance standards need to be established for protection of water quality. Chapter 10D-6 implicitly accepts the treatment performance of the conventional septic tank/subsurface wastewater infiltration system. HRS should accept and explicitly define this as the minimum standard of performance or define some other minimum performance standard.

- Performance standards for areas of specific environmental concern should be established as needed for additional water quality protection. For example, systems constructed over unconfined, sole source aquifers or adjacent to Outstanding Florida Waters or other specially protected surface water bodies may require more stringent performance standards.

- Prescriptive design options which the department accepts as meeting the minimum standard of performance for sites meeting specific criteria should be continued. These designs could be used by the designer in lieu of a performance-based design where site conditions allow. If a performance based design option is used, it should not be required to meet any higher standard.

9.2: TECHNICAL GUIDELINES

Sound technical guidelines provided in a performance based program serve to reduce costs. They facilitate design approval, reduce the level of system surveillance required, and give service providers and regulators a technical basis for what is acceptable in the program. .Chapter 10D-6 has established a basic foundation for such guidelines but needs strengthening. The following program improvements are recommended.

9.2.1: Site Evaluation

- Backhoe excavated soil pits which expose undisturbed soil profiles to a depth of eight feet or groundwater should be used for evaluation of soil characteristics. Proper assessment of soil characteristics for subsurface wastewater infiltration systems requires *in situ* observation of soil profiles to evaluate the capability of the soil for wastewater treatment and subsurface drainage. Hand augured borings may be used to confirm soil variability on the site, but are not acceptable for detailed soil descriptions.

- Soil profile descriptions should address those soil properties that affect permeability of the soil to air and water. In addition, texture, structure, horizonation, and relative bulk densities of each soil horizon should be described. The degree of accuracy of these descriptions should be that which a properly trained evaluator can make using simple field observations and procedures.

- Monitoring of seasonally high groundwater elevations should be required prior to plan approval where soil characteristics and other site indicators are inadequate to determine the wet season water table with confidence. As part of this requirement, acceptable monitoring conditions must be established to qualify results.

- All proposed large subdivision developments that are planned with onsite system use should be required to construct a network of wells to determine wet season water table elevations prior to development. The results of monitoring should be entered into a data base maintained by the department for use in siting onsite systems in the subdivision as well as surrounding areas.

- Consideration should be given to the use of a simple nitrogen mass balance model, such as the model developed by the National Association of Home Builders Research Foundation, to assist in determining maximum onsite system densities in future subdivision developments. Although monitoring of groundwater in high density subdivisions did not indicate that widespread or significant groundwater quality impacts have occurred below the subdivisions, the age and existing home densities may have been too low to show impacts. Mass balance considerations suggest that with time, impacts may be detected. Until more is known about the fate of various wastewater parameters in soil, a nitrogen mass balance model would serve to provide a conservative preliminary estimate of allowable housing densities.

9.2.2: Design

- A two-foot separation of the infiltrative surface from seasonally high water tables or zones of periodic saturation should be maintained. From performance monitoring of onsite systems in Florida, it appears that nearly complete removal of biodegradable organics, fecal indicator bacteria, ammonia, toxic organics (VOCs) and surfactants is achieved provided a true, two-foot unsaturated zone of acceptable soil quality exists. Nitrate nitrogen is able to move through this zone, however, increasing the unsaturated thickness does not significantly increase removal. The time period over which the soil attenuates phosphorus will increase with greater unsaturated separation distances. Limited monitoring of virus movement below systems indicated that virus may penetrate a two-foot unsaturated zone. The impacts of the capillary fringe above the water table could not be determined, but is expected to be significant in fine sands because it reduces soil permeability to air. Therefore, greater separation distances may be required in soils which exhibit a high capillary fringe. Further study of the impacts of this phenomenon is recommended.

- To control soil clogging, wastewater infiltrative surfaces should be sized based on the carbonaceous plus nitrogenous BOD loadings. Hydraulic loadings established in Chapter 10D-6 appear to be appropriate for residential systems, but should not be used for commercial establishments. Maximum organic loadings should be based on those derived from current domestic hydraulic loadings.

- Water conserving plumbing fixtures should be strongly promoted to reduce hydraulic loadings. However, allowing reductions in infiltrative surface sizing based on reduced flow alone should be considered carefully. Water conserving fixtures can significantly reduce wastewater volumes, but they may not reduce the organic or solids mass loadings. Allowance for drainfield size reduction with ultra low volume fixtures should be limited to ten percent.

- Narrow wastewater infiltration surfaces should be promoted. Infiltration surfaces wider than six feet should not be allowed. Narrow geometries enhance performance due to increased infiltrative surface area and soil aeration. To allow narrow geometries, the length of infiltration systems should not be limited to one-hundred feet as they are currently.

- Maximum depth to the infiltrative surface should be established at four feet providing adequate separation from seasonally high zones of saturation can be maintained. Shallow placement of wastewater infiltration systems enhances performance due to increased soil aeration and evapotranspiration.

- Onsite treatment systems should be designed for management. The current code should include the following basic requirements which promote proper system management:

    1. Septic tank servicing manholes should be brought to final grade with a securable cover,

    2. Infiltrative surfaces should be dosed to improve effluent distribution and prolong the life of the system,

3. Infiltration systems should be constructed with a minimum of two separate SWIS with a means for system rotation to allow periodic "resting" of infiltrative surfaces. A wastewater infiltration system cannot be properly managed when only a single cell is available for effluent disposal,

4. Observation ports should be required in each infiltration area to allow monitoring of effluent ponding levels above the infiltrative surface, and

5. Detailed record drawings of all system components and their locations should be provided to the local regulatory agency and the homeowner along with general guidelines for system operation and maintenance.

9.2.3: Construction

- Decisions regarding onsite system design should not be made solely by the contractor. Under current rules, the site evaluator determines the general location, type of system, and elevation of the infiltrative surface. The contractor uses this information to layout the system and construct it according to the prescribed design. Design decisions should be made by the site evaluator and designer prior to design approval. Necessary modifications to the approved design made during construction should be made in consultation between the site evaluator, designer, contractor, and regulator.

- Tanks used for onsite systems must be water tight and of proper design and strength to perform their intended function. Tank standards should be increased to include inlet and outlet pipe boss stops, single piece lids, water tightness testing, and quality control testing. Site constructed tanks of brick or block should not be permitted.

- Geotextile fabrics should be used for the porous barrier material to prevent piping of backfill materials into the infiltration system aggregate. Untreated building paper quickly disintegrates and provides little system protection.

9.2.4: Operation and Maintenance

- Short term, renewable operating permits should be required of all onsite wastewater treatment systems. Routine maintenance must be performed to extend system life and prevent system failures. Under the current program, system maintenance is not assured. Operation performance should be certified through reporting of required system maintenance and monitoring by a qualified third party, and should be a condition of permit renewal.

- Operating permits for onsite treatment systems serving commercial establishments and multi-family dwellings should include maximum flow limits. Flow monitoring based on water use should be used to prevent hydraulic overloading. The data collected would be useful in future estimation of daily flows for similar establishments.

9.2.5: Code Organization

- Chapter 10D-6 should be written clearly in an organized, logical manner. The current code does not place all guidelines related to one aspect together. For example, site evaluation and system design requirements are scattered within several sections of the code. All site evaluation information should be placed in one section and all design information in another for ease of use.

