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PROCESS DESIGN MANUAL
FOR
LAND TREATMENT OF MUNICIPAL WASTEWATER

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CONTRACTOR: Metcalf & Eddy, Inc., Palo Alto, California

Supervision: C. Pound, Vice President

Senior Author: R. Crites, Project Manager

Staff Authors: J. Esmay, D. Griffes, J. Olson, J. Powers, R. Shedden, R. Smith, A. Uiga

Consultant Authors: Dr. F. Broadbent, University of California, Davis
Dr. P. Pratt, University of California, Riverside
Dr. A. Wallace, University of Idaho
Dr. J. Melnick, Dr. C. Gerba, Dr. S. Goyal, Baylor University
Dr. R. Burau, University of California, Davis

INTERAGENCY WORKGROUP

Chairmen: B. Seabrook, OWPO, EPA
N. Urban, COE

Deputy Chairmen: R. Bastian, OWPO, EPA
S. Reed, CRREL, COE

Project Officer: Dr. J.E. Smith, Jr., ERIC, EPA

Members: W. Whittington, OWPO, EPA
R. Thomas, RSKERL, EPA
R. Dean, Region VIII, EPA
H. Thacker, OALWU, EPA
C. Rose, FHA, USDA
G. Loomis, USDA
R. Duesterhaus, USDA
Dr. J. Parr, ARS, USDA
Dr. P. Hunt, ARS, USDA
Dr. W. Larson, ARS, USDA
Col. T. Bishop, COE
D. Knudson, COE
Dr. H. McKim, CRREL, COE

EXPERTS AND INVITED REVIEWERS

Dr. H. Bouwer, Director, WCL, USDA/ARS
Dr. C. Lance, ARS, USDA
Dr. R. Loehr, Cornell University
Dr. W. Jewell, Cornell University
Dr. T. Hinesly, University of Illinois
Dr. M. Kirkham, Oklahoma State University
Dr. E. Lennette, M.D., California Health Department
E. Sepp, California Health Department
Lt. Col. V. Ciccone, USAMD
Capt. J. Glennon, USAMD
H. Pahren, HERL, EPA
Dr. C. Enfield, RSKERL, EPA
Dr. L. Carpenter, M.D., Former Director, Oklahoma State Health Department
R. Sletten; A. Palazzo; J. Bouzon; CRREL, COE
Dr. R. Lee, USAE, WES, COE
R. Madancy, OWRT, USDI
J. Dyer, OWRT, USDI

ABSTRACT

This manual presents a rational procedure for the design of land treatment systems. Slow rate, rapid infiltration, and overland flow processes for the treatment of municipal wastewaters are given emphasis. The basic unit operations and unit processes are discussed in detail, and the design concepts and criteria are presented.

The manual includes design examples as well as actual case study descriptions of operational systems. Information on planning and field investigations is presented along with the process design criteria for both large and small scale systems.

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FOREWORD

Land treatment is a reliable engineering process for wastewater management. Land application of wastewaters has been practiced in a number of modes, including crop and landscape irrigation; as a treatment process with collection and discharge of treated water; as indirect discharge to surface water; and as application to the soil surface for groundwater recharge. It is possible to modify any of these modes to meet project objectives, including the combination of several in a single management system.

The benefits of land treatment systems can go beyond the treatment of wastewater. Land treatment processes involve the recovery and beneficial reuse of wastewater nutrients and other elements through good agriculture, silviculture, and aquaculture practices. These practices permit the achievement of advanced levels of wastewater treatment as well as water reclamation and resource recovery objectives of recent environmental legislation. The production of revenues through the sale of byproducts (e.g., crops) can be realized. Land treatment systems can aid in the reclamation and reuse of water resources, recharge of groundwater aquifers, reclamation of marginal land, and the preservation of open spaces for future greenbelts.

It is the purpose of this manual to describe the basic principles of land treatment and to present a rational procedure for design of land treatment systems. Information contained in this manual can be used in identifying alternatives during planning, in selecting a process alternative or site, in determining necessary field investigations, and in conducting the process design.

This manual is unique in the Technology Transfer Process Design Manual series because its preparation was jointly sponsored by the U. S. Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers. The U.S. Department of Agriculture (USDA) provided technical assistance during the review process. In recognition of these contributions, the cover and title page were designed to clearly indicate the endorsement of these three agencies. The review process included over thirty individuals contributing significantly to the preparation of this manual. They provided a broad range of technical expertise and represented a wide variety of agencies and institutions. This extensive review ensured the accuracy and the authority of the product.

The manual represents the current state-of-the-art with respect to criteria, data, and procedures for the design of land treatment processes for municipal wastewaters. Much of the information is also applicable for design of systems managing industrial wastewaters. Revisions and improvements will be made as results of current and future research and development become available.

CHAPTER 1

INTRODUCTION

1.1 Background and History

Land treatment of municipal wastewater involves the use of plants, the soil surface, and the soil matrix to remove many wastewater constituents. A wide variety of processes can be used to achieve many different objectives of treatment, water reuse, nutrient recycling, and crop production.

The concept of land application of wastewater certainly is not new to the field of sanitary engineering. Evidence of such systems in western civilization extends back as far as ancient Athens [1]. A wastewater irrigation system in Bunzlaw, Germany, is reported to have been in operation for over 300 years beginning in 1559 [2].

The greatest proliferation of land treatment systems occurred in Europe in the second half of the nineteenth century. Pollution of many rivers had reached unacceptable levels, and disposal of sewage on the land was the only feasible means of treatment available at the time. "Sewage farming," the practice of transporting sewage into rural areas for irrigation and disposal, was commonly used by many European cities, including some of those shown in Table 1-1. In the 1870s, the practice was recognized in England as treatment, with many underdrained systems exhibiting sparkling clear effluents [3]. As urban areas expanded and in-plant treatment processes became available, many of these older systems were abandoned because of land development pressures.

Early experiences in the United States also date back to the 1870s [4]. As in Europe, sewage farming became relatively common as a first attempt to control water pollution. In the first half of the twentieth century, these early systems were generally replaced either by in-plant treatment or by (1) managed farms where treated wastewater was used for crop production, (2) landscape irrigation sites, or (3) groundwater recharge sites [1]. These newer land treatment systems tended to predominate in the West where the resource value of wastewater was an added advantage. In addition, experience with land application of food processing and pulp and paper industrial wastewaters has been drawn upon in developing the technology of land treatment [1, 5].

The increasing use of land treatment over the last 40 years is shown in Table 1-2, which was compiled from periodic inventories of municipal wastewater treatment facilities [2]. While it is evident that the number of systems has steadily grown, it still represents only a small percentage of the estimated 15 000 total municipal treatment facilities [2].