9.3: QUALIFICATIONS OF SERVICE PROVIDERS

Regulatory programs based on performance standards place greater responsibilities on the regulating agency, site evaluator, design professional, construction contractor, system operator, and system owner. This requires a greater level of knowledge of onsite treatment system function than currently exists. The following recommendations are provided as guidance for development of a more knowledgeable community of onsite treatment system service providers.

- Certification or licensing of all onsite wastewater treatment system service providers should be established. As environmental protection requirements become greater in Florida, the needs for performance based onsite system designs will increase. Performance based designs will result in the need for more skilled operation and maintenance personnel as systems become more sophisticated. Service providers and regulators must have a better understanding of soil, hydrology, treatment principals and processes, construction requirements and operation skills.

- Certification to practice must be by examination, with continuing education in relevant topics a condition of annual or biannual renewal. Guidelines for suspension and revocation should be defined.

- An onsite wastewater treatment system operator licensing program should be established. Qualified operators are particularly needed for alternative designs. Better control over the fate of system operation is needed to ensure proper performance and prolong system life.

- Regulatory personnel should possess similar levels of certification as the service providers they regulate. They must be as knowledgeable in site evaluation, system design, construction, and operation and maintenance. They should possess relevant college degrees. An apprenticeship training program should be established for new hires.

- Property owners, unless certified, should not be allowed to design or construct their own system. This provision of Chapter 10D-6 should be repealed.

9.4: REGULATORY REVIEW AND ENFORCEMENT

Performance based designs will require greater regulatory review and enforcement. The regulating agency must establish performance standards and develop the competency to review and approve the system designs which may be submitted to meet those standards. The following recommendations are provided as guidance for development of a stronger regulatory review and enforcement program.

- Consideration should be given to establishing a code that is uniform, state-wide. The current minimum code allows counties or municipalities to adopt more strict requirements by ordinance. This can make it more difficult to enforce program standards. It also creates confusion for service providers who cross political jurisdictional boundaries.

- Detailed specifications, design calculations, and scaled drawings should be required with all applications for system approvals. To streamline the regulatory review process, more detailed information on system design and construction is necessary. Standard specifications and drawings could be provided by the agency for approved prescriptive design options.

- More thorough construction inspection is needed, particularly of the wastewater infiltration system, to ensure proper operation of the completed system. Optimal onsite system performance cannot be achieved without proper construction.

- The term **"onsite sewage disposal systems (OSDS)"** should be eliminated from the Florida code. Properly designed, installed, and operated onsite wastewater systems provide significant levels of wastewater treatment before final discharge to groundwater. They are more than sewage disposal systems and should be labeled and considered as such. We recommend "onsite wastewater treatment systems (OWTS)" as a better term.

SECTION 10.0
REFERENCES

Aher, A., A. Chouthai, L. Chantrasekar, W. Corpening, L. Russ, and B. Vijapour. 1991. East Bay Municipal Utility District Water Conservation Study. Rep. No. R 219. Stevens Inst. Technology, Hoboken, New Jersey.

Alhajjar, B.J., J.M. Harkin, and G. Chesters. 1989. Detergent formula and characteristics of wastewater in septic tanks. J. WPCF 61(5):605-613.

Allison, L.E. 1947 Effect of microorganisms on permeability of soil under prolonged submergence. Soil Sci. 63: 439-450.

American Public Health Assoc.. 1989. Standard Methods for the Examination of Water and Wastewater. American Public Health Assoc., Washington, DC.

Anderson, D. and T. Konen. 1993. The impact of water conserving plumbing fixtures on residential water use characteristics: A case study in Tampa, Florida. Ayres Associates final rep. JN: 2317, Tampa, FL.

Anderson, D.L., A.L. Lewis, and K.M. Sherman. 1991. Human enterovirus monitoring at onsite sewage disposal systems in Florida. p. 94-104. In On-Site Wastewater Treatment. Individual and Small Community Sewage Systems. Proc. Sixth National Symp. on Individual and Small Community Sewage Systems. Chicago, IL, Dec. 16-17, 1991. ASAE, St. Joseph, MI.

Anderson, D.L., J.M. Rice, M.L. Voorhees, R.A. Kirkner, and K.M. Sherman. 1988. Groundwater modeling with uncertainty analysis to assess the contamination potential from onsite sewage disposal systems in Florida. p. 264-273. In On-Site Wastewater Treatment. Proc. Fifth National Symp. on Individual and Small Community Sewage Systems. Chicago, IL, Dec. 14-15, 1987. ASAE, St. Joseph, MI.

Anderson, D.L. and R.L. Siegrist. 1989. The performance of ultra-low-volume flush toilets in Phoenix. J. AWWA 81(3):52-57.

Anderson, J.S. and K.S. Watson. 1967. Patterns of household usage. J. AWWA 59:1228-1237.

Andreoli, A., N. Bartilucci, R. Forgione, and R. Reynolds. 1979. Nitrogen removal in a subsurface disposal system. J. WPCF 51(4):841-854.

Avnimelech, Y. and Z. Nevo. 1964. Biological clogging of sands. Soil Sci. 98: 222-226.

AWWA. 1989. Reservoir management for water quality and THM precursor control. American Water Works Assoc. Research Foundation, New York, NY.

Ayres Associates. 1985. Preliminary impact assessment of wastewater phosphorus control measures on phosphorus loads to Flathead Lake, Montana. Rep. to the Soap and Detergent Assoc., New York, NY.

Ayres Associates. 1987. Onsite Sewage Disposal System Research in Florida: Impact of Florida's growth on the use of onsite sewage disposal systems. Rep. to the Florida Dept. of Health and Rehab. Services, Tallahassee, FL.

Ayres Associates. 1989 (revised 1993). Onsite sewage disposal system research in Florida: Performance monitoring and ground water quality impacts of OSDSs in subdivision developments. Rep. to the Dept. of Health and Rehab. Services, Tallahassee, FL.

Ayres Associates. 1991. Study of septic tank performance at highway safety rest area wastewater treatment facilties. Rep. to Wisconsin Dept. of Transportation, Madison, WI.

Ayres Associates. 1991. Nitrogen removal from domestic wastewater in unsewered areas. Onsite nitrogen removal systems research/demonstration project. Phase I rep.. Wisconsin Dept. of Industry, Labor and Human Relations, Safety and Buildings Div., Madison, WI.

Ayres Associates. 1993a. An investigation of the surface water contamination potential from on-site sewage disposal systems (OSDS) in the Turkey Creek Sub-Basin of the Indian River Lagoon. Rep. to the Dept. of Health and Rehab. Services, Tallahassee, FL.

Ayres Associates. 1993b. The capability of fine sandy soils for septic tank effluent treatment: A field investigation at an in-situ lysimeter facility in Florida. Rep. to the Florida Dept. of Health and Rehab. Services, Tallahassee, FL.

Bauer, D.H., E.T. Conrad, and D.G. Sherman. 1979. Evaluation of on-site wastewater treatment and disposal options. U.S. EPA, Cincinnati, OH.

Bauman, B.J. 1989. Soils contaminated by motor fuels: Research activities and perspectives of the American Petroleum Institute. p. 3-19. In P.T. Kostecki and E.J. Calabrese (eds.) Petroleum Contaminated Soils, Vol. I, Remediation Techniques, Environmental Fate, Risk Assessment. Lewis Pub., Inc., Chelsea, MI.

Bendixen, T.W. et al. 1950. Soil clogging effects of septic tank effluents. p. 69-75. In Studies on Household Sewage Disposal Systems, Part II. Public Health Service, Environ. Health Serv., Cincinnati, OH.

Bennett, E.R. and E.K. Linstedt. 1975. Individual home wastewater characterization and treatment. Completion rep. series no. 66. Environmental Resources Cntr., Colorado State Univ., Fort Collins, CO.

Bennett, S.M., J.A. Heidman, and J.R. Kreissl. 1977. Feasibility of treating septic tank waste by activated sludge. U.S. EPA rep. no. EPA/600-2-77/141, District of Columbia, Dept. of Environ. Serv., Washington, DC.

Berg, G. 1964. The virus hazard in water supplies. J. New England Water Works Assoc. 78:79-104.

Berg, G. 1973. Microbiology-dectection and occurrences of viruses. J. WPCF 45:1289-1294.

Bicki, T.J., R.B. Brown, M.E. Collins, R.S. Mansell, and D.J. Rothwell. 1984. Impact of on-site sewage disposal systems on surface and groundwater quality. Rep. to Florida Dept. of Health and Rehab. Services. Inst. of Food and Agric. Sci., Univ. of Florida, Gainesville, FL.

Bitton, G., J.M. Davidson, and S.R. Farrah. 1979. On the value of soil columns for assessing the transport pattern of viruses through soil: A critical look. Water, Air, and Soil Pollution 12:449-457.

Bliss, E.S. and C.E. Johnson. 1952. Some factors involved in groundwater replenishment. Amer. Geography Union 33:547-558.

Bouma, J. 1975. Unsaturated flow during soil treatment of septic tank effluent. J. Environ. Engr. Div. ASCE 101: 976-983.

Bouma, J. et al. 1975. Soil absorption of septic tank effluent in moderately permeable fine silty soils. Trans. ASAE 18:1094-1099.

Bouma, J., W.A. Ziebell, W.G. Walker, P.G. Olcott, E. McCoy, and F.D. Hole. 1972. Soil absorption of septic tank effluent: A field study of some major soils in Wisconsin. Information Circular No. 20. Univ. of Wisconsin Ext. Geological and Natural History Survey, Madison, WI.

Brandes, M. 1972. Studies on subsurface movement of effluent from private sewage disposal systems using radioactive and dye traces. Ontario Ministry of the Environment, Toronto, Ontario.

Briley, K.F. et al. 1984. Alternative water disinfection schemes for reduced trihalomethane formation. Volume II. Algae as precursors for trihalomethanes in chlorinated drinking water. U.S. EPA rep. no. EPA-600/2-84-005. U.S. EPA, Cincinnati, OH.

Brown, K.W., J.F. Slowey, and H.W. Wolf. 1978. The movement of salts, nutrients, fecal coliform and virus below septic leach fields in three soils. p. 208-217 In Home Sewage Treatment. Proc. Second National Home Sewage Treatment Symp. Chicago, IL, Dec. 12-13, 1977. ASAE St. Joseph, MI.

Buckrop, W. 1992. Testimony before the Subcommittee on Investigations and Oversite, Aug. 4, 1992, Committee on Public Works and Transportation, House of Representatives, Washington, DC.

Buras, N. 1974. Recovery of viruses from wastewater and effluent by the direct inoculation method. Water Research 8:19-22.

Bureau of Economic and Business Research (BEBR). 1986. Florida estimates of population - Apr. 1, 1985. BEBR Population Program, Univ. of Florida, Gainesville, FL. p.48.

Burge, W.D. and N.D. Enkiri. 1978. Virus adsorption by fine soils. J. Environ. Qual. 7:73-76.

Cantor, L.W. and R.C. Knox. 1985. Septic Tank System Effects on Ground Water Quality. Lewis Pub., Inc., Chelsea, MI.

Carlile, B.L. 1980. Use of shallow, low-pressure injection systems in large and small installations. p. 371-385. In N.I. McClelland (ed.) Individual Onsite Wastewater Systems, Vol. 6. Proc. Sixth National Conf. 1979. NSP, Ann Arbor, Michigan..

Carlile, B.L., C.G. Cogger, and S.J. Steinbeck. 1981. Movement and treatment of effluent in soils of the lower coastal plain of North Carolina. North Carolina State Univ., Dept. of Soil Sci., Raleigh, NC.

Childs, K.E., S.B. Upchurch and B. Ellis. 1974. Sampling of variable waste-migration patterns in groundwater. Ground Water 12:369-377.

Cogger, C.G. and B.L. Carlile. 1984. Field performance of conventional and alternative septic systems in wet soils. J. Environ. Qual. 13:137-142.

Cohen, S. and H. Wallman. 1974. Demonstration of waste flow reduction from households. U.S. EPA rep. no. EPA-670/2-74-071. U.S. EPA., Cincinnati, OH.

Dagan, G. and E. Bresler. 1984. Solute transport in soil at field scale. p. 17-48. In B. Yaron, G. Dagan, and J. Goldshmid (eds.), Pollutants in Porous Media. Spring-Verlag, Berlin, Germany.

Daniel, T.C. and J. Bouma. 1974. Column studies of soil clogging in a slowly permeable soil as a function of effluent quality. J. Environ. Qual. 3:321-326.

DeWalle, F.B., D. Kalman, D. Norman, and J. Sung. 1985. Trace volatile organic removals in a community septic tank. Univ. of Washington, Dept. of Environ. Health, Seattle, WA. U.S. EPA rep. no. EPA/600/2-85/050. U.S. EPA, Water Eng. Research Lab., Cincinnati, OH.

Drewry, W.A. 1969. Virus movement in groundwater systems. OWRR-A-005-ARK (2). Water Resources Research Cntr., Univ. of Arkansas, Fayetteville, AR.

Drewry, W.A. 1973. Virus-soil interactions. In Proc. Landspreading Municipal Effluent and Sludge in Florida. Inst. of Food and Agric. Sci., Univ. of Florida, Gainesville, FL.

Drewry, W.A. and R. Eliassen. 1968. Virus movement in groundwater. J. WPCF 40:R257-R271.

Duboise, S.M., B.E. Moore, and B.P. Sagik. 1976. Poliovirus survival and movement in a sandy forest soil. Applied Environ. Microbiology 31:536-543.

Dudley, J.G. and P.A. Stephenson. 1973. Nutrient enrichment of groundwater from septic tank disposal systems. Upper Great Lakes Regional Commission, Univ. of Wisconsin, Madison, WI.

Duffy, C.P. et al. 1978. Technical performance of the Wisconsin mound system for on-site wastewater disposal - An interim evaluation. In Preliminary Environmental Rep. for Three Alternative Systems (Mounds) for On-Site Individual Wastewater Disposal in Wisconsin. Wisconsin Dept. of Health and Soc. Serv.

Ellis, B.G. and K.E. Childs. 1973. Nutrient movement from septic tanks and lawn fertilization. Michigan Dept. Natural Resources Tech. Bull. 73-5. Lansing, MI.

Ellis, B.G. and A.E. Erickson. 1969. Movement and transformation of various phosphorus compounds in soils. Soil Sci. Dept., Michigan State Univ., East Lansing, MI.

Erickson, A.E., and J.W. Bastian. 1980. The Michigan freeway rest area system - Experiences and experiments with onsite sanitary systems. In Individual Onsite Wastewater Systems. Proc. Sixth National Conf. Ann Arbor Sci. Pub. Ann Arbor, MI.