TABLE 1-1

SELECTED EARLY LAND APPLICATION SYSTEMS [1, 2, 5, 6, 7, 8]

Location	Date started	Type of system	Area, acres	Flow, Mgal/d
International				
Croydon-Beddington, England	1860	Sewage farm	630	5.6
Paris, France	1869	SR[a]	16 000	79
Leamington, England	1870	Sewage farm	400	1
Berlin, Germany	1874	Sewage farm	68 000	.....
Wroclaw, Poland	1882	Sewage farm	2 000	28
Melbourne, Australia	1893	SR OF[b]	10 400 3 500	50 70
Braunschweig, Germany	1896	Sewage farm	11 000	16
Mexico City, Mexico	1900	SR	112 000	570
United States				
Calumet City, Michigan	1888	RI[c]	12	1.2
Woodland, California	1889	SR	240	4.2
Fresno, California	1891	SR	4 000	26
San Antonio, Texas	1895	SR	4 000	20
Vineland, New Jersey	1901	RI	14	0.8
Ely, Nevada	1908	SR	1 400	1.5

a. SR = slow rate.
b. OF = overland flow.
c. RI = rapid infiltration.

1 acre = 0.405 ha
1 Mgal/d = 43.8 L/s

TABLE 1-2

U.S. MUNICIPALITIES USING LAND TREATMENT [2]

Year	No. of systems	Population served, millions
1940	304	0.9
1945	422	1.3
1957	461	2.0
1962	401	2.7
1968	512	4.2
1972	571	6.6

In recent years, much effort has been spent on developing land treatment technology and improving methods of control. The various types of land treatment systems have become accepted as viable wastewater management techniques that should be considered equally with any others. The regulations developed pursuant to the Federal Water Pollution Control Act Amendments of 1972 (Public Law 92-500) require that such consideration be given for federally funded municipal wastewater projects. In the Act, the Environmental Protection Agency administrator is directed to encourage waste treatment management that results in facilities for (1) the recycling of potential pollutants through the production of agricultural, silvicultural, and aquacultural products; (2) the reclamation of wastewater; and (3) the elimination of the discharge of pollutants.

1.2 Objectives

The primary objective of this manual is to provide a comprehensive source of information to be used in the planning and design of land treatment systems. It is not intended to serve as a definition of policy on land treatment, but rather to set forth and extend the present state-of-the-art technology. Recommended procedures, case studies, and several examples are presented which are intended to serve as planning and design aids.

Throughout the manual, emphasis is given to the wide range of design possibilities available for land treatment systems. The user is encouraged to adapt the techniques and procedures described to suit local needs and conditions.

1.3 Scope of the Manual

Planning and technical information for each of the following major wastewater treatment processes involving land application are presented:

- Slow rate (SR), also referred to as crop irrigation
- Rapid infiltration (RI)
- Overland flow (OF)

Other types of systems, such as wetland and subsurface systems, which are uncommon or new, are also described but in less detail. Systems specifically involving the land application of sludge, injection wells, sealed evaporation ponds, and conventional septic tank leach fields are not covered.

The scope of most of the information in the manual is directed to medium-to-large systems. For small systems, say 0.1 million gallons per day (Mgal/d) [4.4 L/s], or less, many of the design procedures presented in the manual must be realistically simplified. Special considerations for small systems are discussed separately in Chapter 6.

To minimize the amount of theoretical and background information in the manual, papers on six topics of special interest are included as appendixes. These papers were written by recognized experts in their particular fields and cover the following topics: (1) nitrogen, (2) phosphorus, (3) hydraulic capacity, (4) pathogens, (5) metals, and (6) field investigation procedures. These appendixes form the technical foundation for the body of the report. Research results reported in this manual are current through 1976. Detailed procedures for determining capital and operation and maintenance costs are not included in this manual. Sources for such information are given in Chapter 3 along with general summaries.

1.4 Guide to Intended Use

The contents of the manual should be helpful to a variety of different users, including those seeking to gain a general perspective on land treatment and those looking for specific design information. Consequently, the manual is organized to allow the user to locate particular information and to concentrate on specific areas of interest as easily as possible. Subject, location, and author indexes are provided to allow easy access to specific information. A glossary is also provided to give definitions of terms germane to land treatment which might not be familiar to the traditional civil/sanitary engineer. The following brief chapter descriptions are provided as an introduction to the organization of the manual.

<u>Chapter 2</u> - Treatment Process Capabilities and Objectives.
The basic concepts of each process of land treatment are described. Standard terminology and ranges of important design criteria that are encountered throughout the rest of the manual are presented.

<u>Chapter 3</u> - Technical Planning and Feasibility Assessment.
Information for those users involved in both regional and facilities planning efforts is provided. Most of the technical information and guidance contained in the manual is presented here and in Chapter 5. Procedures are described for investigating sites and for developing and evaluating land treatment alternatives. Wherever possible, desirable ranges of criteria associated with physical characteristics are given.

<u>Chapter 4</u> - Field Investigations.
Field investigations are outlined for each land treatment process. Reasons for field tests are given along with guidance on possible interpretation of test results.

Chapter 5 - Process Design.
Design guidelines are presented for projects in which the site and process have been determined. In the first part of the chapter, each of the major treatment processes is discussed separately with respect to application rates and removals of various wastewater constituents. Subsequent sections are devoted to design components of land treatment systems. These include preapplication treatment, storage, distribution, and management of renovated water. Discussions are then provided on vegetation and agricultural management, system monitoring, and facilities design guidance.

Chapter 6 - Small Systems.
Simplified designs that are possible for small community systems are described. Shortcuts for the planning and design procedures described in Chapters 3 and 5 are given along with special considerations. A design example is also included.

Chapter 7 - Case Studies.
Brief descriptions of the design criteria and operational characteristics of 11 successful land treatment systems are presented. The systems were chosen to represent as broad a cross-section as possible with respect to type of system, size, and location.

Chapter 8 - Design Example.
An example that illustrates the principles described in Chapters 3 and 5 is presented. For a flow of 10 Mgal/d (0.44 m^3/s), in a humid eastern climate, alternatives are developed and compared for slow rate and a combination of the overland flow and rapid infiltration processes.

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CHAPTER 2

TREATMENT PROCESS CAPABILITIES AND OBJECTIVES

2.1 Introduction

Land treatment of municipal wastewater encompasses a wide variety of processes or methods. The three principal processes, as referred to in this manual, are:

1. Slow rate

2. Rapid infiltration

3. Overland flow

Other processes, which are less widely used and generally less adaptable to large-scale use than the three principal ones, include:

1. Wetlands

2. Subsurface

The major concepts involved in these processes are introduced in this chapter. Descriptions are given of system objectives and treatment mechanisms.

Typical design features for the various land treatment processes are compared in Table 2-1, with more detail provided in Chapter 5. The major site characteristics are compared for each process in Table 2-2, with more detail provided in Chapter 3. The expected quality of treated water from the three principal land treatment processes is shown in Table 2-3. The major removal mechanisms responsible for the quality improvement are described for each land treatment process in the following sections.