Federle, T.W. and G.M. Pastwa. 1988. Biodegradation of surfactants in saturated subsurface sediments: A field study. Ground Water 26(6):761-770.

Feige, W.A., E.T. Oppelt, and J.F. Kreissl. An alternative septage treatment method: Lime stabilization/sand-bed dewatering. U.S. EPA rep. no. EPA-600/2-75/036. U.S. EPA, Municipal Environ. Research Lab., Cincinnati, OH.

Fernald, E.A. and D.J. Patton. 1984. Water resources atlas of Florida. Inst. of Sci. and Public Affairs, Florida State Univ. Foundation, Tallahassee, FL.

Fetter, C.W. 1988. Applied Hydrogeology. Merrill Pub. Co., Columbus, OH.

Florida Administrative Code, State of Florida.

Florida Bureau of Comprehensive Planning. 1974. The Florida General Soils Atlas - With Interpretations. Florida Dept. of Admin., Div. of State Planning.
Regional Planning Districts 1 & 2 - DSP-BCP-S-1-5-74.
Regional Planning Districts 3 & 4 - DSP-BCP-S-15-74.
Regional Planning Districts 5 & 6 - DSP-BCP-S-28-74.
Regional Planning Districts 7 & 8 - DSP-BCP-S-29-74.
Regional Planning Districts 9 & 10 - DSP-BCP-S-30-74.

Frankenberger, W.T. et al. 1979. Bacterial effects on hydraulic conductivity of soils. Soil Sci. Soc. Amer. J. 43:333-338.

Ganzel, P.R. 1991. Personal communication, April. Washington Co. Dept. of Public Health, Stillwater, MN.

Gerba, C.P. 1982. Virus occurrence in ground water. In Microbial Health Considerations of Soil Disposal of Domestic Wastewaters, Proc. of a Conf. Univ. of Oklahoma, May 11-12, 1982. U.S. EPA rep. no. EPA-600/9-83-017. U.S. EPA, Cincinnati, OH.

Gerba, C.P. and B.H. Keswick. 1981. Survival and transport of enteric bacteria and viruses in groundwater. Studies Environ. Sci. 17:511-515.

Gerba, C.P., C. Wallis, and J.L. Melnick. 1975. Fate of wastewater bacteria and viruses in soil. J. Irr. Drain. Div. ASCE 101:157-175.

Gibbs, M.M. 1977a. Soil renovation of effluent from a septic tank on a lake shore. New Zealand J. of Sci. 20:255-263.

Gibbs, M.M. 1977b. Study of a septic tank system on a lake shore: Temperature and effluent flow patterns. New Zealand J. of Sci. 20:55-61.

Gilliom, R.J. and F.R. Patmont. 1983. Lake phosphorus loading from septic systems by seasonally perched groundwater. J. WPCF 55:1297-1305.

Goldsmith, J.D., D. Zohar, Y. Argaman, and Y. Kott. 1973. Effect of dissolved salts on the filtration of coliform bacteria in sand dunes. p 147-157. In S.H. Jenkins (ed.) Advances in Water Pollution Research. Pergamon Press, New York, NY.

Green, K.M. and D.O. Cliver. 1975. Removal of virus from septic tank effluent by sand columns. p. 137-143. In Home Sewage Disposal. Proc. National Home Sewage Disposal Symp. Chicago, IL, Dec 9-10, 1974. ASAE St. Joseph, MI.

Gupta, R.P. and D. Swartzendruber. 1962. Flow-associated reduction in the hydraulic conductivities of quartz sand. Soil Sci. Soc. Amer. Proc. 26:6-10.

Hagedorn, C. 1982. Transport and fate: Bacterial pathogens in ground water. p. 153-171. In Microbial Health Considerations of Soil Disposal of Domestic Wastewaters. Proc. of a Conf. Univ. of Oklahoma, Norman, May 11-12, 1982. U.S. EPA rep. no. EPA-600/9-83-017. U.S. EPA Cincinnati, OH.

Hain, K.E. and R.T. O'Brien. 1979. The survival of enteric viruses in septic tanks and septic tank drainfields. Water Resources Research Inst. rep. no. 108. New Mexico Water Resources Research Inst., New Mexico State Univ., Las Cruses, NM.

Hargett, D.L. 1984. Shallow, narrow trench dosed wastewater absorption systems; characteristics of the North Carolina low-pressure pipe system. U.S. EPA, MERL, ORD, Cincinnati, OH.

Hargett, D.L., E.J. Tyler, and R.L. Siegrist. 1982. Soil infiltration capacity as affected by septic tank effluent application strategies. p. 72-84. In On-Site Sewage Treatment, Proc. of Third National Symp. on Individual and Small Community Sewage Treatment. Chicago, IL, Dec.14-15, 1981. ASAE, St. Joseph, MI.

Harkin, J.M., C.J. Fitzgerald, C.P. Duffy and D.G. Kroll. 1979. Evaluation of mound systems for purification of septic tank effluent. Tech. rep. WIS WRC 79-05. Water Resources Cntr., Univ. of Wisconsin, Madison, WI.

Hathaway, S.W. 1980. Sources of toxic compounds in household wastewater. U.S. EPA rep. no. EPA-600/2-80-128. U.S. EPA, Municipal Environ. Research Lab., Cincinnati, OH.

Healy, K.A. and R. Laak. 1974. Site evaluation and design of seepage beds. J. Environ. Eng. Div., ASCE 100:1133-1146.

Hembra, R.L. 1992. Testimony before the Subcommittee on Investigations and Oversite, Aug. 4, 1992. Committee on Public Works and Transportation, House of Representatives, Washington, D.C.

Hillel, D. 1989. Movement and retention of organics in soil: A review and a critique of modeling. p. 81-86. In P.T. Kostecki and E.J. Calabrese (eds.) Petroleum Contaminated Soils, Vol. I, Remediation Techniques, Environmental Fate, Risk Assessment. Lewis Pub., Inc., Chelsea, MI.

Hills, D.J. and R.B. Krone. 1971. Hydraulically ventilated underground filter. J. San. Engr. Div. ASCE 97:851-866.

Hori, D.H., N.C. Burbank, R.H.F. Young, L.S. Lau, and H.W. Klemmer. 1971. Migration of poliovirus type II in percolating water through selected Oahu soils. In S.H. Jenkins (ed.) Advances in Water Pollution Research, Vol. 2.Pergamon Press, New York.

HRS. 1993. Environmental Health Program. Personal Communication.

Jansons, J., L.W. Edmonds, B. Speight, and M.R. Bucens. 1989a. Movement of virus after artificial recharge. Water Research 23:293-299.

Jansons, J., L.W. Edmonds, B. Speight, and M.R. Bucens. 1989b. Survival of virus in groundwater. Water Research 23:301-306.

Jennsen, P.D. and R.L. Siegrist. 1988. Nitrogen removal from wastewater in soil infiltration systems. In Fjerning Av Nitrogen I Avlopsvann. Tech. Univ. of Trondheim, Trondheim, Norway.

Johnson, C.E. 1957. Utilizing the decomposition of organic residues to increase infiltration rates in water spreading. EOS Trans., AGU 38:326-332.

Jones, J.H. and G.S. Taylor. 1965. Septic tank effluent percolation through sand under laboratory conditions. Soil Sci. 99:301-309.

Jones, R.A. and G.F. Lee. 1977a. Septic tank wasteewater disposal systems as phosphorus sources for surface waters. Occasional Paper no. 13. Colorado State Univ., Dept of Environ. Eng., Fort Collins, Co.