2.2 Slow Rate Process

In several previous EPA reports, including <u>Evaluation of Land Application Systems</u> [1], slow rate land treatment was referred to as irrigation. The term slow rate land treatment is used to focus attention on wastewater treatment rather than on irrigation of crops. However, in slow rate systems, vegetation is a critical component for managing water and nutrients.

TABLE 2-1

COMPARISON OF DESIGN FEATURES FOR LAND TREATMENT PROCESSES

Feature	Principal processes			Other processes	
	Slow rate	Rapid infiltration	Overland flow	Wetlands	Subsurface
Application techniques	Sprinkler or surface[a]	Usually surface	Sprinkler or surface	Sprinkler or surface	Subsurface piping
Annual application rate, ft	2 to 20	20 to 560	10 to 70	4 to 100	8 to 87
Field area required, acres[b]	56 to 560	2 to 56	16 to 110	11 to 280	13 to 140
Typical weekly application rate, in.	0.5 to 4	4 to 120	2.5 to 6[c] 6 to 16[d]	1 to 25	2 to 20
Minimum preapplication treatment provided in United States	Primary sedimentation[e]	Primary sedimentation	Screening and grit removal	Primary sedimentation	Primary sedimentation
Disposition of applied wastewater	Evapotranspiration and percolation	Mainly percolation	Surface runoff and evapotranspiration with some percolation	Evapotranspiration, percolation, and runoff	Percolation with some evapotranspiration
Need for vegetation	Required	Optional	Required	Required	Optional

a. Includes ridge-and-furrow and border strip.

b. Field area in acres not including buffer area, roads, or ditches for 1 Mgal/d (43.8 L/s) flow.

c. Range for application of screened wastewater.

d. Range for application of lagoon and secondary effluent.

e. Depends on the use of the effluent and the type of crop.

1 in. = 2.54 cm
1 ft = 0.305 m
1 acre = 0.405 ha

TABLE 2-2

COMPARISON OF SITE CHARACTERISTICS FOR LAND TREATMENT PROCESSES

	Principal processes			Other processes	
Characteristics	Slow rate	Rapid infiltration	Overland flow	Wetlands	Subsurface
Slope	Less than 20% on cultivated land; less than 40% on noncultivated land	Not critical; excessive slopes require much earthwork	Finish slopes 2 to 8%	Usually less than 5%	Not critical
Soil permeability	Moderately slow to moderately rapid	Rapid (sands, loamy sands)	Slow (clays, silts, and soils with impermeable barriers)	Slow to moderate	Slow to rapid
Depth to groundwater	2 to 3 ft (minimum)	10 ft (lesser depths are acceptable where underdrainage is provided)	Not critical	Not critical	Not critical
Climatic restrictions	Storage often needed for cold weather and precipitation	None (possibly modify operation in cold weather)	Storage often needed for cold weather	Storage may be needed for cold weather	None

1 ft = 0.305 m

TABLE 2-3

EXPECTED QUALITY OF TREATED WATER FROM LAND TREATMENT PROCESSES
mg/L

Constituent	Slow rate[a]		Rapid infiltration[b]		Overland flow[c]	
	Average	Maximum	Average	Maximum	Average	Maximum
BOD	<2	<5	2	<5	10	<15
Suspended solids	<1	<5	2	<5	10	<20
Ammonia nitrogen as N	<0.5	<2	0.5	<2	0.8	<2
Total nitrogen as N	3	<8	10	<20	3	<5
Total phosphorus as P	<0.1	<0.3	1	<5	4	<6

a. Percolation of primary or secondary effluent through 5 ft (1.5 m) of soil.

b. Percolation of primary or secondary effluent through 15 ft (4.5 m) of soil.

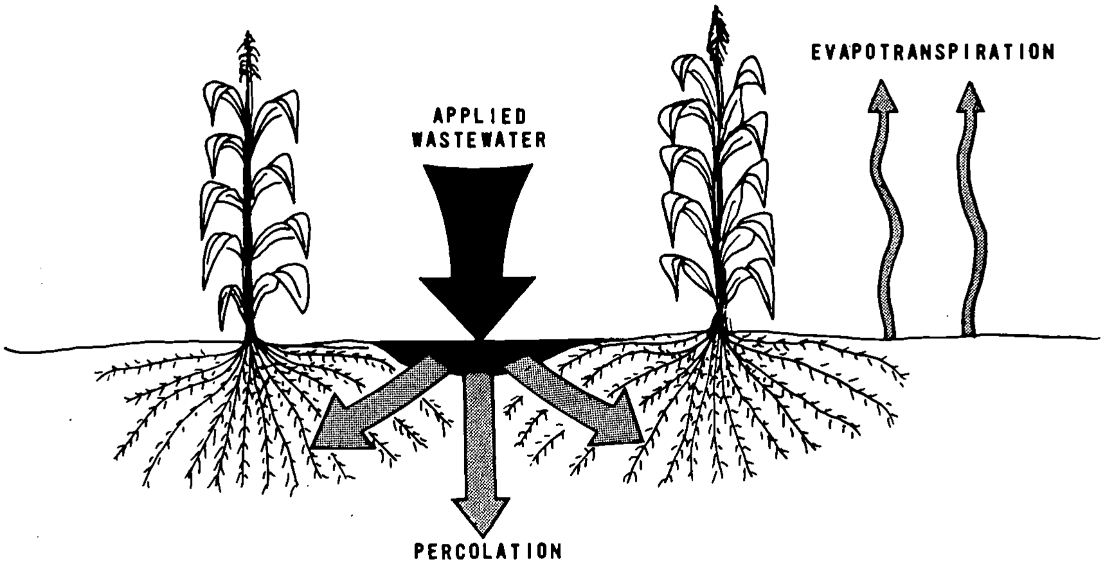
c. Runoff of comminuted municipal wastewater over about 150 ft (45 m) of slope.

The applied wastewater is treated as it flows through the soil matrix, and a portion of the flow percolates to the groundwater. Surface runoff of the applied water is generally not allowed. A schematic view of the typical hydraulic pathway for slow rate treatment is shown in Figure 2-1(a). Typical views of slow rate land treatment systems, using both surface and sprinkler application techniques, are also shown in Figure 2-1(b, c). Surface application includes ridge-and-furrow and border strip flooding techniques. The term sprinkler application is correctly applied to impact sprinklers and the term spray application should only be used to refer to fixed spray heads.

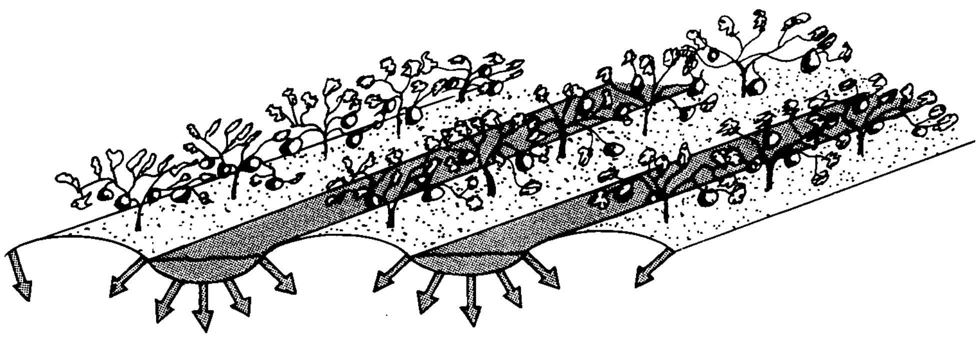
The case studies in Chapter 7 include six slow rate systems that are fairly representative of those found throughout the United States: Pleasanton, California; Walla Walla, Washington; Bakersfield, California; San Angelo, Texas; Muskegon, Michigan; and St. Charles, Maryland. These case studies provide an insight into actual experiences with slow rate systems.

FIGURE 2-1

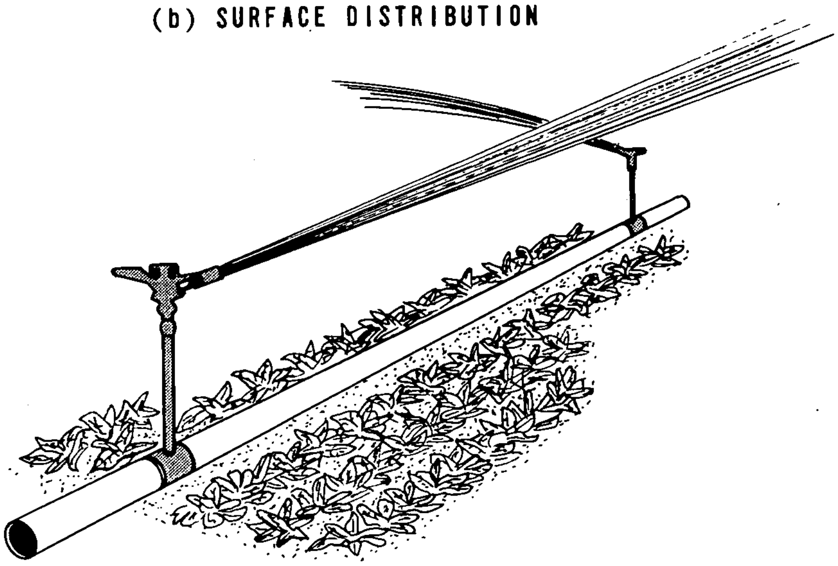
SLOW RATE LAND TREATMENT



(a) HYDRAULIC PATHWAY



(b) SURFACE DISTRIBUTION



(c) SPRINKLER DISTRIBUTION

2.2.1 System Objectives

Slow rate systems can be operated to achieve a number of objectives including:

1. Treatment of applied wastewater

2. Economic return from use of water and nutrients to produce marketable crops (irrigation)

3. Water conservation, by replacing potable water with treated effluent, for irrigating landscaped areas, such as golf courses

4. Preservation and enlargement of greenbelts and open space

When requirements for surface discharge are very stringent for nitrogen, phosphorus, and biochemical oxygen demand (BOD), they can be met with slow rate land treatment. If the percolating water must meet EPA drinking water standards, reduction in nitrogen below the 10 mg/L standard for nitrate-nitrogen is often the limiting criterion. In arid regions, however, increases in chlorides and total dissolved salts in the groundwater may be limiting. Management approaches to meet the above objectives within the slow rate process are discussed under the topics (1) wastewater treatment, (2) crop irrigation, (3) turf irrigation, and (4) silviculture.

2.2.1.1 Wastewater Treatment

When the primary objective of the slow rate process is treatment, the hydraulic loading is limited either by the infilration capacity of the soil or the nitrogen removal capacity of the soil-vegetation complex. If the hydraulic capacity of the site is limited by a relatively impermeable subsurface layer or by a high groundwater table, underdrains can be installed to increase the allowable loading. Grasses are usually chosen for the vegetation because of their high nitrogen uptake capacities.

2.2.1.2 Crop Irrigation

When the crop yields and economic returns from slow rate systems are emphasized, crops of higher values than grasses are usually selected. In the West, application rates are generally between 1 and 3 in./wk (2.5 to 7.6 cm/wk), which reflect the consumptive use of crops. Consumptive use rates are those required to replace the water lost to evaporation, plant transpiration, and stored in plant tissue. In areas where water does not limit plant growth, the nitrogen and phosphorus in wastewater can be recycled in crops. These nutrients can increase yields of corn, grain sorghum, and similar crops and provide an economic return.

2.2.1.3 Turf Irrigation

Golf courses, parks, and other turfed areas can be irrigated with wastewater, thus, conserving potable water supplies. These areas generally have considerable public access and this usually requires that a disinfected effluent be applied.

2.2.1.4. Silviculture

Silviculture, the growing of trees, is being conducted with wastewater effluent in at least 11 existing sites in Oregon, Michigan, Maryland, and Florida [2]. In addition, experimental systems at Pennsylvania State University [3], Michigan State University [4], and the University of Washington [5] are being studied extensively to determine permissible loading rates, responses of various tree species, and environmental effects.

Forests offer several advantages as potential sites for land treatment:

1. Large forested areas exist near many sources of wastewater.

2. Forest soils often exhibit better infiltration properties than agricultural soils.

3. Site acquisition costs for forestland are usually lower than site acquisition costs for agricultural land because of lower land values for forestlands.

4. During cold weather, soil temperatures are often higher in forestlands than in comparable agricultural lands.

The principal limitations on the use of wastewater for silviculture are that:

1. Water tolerances of the existing trees may be low.

2. Nitrogen removals are relatively low.

3. Fixed sprinklers, which are expensive, must generally be used.

Existing forests are adapted to the water supply from natural precipitation. Unless soils are well drained, the increase in hydraulic loading from wastewater application will drown existing trees. At Seabrook Farms, New Jersey, the types of vegetation have changed from predominantly oak trees to wild berries, marsh grass, and other grasses [6].

2.2.2 Treatment Performance

Slow rate treatment is generally capable of producing the best results of all the land treatment systems. The quality values shown in Table 2-3 can be expected for most well-designed and well-operated slow rate systems.

Organics are reduced substantially by slow rate land treatment by biological oxidation within the top few inches of soil. At Muskegon, Michigan, the BOD of renovated water from the drain tiles has ranged from 1.2 to 2.2 mg/L, and the BOD of renovated water intercepted by two nearby creeks has ranged from 2 to 3.3 mg/L [7]. Preliminary results for six test cells at a research project in Hanover, New Hampshire, show average annual BOD concentrations in the percolate ranging from 0.6 to 2.1 mg/L [8]. These results were consistently achieved with application rates ranging up to 6 in./wk (15 cm/wk) with both primary and secondary effluents applied. Filtration and adsorption are the initial mechanisms in BOD removal, but biological oxidation is the ultimate treatment mechanism.