Jones, R.A. and G.F. Lee. 1977b. Septic tank wastewater disposal systems as phosphorus sources for surface waters. U.S. EPA rep. no EPA 600/3-77-129. U.S. EPA, Robert S. Kerr Environ. Research Lab. Ada, OK.

Jones, R.A. and G.F. Lee. 1979. Septic tank wastewater disposal systems as phosphorus sources for surface waters. J. WPCF 51:2764-2775.

Kaplan, J.E., G.W. Gary, R.C. Baron, N.Singh, L.B. Schonberger, R. Feldman, and H.B. Greenberg. 1982. Epidemiology of Norwalk gastroenteritis and the role of Norwalk virus in outbreaks of acute nonbacterial gastroenteritis. Annals of Internal Medicine 96:756-761.

Katz, B.G. and S.A. Plotkin. 1967. Minimal infective dose of attenuated poliovirus for man. Amer. J. Public Health 57:1837-1840.

Keswick, B.H., T.K. Satterwhite, P.C. Johnson, H.L. DuPont, S.L. Secor, J.A. Bitsura, G.W. Gary, and J. C. Hoff. 1985. Inactivation of Norwalk virus in drinking water by chlorine. Applied Environ. Microbiology 50:261-264.

Kiernan, K., C. Brown, M. Benjamin and J. Ferguson. 1983. Recreational vehicle waste disposal stations at highway rest areas. Final rep. to the Washington State Dept. of Transportation, rep. no. WA-RD 60.1.

Kirkner & Associates, 1987. Risk assessment of onsite sewage disposal systems for selected Florida hydrologic regions. Rep. to the Florida Dept. of Health and Rehab. Services, Tallahassee, FL.

Kolega, J.J. 1989. Impact of toxic chemicals to ground water. p.247-256. In R.W. Seabloom and D. Lenning (eds.) Proc. Sixth Northwest On-site Wastewater Treatment Short Course, Seattle, WA, 18-19 September 1989. Univ. of Washington, Seattle, WA.

Kristiansen, R.V. 1981a. Sand filter trenches for purification of septic tank effluent: I. The clogging mechanism and soil physical environment. J. Environ. Qual. 10:353-357.

Kristiansen, R.V. 1981b. Sand filter trenches for purification of septic tank effluent: II. The fate of nitrogen. J. Environ. Qual. 10:358-361.

Kropf, F.W. et al. 1975. Soil clogging in subsurface absorption systems for liquid domestic wastes. Prog. in Water Tech. 7:301-309.

Laak, R. 1970. Influence of domestic wastewater pretreatment on soil clogging. J. WPCF 42:1495-1500.

Laak, R. 1975. Relative pollution strengths of undiluted waste materials discharged in households and the dilution water used for each. Manual of Grey Water Treatment Practice - Part II. Monogram Industries, Inc., Santa Monica, CA.

Laak, R. 1976. Pollutant loads from plumbing fixtures and pretreatment to control soil clogging. J. Environ. Health 39:48-50.

Laak, R. 1986. Wastewater Engineering Design for Unsewered Areas, 2nd Ed. Technomic Pub. Co., Inc., Lancaster, PA.

Lance, J.C. and C.P. Gerba. 1980. Poliovirus movement during high rate land application of sewage water. J. Environ. Qual. 9:31-34.

Lance, J.C., C.P. Gerba, and J.L. Melnick. 1976. Virus movement in soil columns flooded with secondary sewage effluent. Applied Environ. Microbiology 32:520-526.

Lance, J.C., C.P. Gerba, and D.S. Wang. 1982. Comparative movement of different enteroviruses in soil columns. J. Environ. Qual. 11:347-351.

Lance, J.C. et al. 1973. Oxygen utilization in soils flooded with sewage water. J. Environ. Qual. 2:345-350.

LeBlanc, D.R. 1982. Sewage plume in a sand and gravel aquifer, Cape Cod, Massachusetts. Open-File Rep. 82-274. U.S. Geological Survey, Boston, MA.

Lefler, E. and Y. Kott. 1973. Enteric virus behavior in sand dunes. p. A73-A91. In Proc. of the Fourth Sci. Conf. of the Israel Ecological Society. Tel-Aviv, Israel.

Lefler, E. and Y. Kott. 1974. Virus retention and survival in sand. p. 84-91. In J.F. Malina, Jr. and B.P. Sagik (eds.). Virus Survival in Water and Wastewater Systems. Univ. of Texas. Austin, TX.

Lewis, A. and L. Stark. 1993. Florida OSDS Research Project: Viral study summary. HRS Tampa Branch Laboratory.

Ligman, K., N. Hutzler, and W.C. Boyle. 1974. Household wastewater characterization. J. Environ. Eng. Div. ASCE 150:201-213.

Linaweaver, Jr., F.P., J.C. Geyer, and J.B. Wolff. 1967. A study of residential water use. Johns Hopkins Univ., Dept. of Environ. Studies, Baltimore, MD.

Lippy, E.C. and S.C. Waltrip. 1984. Waterborne disease outbreaks -- 1946-1980: A thirty-five year perspective. J. AWWA 76:60-67.

Mahuta, Jr., F. and W.C. Boyle 1991. Gas transport in the unsaturated zone of soil absorption systems. p. 233-242. In On-Site Wastewater Treatment. Proc. Sixth National Symp. of Individual and Small Community Sewage Systems. Chicago, IL, Dec. 16-17, 1991. ASAE, St. Joseph, MI.

Martin, J.P. 1945. Microorganisms and soil aggregation: II. Influence of bacterial polysaccharides on soil structure. Soil Sci. 61:157-166.

Matthess, G. 1984. Unsaturated zone pollution by heavy metals. p. 79-122. In B. Yaron, G. Dagan, and J. Goldshmid (eds.) Pollutants in Porous Media. Spring-Verlag, Berlin, Germany.

McAvoy, D.C., C.E. White, B.L. Moore, and R.A. Rapaport. 1991. Sorption and transport of anionic and cationic surfactants below a Canadian septic tank/tile field. Environ. Toxicology and Chem. (in press).

McCalla, T.M. 1945. Influence of microorganisms and some organic substances on soil structure. Soil Sci. 59:287-297.

McCalla, T.M. 1945. Influence of microorganisms and some organic substances on water percolation through a layer of Peorian Leoss. Soil Sci. 59:287-297.

McCalla, T.M. 1950. Studies on the effect of microorganisms on rate of percolation of water through soils. Soil Sci. Soc. Amer. Proc. 15:182-186.

McGauhey, P.H. and R.B. Krone. 1967. Soil mantle as a wastewater treatment system. San. Eng. Research Lab. rep. no. 67-11. Univ. of California, Berkeley, CA.

McGauhey, P.H. and J.T. Winneberger. 1964. Studies of the failure of septic tank percolation systems. J. WPCF 36:593-606.

McGauhey, P.H. and J.T. Winneberger. 1965. A study of methods of preventing failure of septic tank percolation systems - Final Rep.. SERL rep. no. 65-17, San. Engr. Res. Lab., Univ. of California, Berkley, CA.

Meek, B.D. and L.B. Grass. 1975. Redox potential in irrigated desert soils as an indicator of aeration status. Soil Sci. Soc. Amer. Proc. 39:870-875.

Mitchell, R. and Z. Nevo. 1964. Effect of bacterial polysaccharide accumulation on infiltration of water through sand. Applied Microbiology. 12:219-223.

Mitchell, D.T. et al. 1982. Soil column comparison of aerobically pretreated wastewater to septic tank effluent. p. 266-272. In On-Site Sewage Treatment. Proc. Third National Symp. on Individual and Small Community Sewage Treatment. Chicago, IL, Dec. 14-15, 1981. ASAE. St. Joseph, MI.

Morris, J.C. and B. Baum. 1978. Precursors and mechanisms of haloform formation in the chlorination of water supplies. In R.L. Jolly, H. Gorshen, and D.H. Hamilton, Jr. (eds.) Water Chlorination--Environment Impact and Health Effects. Ann Arbor Sci., Ann Arbor, MI.

Morrison, S.M. 1983. A critique of microbiology research needs. p. 102-106. In R.C. Ward and S.M. Morrison (eds.) Research Needs Relating to Soil Absorption of Wastewater. Results of Workshop, Colorado State Univ., Fort Collins, CO, 1-3 June 1983. National Science Foundation, Washington, DC.

National Academy of Sciences. 1974. Water quality criteria, 1972. U.S. Government Printing Office, Washington, D.C.

Nestor, I. and L. Costin. 1971. The removal of Coxsackie virus from water by sand obtained from the rapid sand filters of water-plants. J. Hygiene, Epidemiology, Microbiological and Immunology 15:129-136.

Nevo, Z. and R. Mitchell. 1967. Factors affecting biological clogging of sand associated with ground water recharge. Water Research. 1:231-236.

Nykiel, D. 1983. A treatability comparison study of greywater and combined tank effluent using intermittent sand filtration. M.S. thesis. Univ. of Wisconsin, Dept. of Civil and Environ. Eng., Madison, WI.

Okubo, T. and J. Matsumoto. 1979. Effect of infiltration rate on biological clogging and water quality changes during artificial recharge. Water Resources Research 15: 1536-1542.

Okubo, T. and J. Matsumoto. 1983. Biological clogging of sand and changes of organic constituents during artificial recharge. Water Research 17:813-821.

Olsson, E., L. Karlgren, and V. Tullander. 1968. Household wastewater. Rep. 24:1968, National Swedish Inst. for Building Research, Stockholm, Sweden.

Otis, R.J. 1978. An alternative public wastewater facility for a small rural community. Small Scale Waste Management Project. Univ. of Wisconsin, Madison, WI.

Otis, R.J. 1985. Soil clogging: mechanisms and control. p. 238-250. In On-Site Wastewater Treatment. Proc. of the Fourth National Symp. on Individual and Small Community Sewage Systems. New Orleans, LA, Dec. 10-11, 1984. ASAE, St. Joseph, MI.

Pearson, F.C., P. Jenkins, J. McLean and S. Klein. 1980. Recreational vehicle waste disposal in roadside rest area septic tank systems. Final rep. to the Federal Highway Admin., Washington, DC, rep. no. FHWA/CA/UC-80/01.

Peavy, H.S. and C.E. Brawner. 1979. Unsewered subdivisions as a non-point source of groundwater pollution. National Conf. on Environ. Eng., San Francisco, CA.

Peavy, H.S. and K.S. Groves. 1978. The influence of septic tank drainfields on groundwater quality in areas of high groundwater. p. 218-225. In Home Sewage Treatment. Proc. of the Second National Home Sewage Treatment Symp. Chicago, IL, Dec. 12-19, 1977. ASAE, St. Joseph, MI.

Pekdeger, A. 1984. Pathogenic bacteria and viruses in the unsaturated zone. p. 195-206. In B. Yaron, G. Dagan, and J. Goldshmid (eds.) Pollutants in Porous Media. Spring-Verlag, Berlin, Germany.

Perlmutter, N.M. and E. Koch. 1971. Preliminary findings on the detergent and phosphate contents of water of southern Nassau County, New York. U.S. Geol. Survey Prof. Paper 750-D.

Polta, R.C. 1969. Septic tank effluents, water pollution by nutrients-sources, effects, and controls. Univ. of Minnesota. Water Resources Bull. 13:53-57.

Popkin, R. A. and T.W. Bendixen. 1968. Feasibility of improved subsurface disposal. J. San. Engr. Div., ASCE 94:271.

Preslo, L., M. Miller, W. Suyama, M. McLearn, P. Kostecki, and E. Fleischer. 1989. Available remedial technologies for petroleum contaminated soils. p. 115-125. In P.T. Kostecki and E.J. Calabrese (eds.) Petroleum Contaminated Soils, Vol. I, Remediation Techniques, Environmental Fate, Risk Assessment. Lewis Pub., Inc., Chelsea, MI.

Preul, H.C. 1966. Underground movement of nitrogen. Advanced Water Pollution Research 1:309-323.

Rail, C.D. 1989. Groundwater Contamination: Sources, Control, and Preventative Measures. Technomic Pub. Co., Lancaster PA.

Rapaport, R.A. 1991. Personal communication, September. The Proctor and Gamble Company, Cincinnati, OH.

Rea, R.A. and J.B. Upchurch. 1980. Influence of regolith properties on migration of septic tank effluents. Ground Water 18:118-125.

Reneau, Jr., R.B. 1977. Changes in inorganic nitrogenous compounds from septic tank effluent in a soil with a fluctuating water table. J. Environ. Qual. 6:173-178.

Reneau, Jr. R.B. 1979. Changes in concentrations of selected chemical pollutants in wet, tile-drained soil systems as influenced by disposal of septic tank effluents. J. Environ. Qual. 8:189-196.

Reneau, R.B. and D.E. Pettry. 1976. Phosphorus distribution from septic tank effluent in coastal plain soils. J. Environ. Qual. 5:34-39.

Rezek, J.W. and I.A. Cooper. 1980. Septage management. U.S. EPA rep. no. EPA-600/8-80-032. U.S. EPA, Cincinnati, OH.

Rice, R.C. 1974. Soil clogging during infiltration of secondary effluent. J. WPCF 46:208-216.

Roats, G.G. 1974. Effect of domestic wastewater pretreatment on soil clogging and aerobic and anaerobic conditions. M.S. Thesis, Univ. of Washington, Dept. of Civil Eng.

Robeck, G.C., N.A. Clarke, and K.A. Dostall. 1962. Effectiveness of water treatment processes in virus removal. J. AWWA 54:1275-1290.

Robertson, W.D. 1991. A case study of groundwater contamination from a domestic septic system: 7. Persistance of dichlorobenzene. Environ. Toxicology and Chem. (in press).

Robertson, W.D., J.A. Cherry, and E.A. Sudicky. 1989. Groundwater contamination at two small septic systems on sand aquifers. Ground Water 29(1):82-92.

Robertson, W.D., E.A. Sudicky, J.A. Cherry, R.A. Rapaport, and R.J. Shimp. 1989. Impact of a domestic septic system on an unconfined sand aquifer. In Kobus and Kinzelbach (eds.) Contaminant Transport in Groundwater. Balkema, Rotterdam. Germany.

Ronayne, M.P., R.C. Paeth, and S.A. Wilson. 1982. Oregon on-site experimental systems program. Final rep. to U.S. EPA, proj. no. 5806349. Oregon Dept. of Environ. Qual. Portland, OR.

Sandhu, S.S., W.J. Warreu, and P. Nelson. 1977. Trace inorganics in rural potable water and their correlation to possible sources. Water Research 12:257-261.