Suspended solids removals are not as well documented as BOD removals, but concentrations of 1 mg/L or less can generally be expected in the renovated water. Filtration is the major removal mechanism for suspended solids. Volatile solids are biologically oxidized, and fixed or mineral solids become part of the soil matrix.

Nitrogen is removed primarily by crop uptake, which varies with the type of crop grown and the crop yield. To remove the nitrogen effectively, the portion of the crop that contains the nitrogen must be physically removed from the field. Denitrification can also be significant, even if the soil is in an aerobic condition most of the time. In a laboratory study using radioactive tracer materials, Broadbent reported denitrification losses of up to 32% of the applied nitrogen [9]. In the test cells at Hanover, denitrification losses were found to be 5 to 28% [8]. In both of these cases, the soils were considered to be essentially aerobic.

Phosphorus is removed from solution by fixation processes in the soil, such as adsorption and chemical precipitation. Removal efficiencies are generally very high for slow rate systems and are usually more dependent on the soil properties than on the concentration of the phosphorus applied. A small but significant portion of the phosphorus applied (15 to 30% depending on the soil and the crop) is taken up and removed with the crop.

2.3 Rapid Infiltration

In rapid infiltration land treatment (referred to in previous EPA reports as infiltration-percolation), most of the applied wastewater percolates through the soil, and the treated effluent eventually reaches the groundwater. The wastewater is applied to rapidly permeable soils, such as sands and loamy sands, by spreading in basins or by sprinkling, and is treated as it travels through the soil matrix. Vegetation is not usually used, but there are some exceptions.

The schematic view in Figure 2-2(a) shows the typical hydraulic pathway for rapid infiltration. A much greater portion of the applied wastewater percolates to the groundwater than with slow rate land treatment. There is little or no consumptive use by plants and less evaporation in proportion to a reduced surface area.

In many cases, recovery of renovated water is an integral part of the system. This can be accomplished using underdrains or wells, as shown in Figure 2-2(b, c).

Among the case studies in Chapter 7 are three that serve as representative examples of rapid infiltration systems: Phoenix, Arizona; Lake George, New York; and Fort Devens, Massachusetts.

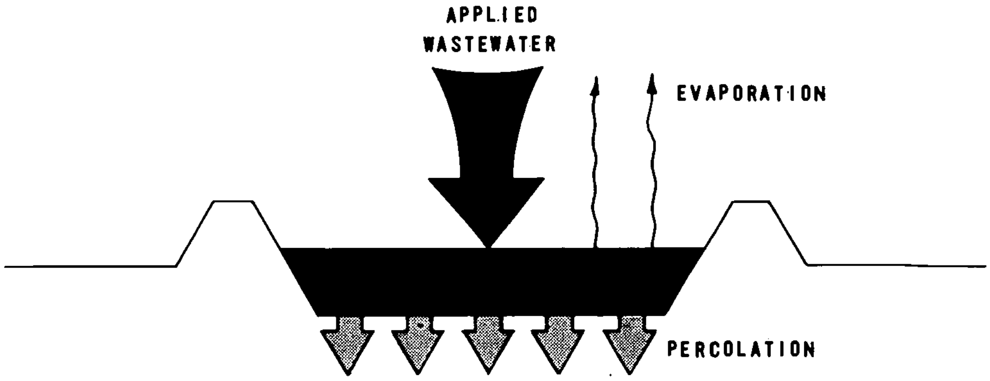
2.3.1 System Objectives

The principal objective of rapid infiltration is wastewater treatment. Objectives for the treated water can include:

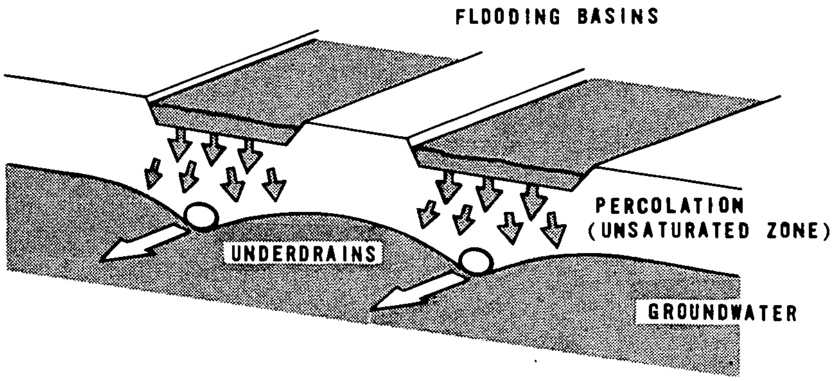
1. Groundwater recharge
2. Recovery of renovated water by wells or underdrains with subsequent reuse or discharge
3. Recharge of surface streams by interception of groundwater
4. Temporary storage of renovated water in the aquifer

FIGURE 2-2

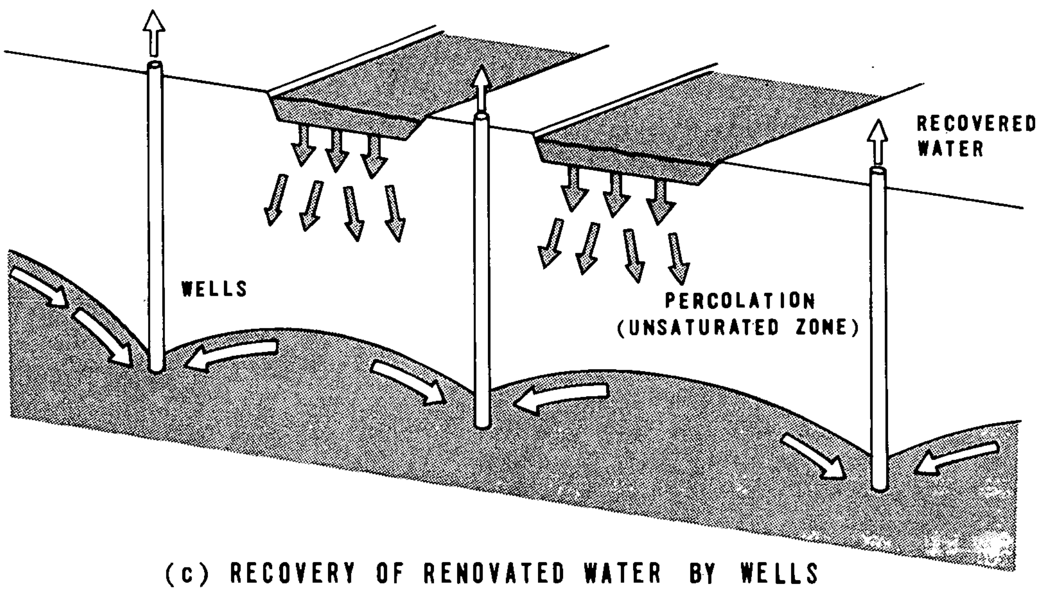
RAPID INFILTRATION



(a) HYDRAULIC PATHWAY



(b) RECOVERY OF RENOVATED WATER BY UNDERDRAINS



(c) RECOVERY OF RENOVATED WATER BY WELLS

If groundwater quality is being degraded by salinity intrusion, groundwater recharge by rapid infiltration can help to reverse the hydraulic gradient and protect the existing groundwater.

Return of the renovated water to the surface by wells, underdrains, or groundwater interception may be necessary or advantageous when discharge to a particular surface water body is dictated by senior water rights, or when existing groundwater quality is not compatible with expected renovated water quality. At Phoenix, for example, treated water is withdrawn immediately by wells to prevent spreading into the groundwater and to allow reuse of the water for irrigation [10].

2.3.2 Treatment Performance

Removals of wastewater constituents by the filtering and straining action of the soil are excellent. Suspended solids, BOD, and fecal coliforms are almost completely removed in most cases [10, 11] .

Nitrogen removals are generally poor unless specific operating procedures are established to maximize denitrification. At Flushing Meadows, total nitrogen removals of 30% were obtained consistently. In laboratory studies it was shown, however, that increased denitrification could have been obtained by: (1) adjusting application cycles, (2) supplying an additional carbon source, (3) using vegetated basins, (4) recycling the portions of the renovated water containing high nitrate concentrations, and (5) reducing application rates [12]. Applying some of these practices in the field increased nitrogen removal, resulting from denitrification, to about 50%. Although total nitrogen removals may be poor, rapid infiltration is an excellent method for achieving a nitrified effluent.

Phosphorus removals can range from 70 to 99%, depending on the physical and chemical characteristics of the soil. As with slow rate systems, the primary removal mechanism is adsorption with some chemical precipitation, so the long-term capacity is limited by the mass of soil in contact with the wastewater. Removals are related also to the residence time of the wastewater in the soil and the travel distance (see Section 5.1.3).

2.4 Overland Flow

In overland flow land treatment, wastewater is applied over the upper reaches of sloped terraces and allowed to flow across the vegetated surface to runoff collection ditches. The wastewater is renovated by physical, chemical, and biological means as it flows in a thin film down the relatively impermeable slope. A schematic view of overland flow treatment is shown in Figure 2-3(a), and a pictorial view of a typical system is shown in Figure 2-3(b). As shown in Figure 2-3(a), there is relatively little percolation involved either because of an impermeable soil or a subsurface barrier to percolation.

Overland flow is a relatively new treatment process for municipal wastewater in the United States. As of August 1976, only three relatively small, full-scale municipal systems have been constructed. These are located in Oklahoma, Mississippi, and South Carolina. The earliest of these systems, at Pauls Valley, Oklahoma, is described as a case study in Chapter 7. In Melbourne, Australia, overland flow has been used to treat settled wastewater for several decades [13, 14]. The Campbell Soup Company treatment plant at Paris, Texas, which is perhaps the best known of approximately 10 industrial systems in the county, is also described as a case study in Chapter 7. Besides these full-scale examples, extensive reference is made throughout this manual to the pilot scale municipal studies sponsored by the EPA at Ada, Oklahoma, and the bench-scale greenhouse studies sponsored by the Corps of Engineers at Vicksburg, Mississippi.

2.4.1 System Objectives

The objectives of overland flow are wastewater treatment and, to a minor extent, crop production. Treatment objectives may be either (1) to achieve secondary or better effluent quality from screened primary treated, or lagoon treated wastewater, or (2) to achieve high levels of nitrogen and BOD removals comparable to conventional advanced wastewater treatment from secondary treated wastewater. Treated water is collected at the toe of the overland flow slopes and can be either reused or discharged to surface water. Overland flow can also be used for production of forage grasses and the preservation of greenbelts and open space.

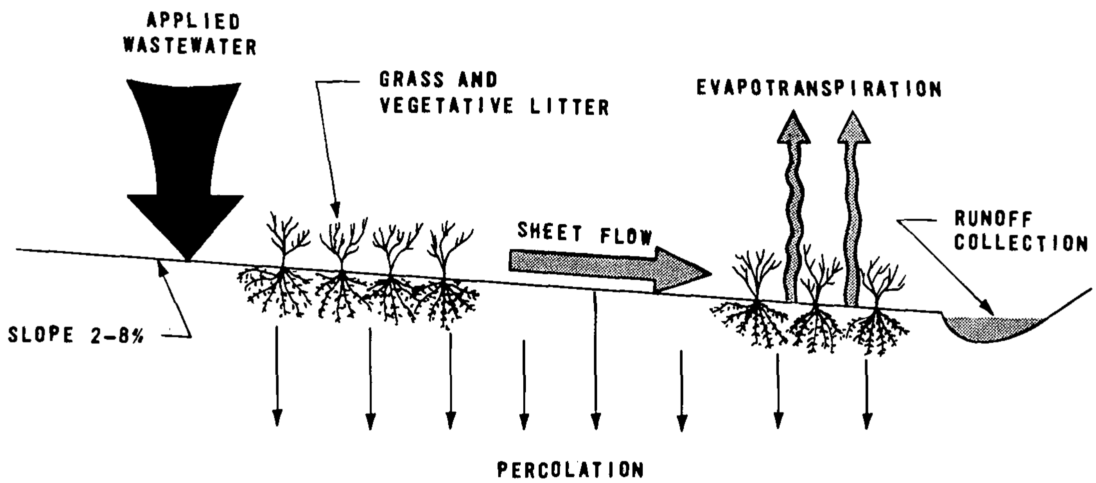
2.4.2 Treatment Performance

Biological oxidation, sedimentation, and grass filtration are the primary removal mechanisms for organics and suspended solids. At Ada, using raw comminuted wastewater, Thomas reported total suspended solids concentrations of 6 to 8 mg/L in the runoff during the summer and 8 to 12 mg/L in the winter [15]. BOD concentrations during the same period were 7 to 11 mg/L in the summer and 8 to 12 mg/L in the winter. An acclimation or seasoning period of about 3 months was required before optimum removals were achieved.