Sauer, P. 1991. Personal communication, August. Univ. of Wisconsin, Dept. of Soil Sci., Madison, WI.

Sawhney, B.L. 1977. Predicting phosphate movement through soil columns. J. Environ. Qual. 6:86-69.

Sawhney, B.L. and D.E. Hill, 1975. Phosphate sorption characteristics of soils treated with domestic wastewater. J. Environ. Qual. 4:343-346.

Scandura, J.E. and M.D. Sobsey. 1981. Survival and fate of enteric viruses in on-site wastewater disposal systems in coastal plains soils. Abstracts Annual Mtg. Amer. Society Microbiology. p. 175.

Schaub, S.A. and C.A. Sorber. 1977. Virus and bacteria removal from wastewater by rapid infiltration through soil. Applied Environ. Microbiology 33:609-619.

Sedlak, R. 1991. Personal communication, September. Soap and Detergent Assoc., New York, NY.

Segall, B.A., C.R. Ott, and W.B. Moeller. 1979. Monitoring septage addition to wastewater treatment plants, Vol. 1: Addition to the liquid stream. U.S. EPA rep. no. EPA-600/2-79-132. U.S. EPA, Cincinnati, OH.

Shaw, B. 1991. Personal communication. Univ. of Wisconsin, Stevens Point, WI.

Sherman, K.M, and D.L. Anderson. 1991. An evaluation of volatile organic compounds and conventional parameters from onsite sewage disposal systems in Florida. p. 62-75. In On-Site Wastewater Treatment. Proc. Sixth National Symp. on Individual and Small Community Sewage Disposal Systems. Chicago, IL, Dec. 16-17, 1991. ASAE, St. Joseph, MI.

Sherman, K.M., D.L. Anderson, D.L. Hargett, R.J. Otis and J.C. Heber. 1988. Florida's onsite sewage disposal system (OSDS) research project. p. 47-56. In On-Site Sewage Treatment. Proc. of the Fifth National Symp. on Individual and Small Community Sewage Treatment. Chicago, IL, Dec. 14-15, 1987. ASAE, St. Joseph, MI.

Shimp, R.J., E.V. Lapsins, and R.M. Ventullo. Biodegradation of linear alkyl benzene sulfonate (LAS) and nitrilotriacetic acid (NTA) in surface and subsurface soils and groundwater near a septic tank tile field. Environ. Toxicology and Chem. (in press).

Siegrist, R.L. 1983. Minimum-flow plumbing fixtures. J. AWWA 75(7):342.

Siegrist, R.L. 1986. Wastewater-induced soil pore clogging development as affected by effluent composition land loading rate. Ph.D. thesis. Univ. of Wisconsin, Dept. of Civil and Environ. Eng., Madison, WI.

Siegrist, R.L. 1987. Soil clogging during subsurface wastewater infiltration as affected by effluent composition and loading rate. J. Environ. Qual. 16(2):181-187.

Siegrist, R.L., D.L. Anderson, and J.C. Converse. 1984. Onsite Treatment and Disposal of Restaurant Wastewater. SSWMP. Univ. of Wisconsin, Madison, WI.

Siegrist, R.L., D.L. Anderson, and J.C. Converse. 1985. Commercial wastewater onsite treatment and disposal. p. 210-219. In On-site Wastewater Treatment. Proc. of the Fourth National Symp. on Individual and Small Community Sewage Systems. New Orleans, LA, Dec. 10-11,1984. ASAE, St. Joseph, MI.

Siegrist, R.L. , D.L. Anderson, D.L. Hargett, and R.J. Otis. 1985. Performance characteristics of a community wastewater absorption system. p. 144-145. In On-Site Wastewater Treatment. Proc. of the Fourth National Symp. on Individual and Small Community Sewage Systems. New Orleans, LA, Dec. 10-11, 1984. ASAE, St. Joseph, MI.

Siegrist, R.L., W.C. Boyle, and D.L. Anderson. 1981. A field evaluation of selected water conservation and wastewater reduction systems for residential application. Tech. rep. WIS WRC 81-02. Water Resources Cntr., Univ. of Wisconsin, Madison, WI.

Siegrist, R.L., M. Witt, and W.C. Boyle. 1976. Characteristics of rural household wastewater. J. Environ. Eng. Div. ASCE 102:533-548.

Siegrist, R.L., T. Woltanski, and L.E. Waldorf. 1978. Water conservation and wastewater disposal. p. 121-136. In Home Sewage Treatment. Proceedings Second National Home Sewage Treatment Symp. Chicago, IL, Dec. 12-13, 1977. ASAE, St. Joseph, MI.

Sikora, L.J. and R.B. Corey. 1976. Fate of nitrogen and phosphorus in soils under septic tank waste disposal fields. Trans. ASAE 19:866.

Simons, A.P. and F.R. Magdoff. 1979. Disposal of septic tank effluent in mound and sand filter - trench systems on a clay soil. J. Environ. Qual. 8:469-473.

Sobsey, M.D. 1983a. Transport and fate of viruses in soils. In Microbial Health Considerations of Soil Disposal of Domestic Wastewaters. Proc.of a Conf. Univ. of Oklahoma. May 11-12, 1982. U.S> EPA rep. no. EPA-600/9-83-017. U.S. EPA, Cincinnati, OH.

Sobsey, M.D. 1983b. Virology considerations in soil absorption of wastewater: Review of the literature and research needs. In R.C. Ward and S.M. Morrison (eds.) Proc. from a Workshop on Research Needs Relating to Soil Absorption of Wastewater. Colorado State Univ., Fort Collins, CO.

Sobsey, M.D., C.H. Dean, M.E. Knuckles, and R.A. Wagner. 1980. Interactions and survival of enteric viruses in soil materials. Applied Environ. Microbiology 40:92-101.

Starr, J.L. and B.L. Sawhney. 1980. Movement of nitrogen and carbon from a septic system drainfield. Water, Air, and Soil Pollution 13:113-123.

Stramer, S.L. and D.O. Cliver. 1981. Poliovirus removal from septic systems under conditions of soil saturation. Abstracts Annual Mtg. Amer. Society Microbiology. p.178.

Tchobanoglous, G. and F.L. Burton. 1991. Wastewater Engineering: Treatment, Disposal, Reuse, Metcalf & Eddy, 3rd ed. McGraw-Hill, Inc., New York, NY.

Thomas, R.E. et al. 1966. Soil chemical changes and infiltration rate reduction under sewage spreading. Soil Sci. Soc. Amer. Proc. 30:641-646.

Thurman, E.M., L.B. Barber, Jr., and D. Leblanc. 1986. Movement and fate of detergents in groundwater: A field study. J. of Contaminant Hydrology 1:143-161.

Tinker, Jr., J.R. 1991. An analysis of nitrate-nitrogen in ground water beneath unsewered subdivisions. Ground Water Monitoring Review 141-150.

Tofflemire, T.J. and M. Chen. 1977. Phosphate removal by sands and soil. Groundwater 15:377-387.

Tomson, M., C. Curran, J.M. King, H. Wang, J. Dauchy, V. Gordy, and B.H. Ward. 1984. Characterization of soil disposal system leachates. Rice Univ., Dept. of Environ. Sci. and Eng., Houston, TX. U.S. EPA rep. no. EPA-600/2-84-101. U.S. EPA, Municipal Environ. Research Lab., Cincinnati, OH.