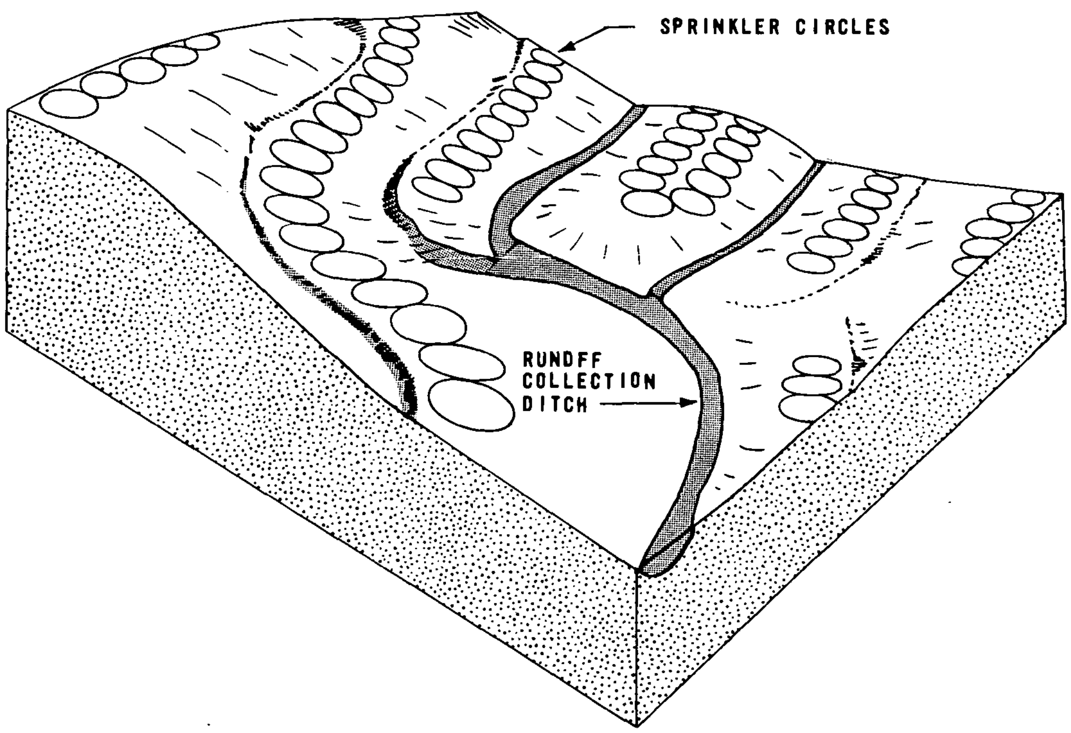
Nitrogen removal is attributed primarily to denitrification. Hunt has reasoned that an aerobic-anaerobic double layer exists at the surface of the soil and allows both nitrification and denitrification to occur [16, 17]. Because this process depends on two stages of microbial activity, it is sensitive to environmental conditions. Plant uptake of nitrogen

FIGURE 2-3

OVERLAND FLOW



(a) HYDRAULIC PATHWAY



(b) PICTORIAL VIEW OF SPRINKLER APPLICATION can also be a significant removal mechanism. Permanent nitrogen removal by plant uptake is only possible if the crop is harvested and removed from the field. Ammonia volatilization can be significant if the pH of the water is above 7. Nitrogen removals usually range from 75 to 90% with runoff nitrogen being mostly in the nitrate form. Higher levels of nitrate and ammonium may occur during cold weather as a result of reduced biological activity and limited plant uptake.

Phosphorus is removed by adsorption and precipitation in essentially the same manner as with the slow rate and rapid infiltration methods. Treatment efficiencies are somewhat limited because of the incomplete contact between the wastewater and the adsorption sites within the soil. Phosphorus removals usually range from 30 to 60% on a concentration basis. Increased removals may be obtained by adding alum or ferric chloride prior to application (see Section 5.1.4).

2.5 Other Processes

The three principal land treatment processes, when implemented, represent planned and engineered changes to the existing environment. Recently, the concept of using natural ecosystems, such as wetlands, for wastewater treatment has received considerable attention. Applications of wastewater (1) to wetlands for treatment, and (2) to the soil by subsurface techniques are described in this section.

2.5.1 Wetlands

Wetlands, which constitute 3% of the land area of the continental United States [18], are intermediate areas in a hydrological sense: they have too many plants and too little water to be called lakes, yet they have enough water to prevent most agricultural or silvicultural uses. The term wetlands is used in this manual to encompass areas also known as marshes, bogs, wet meadows, peatlands, and swamps. The ability of wetlands to influence water quality is the reason for much current research on their use for wastewater management.

Three categories of wetlands are currently used for municipal wastewater treatment:

- Artificial wetlands
- Existing wetlands
- Peatlands

Wetlands are discussed in more detail in Section 5.1.5. Peatlands are discussed separately because these highly organic soils can be drained and managed in a manner similar to that used in slow rate land treatment.

2.5.1.1 Artificial Wetlands

Two artificial wetlands treatment systems have been developed at the Brookhaven National Laboratory on Long Island, New York [19]. Both are wetlands-pond systems. In the first, the wetlands consist of wet meadows merging into a marsh followed by a pond (meadow-marsh). In the second system, the wet meadows are deleted. Both systems are being loaded at an application rate of about 25 in./wk (63 cm/wk). Aerated wastewater is applied and recycling is no longer employed.

These artificial wetlands were formed in sandy soil by installing an impervious plastic liner under the soil. They were placed in operation in June 1973. Operating modes have evolved from the original recycling to the present once-through approach with increasing loading rates until April 1976, when the present rates were established. Typical averaged results for July through September 1975 for operation with a one-to-one recycling of pond effluent are presented in Table 2-4.

TABLE 2-4

TREATMENT PERFORMANCE FOR TWO ARTIFICIAL WETLAND SYSTEMS ON LONG ISLAND [19] mg/L

Constituent	Influent	Meadow-marsh effluent	Wetlands effluent
BOD	520	15	16
Suspended solids	860	43[a]	57[a]
Total nitrogen	36	3	4
Fecal coliforms, count/100 mL	3,000	17[b]	21[b]

a. Principally algae.

b. Geometric mean.

The wetlands area occupies 0.2 acre (0.08 ha) and is flooded to a depth of about 0.5 ft (0.15 m). Small recommends a 1 ft (0.3 m) depth or more to prevent volunteer weed growth and to prevent washout during storms [20]. Cattails were planted and duckweed (Lemna minor) is prevalent. Regular harvest of cattails is not practiced but weeds, grasses, and cattails were thinned out in March 1976.

2.5.1.2 Existing Wetlands

The application of secondary effluent to existing freshwater and salt water wetlands is being studied in Mississippi, as well as in California, Michigan, Louisiana, Florida, and Wisconsin. In Mississippi, Wolverton has studied the use of water hyacinths in secondary wastewater lagoons to effect removals of BOD, suspended solids, and nutrients [21]. A surface area of 0.7 acre (0.28 ha) was used, and detention times ranged from 14 to 21 days. The treatment performance of this system is compared to that of a control lagoon free of water hyacinths for September 1975 as shown in Table 2-5.