Uebler, R.L. 1984. Effect of loading rate and soil amendments on inorganic nitrogen and phosphorus leached from a wastewater soil absorption system. J. Environ. Qual. 13:475-479.

U.S. Dept. of Commerce. 1973. 1970 Census of Housing Characteristics, Part 11, Florida.

U.S. Dept. of Commerce. 1983. 1980 Census of Housing, Volume 1, Chapter B. Detailed Housing Characteristics, Part 11, Florida.

U.S. Dept. of Commerce. 1993. 1990 Census of Population and Housing. Summary tape file 3A, 040, Florida.

U.S. EPA. 1980. Design manual: Onsite wastewater treatment and disposal systems. U.S. EPA rep. no. EPA-625/1-80-012. U.S. EPA, Cincinnati, OH.

U.S. EPA. 1984a. Drinking water microbiology - Committee on the challenges of modern society. U.S. EPA rep. no. EPA 570/9-84-006. U.S. EPA, Cincinnati, OH.

U.S. EPA. 1984b. Septage management and disposal. U.S. EPA rep. no. EPA-625/6-84-009. U.S. EPA, Cincinnati, OH.

U.S. EPA. 1987. Statistical abstract of the unsewered U.S. population. U.S. rep. no. EPA/600/5-87/004. Water Eng. Research Lab., Cincinnati, OH.

U.S. Public Health Service. 1957 (revised 1967). Manual of septic tank practice. Pub. No. 526, U.S. Dept. HEW, Pub. Health Serv., Washington, DC.

Univ. of Wisconsin. 1978. Management of small wastewater flows. U.S. EPA rep. no. EPA-600/7-78-173. MERL, ORD. U.S. EPA, Cincinnati, Ohio.

Vaughn, J.M. and E.F. Landry. 1977. Data rep.: An assessment of the occurrence of human viruses in Long Island aquatic systems. Dept. of Energy and Environ., Brookhaven National Lab., Upton, NY.

Vaughn, J.M. and E.F. Landry. 1980. The fate of human viruses in groundwater recharge systems. BNL 51214, UC-11. Dept. of Energy and Environ., Brookhaven National Lab., Upton, NY.

Vaughn, J.M., E.F. Landry and M.Z. Thomas. 1982. The lateral movement of indigenous enteroviruses in a sandy sole-source aquifer. In Microbial Health Considerations of Soil Disposal of Domestic Wastewaters. Proc. of a Conf. Univ. of Oklahoma, May 11-12, 1982. U.S. EPA rep. no. EPA-600/9-83-017. U.S. EPA, Cincinnati, OH.

Vaughn, J.M., E.F. Landry and M.Z. Thomas. 1983. Entrainment of viruses from septic tank leach fields through a shallow, sandy soil aquifer. Applied Environ. Microbiology 45:1474-1480.

Vaughn, J.M., E.F. Landry, C.A. Beckwith, and M.Z. Thomas. 1981. Applied Environ. Microbiology 41:139-147.

Vigon, B.W. and A.J. Rubin. 1989. Practical considerations in the surfactant-aided mobilization of contaminants in aquifers. J. WPCF 61(7):1233-1240.

Viraraghavan, T. and R.G. Warnock. 1976a. Efficiency of a septic tank tile system. J. WPCF 48:934-944.

Viraraghavan, T. and R.G. Warnock. 1976b. Groundwater pollution from a septic tile field. Water, Air, and Soil Pollution 5:281-287.

Viraraghavan, T. and R.G. Warnock. 1976c. Groundwater quality adjacent to a septic tank system. J. AWWA 68:611-614.

Vollenweider, R.A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing water, with particular reference to nitrogen and phosphorus as factors of eutrophication. Organization of Economic Cooperative Development, Director of Scientific Affairs. Tech. rep. DAS/CSI/68.27.

Voorhees, M.L. and J.M. Rice. 1987. Application of sensitivity and second order uncertainty analysis in formulating regional groundwater contamination risks and data sensitivities. In Solving Groundwater Problems with Models. Proc. of a National Water Well Assoc. Conf., Denver, CO, Feb. 10-12.

Walker, W.G. et al. 1973a. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: I. Soil transformations. J. Environ. Qual. 2:475.

Walker, W.G. et al. 1973b. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: II. Groundwater quality. J. Environ. Qual. 2:521-525.

Walton, G. 1951. Survey of literature relating to infant methemoglobinemia due to nitrate-contaminated water. Amer. J. Public Health. 41:986.

Watkins, R.E. 1991. Personal communication, March and August. Elkhart Co. Health Dept., Environ. Health Serv., Goshen, IN.

Watson, K.S., R. P. Farrell, and J.S. Anderson. 1967. The contribution from the individual home to the sewer system. J. WPCF. 39:2039-2054.

Weibel, S.R., C.P. Straub, and J.R. Thoman. 1949. Studies on household sewage disposal systems. Part I. Environ. Health Research Lab., Cincinnati, Ohio.

Wellings, F.M., A.L. Lewis, C.W. Mountain, and L.V. Pierce. 1975. Demonstration of virus in groundwater after effluent discharge onto soil. Applied Microbiology 29:751-757.

Westall, J. 1991. Personal communication, September. Oregon State Univ., Dept of Chem., Corvalis, OR.

Whelan, B.R. and N.J. Barrow. 1984. The movement of septic tank effluent through sandy soils near Perth. II: Movement of phosphorus. Australian J. Soil Research 22:293-302.

Whelan, B.R. and Z.V. Titamnis. 1982. Daily chemical variability of domestic septic tank effluent. Water, Air, and Soil Pollution 17:131-139.

Winneberger, J.H. et al. 1960. Biological aspects of failure of septic tank percolation systems. Final rep. San. Engr. Res. Lab., Univ. of California, Berkley, CA.

WPCF, 1990. Natural Systems for Wastewater Treatment. Manual of Practice FD-16. Water Pollution Control Federation, Alexandria, VA.

Wolterink, T.J. et al. 1979. Identifying sources of subsurface nitrate pollution with stable nitrogen isotopes. U.S. EPA rep. no EPA 600/4-79-050.

Wood, W.W. and R.L. Bassett. 1975. Water quality changes related to the development of anaerobic conditions during artificial recharge. Water Resources Research 11:553-558.

Yates, M.V. 1985. Septic tank density and ground water contamination. Ground Water 23:586-591.

Yates, M.V., C.P. Gerba and L.M. Kelley. 1985. Virus persistence in groundwater. Applied Environ. Microbiology 49:778-781.

Yates, M.V., S.R. Yates, A.W. Warrick, and C.P. Gerba. 1986. Use of geostatistics to predict virus decay rates for determination of septic tank setback distances. Applied Environ. Microbiology 52:479-483.

Yeager, J.G. and R.T. O'Brien. 1977. Enterovirus and bacteriophage inactivation in subsurface waters and translocation in soil. Water Resources Research Inst. rep. no. 083. New Mexico Water Resources Research Inst., New Mexico State Univ., Las Cruses, NM.

Young, R.H.F. and N.C. Burbank., Jr. 1973. Virus removal in Hawaiian soils. J. AWWA 65:698-604.

Ziebell, W.A., D.H. Nero, J.F. Deininger, and E. McCoy. 1975. Use of bacteria in assessing waste treatment and soil disposal systems. p. 58-63. In Home Sewage Disposal. Proc.of the National Home Sewage Disposal Symp. Chicago, IL, Dec. 19-20, 1974. ASAE, St. Joseph, MI.