TABLE 2-5

TREATMENT CAPABILITY OF WATER HYACINTHS FED OXIDATION POND EFFLUENT [21]
mg/L

	Hyacinth pond		Control pond	
Constituent	Influent	Effluent	Influent	Effluent
BOD	22	7	27	30
Suspended solids	43	6	42	46
Total Kjeldahl nitrogen	4.4	1.1	4.5	4.5
Total phosphorus	5.0	3.8	4.8	4.6
TDS	187	183	390	380

Hyacinths must be harvested for effective nutrient removal. Wolverton suggests harvesting every 5 weeks during the warm growing season. The harvested plants may be processed into high-protein feed products, organic fertilizer and soil conditioner, or methane gas [22].

The use of existing wetlands appears to hold promise as an emerging technology for wastewater management. Management techniques for nutrient removal, loading rates, climatic constraints and suitable site characteristics need further study.

2.5.1.3 Peatlands

The use of peatlands or organic soils for land application has been studied by Farnham in Minnesota [23] and by Kadlec in Michigan [24]. A system has been designed by Stanlick [25] on the basis of Farnham's research.

Although sprinkler or surface application techniques can be used on peatlands, the North Star Campground system in Minnesota uses sprinklers [25]. It was designed for 13.3 in./wk (33.8 cm/wk) and is underdrained at a depth of about 3 ft (1 m). Treatment efficiency for 1975 is summarized in Table 2-6. Secondary effluent was applied.

TABLE 2-6

TREATMENT PERFORMANCE OF PEATLAND IN MINNESOTA [25]
mg/L

Constituent	Influent	Effluent
BOD	....	5
Suspended solids	....	5
Total nitrogen	20-40	1-10
Total phosphorus	10	0.1
Fecal coliforms, count/100 mL	10^3-10^5	0-4

Because of the high loading rate, the nitrogen uptake of the grass planted on the peat surface was surpassed. Although the peat pH was 4, the effluent pH was consistently between 6.5 and 7.5.

2.5.2 Subsurface Application

Two systems that are quite similar to the peatland system are the soil mound and the subsurface filter systems. The subsurface filter is described in the Manual of Septic-Tank Practice [26]. The soil mound system used by Bouma [27] and others is similar to the peatland system in Minnesota, except that the application is by subsurface pipe.

The soil mound system for a shallow soil over creviced rock is shown in Figure 2-4. Such systems are alternatives to treatment and discharge to surface waters where adverse soil conditions exist. The soil mound can be used for [28]:

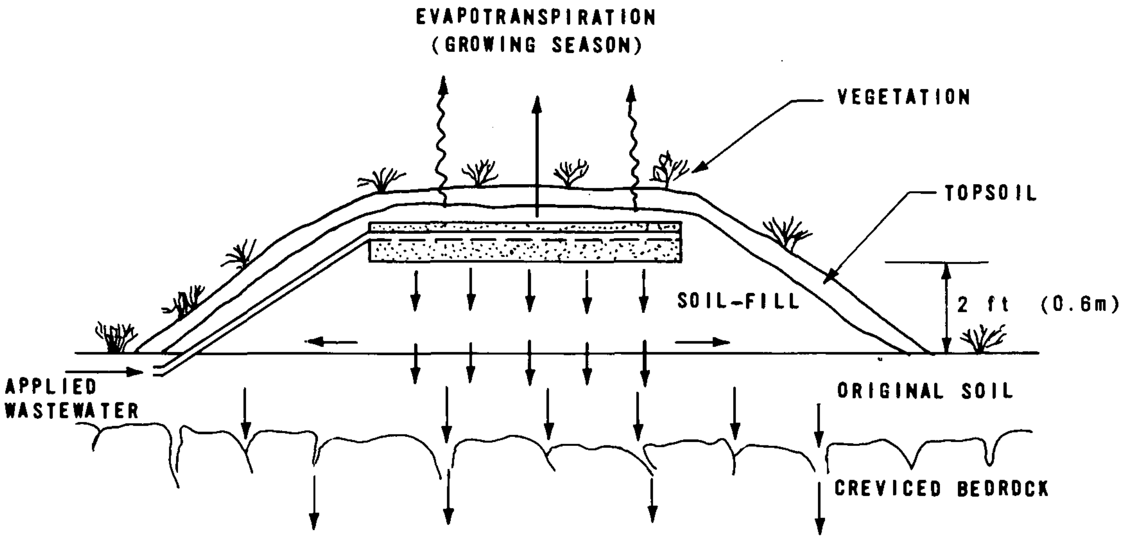
1. Shallow soils (<3 ft or 1 m) over creviced or otherwise rapidly permeable bedrock

2. Sites with slowly permeable soils

3. Sites with seasonally high groundwater

FIGURE 2-4

SUBSURFACE APPLICATION TO SOIL MOUND OVER CREVICED BEDROCK



Bouma has reported on an experimental soil mound system at Sturgeon Bay, Wisconsin [27]. The work is part of the Small Scale Waste Management Project at the University of Wisconsin. The mound, shown in Figure 2-4, was designed for 2 in./d (5 cm/d) but was actually dosed at about half of that rate. Septic tank effluent was dosed four times a day through a network of 1 in. (2.5 cm) PVC pipes. The actual loading was 6.4 in./wk (16.3 cm/wk). Treatment performance of this mound system is given in Table 2-7.

TABLE 2-7
TREATMENT PERFORMANCE OF AN EXPERIMENTAL
SOIL MOUND SYSTEM IN WISCONSIN [27]
mg/L

Constituent	Influent	Effluent
BOD	90	0
COD	256	42
Ammonia nitrogen	56	2
Total nitrogen	62	56
Total phosphorus	15	8
Fecal coliforms, count/mL	2,500	5
Total coliforms, count/mL	37,000	54

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CHAPTER 3

TECHNICAL PLANNING AND FEASIBILITY ASSESSMENT

3.1 Purpose and Scope

The purpose of this chapter is to describe those aspects of land treatment that are important to a technical and economic feasibility assessment. The major divisions of this chapter are:

- Approach to development of alternatives
- Evaluation of unit processes
- Wastewater quality
- Regional site characteristics
- Other planning considerations
- Evaluation of alternatives

The scope of the chapter is directed at those factors that are unique to the formulation and evaluation of land treatment alternatives. Planning and feasibility considerations that are common to conventional wastewater management systems are adequately discussed elsewhere.

It is important to be aware of the distinction made between "alternative land treatment processes" (described in Chapter 2) and "system alternatives." The term "land treatment process" refers to the unit process only (e.g., slow rate, overland flow) whereas the term "system alternatives" includes the entire wastewater management facility (transmission, treatment processes, storage, collection, and discharge facilities).

This chapter presents planning level information related to unit process selection, the wastewater characteristics important to land treatment systems, and the significant regional characteristics involved in developing land treatment system alternatives. It is expected that the user will also refer to Chapter 5, or for small systems Chapter 6, during the feasibility assessment to obtain more technical details for the development of alternative systems. The evaluation of the resulting systems is then discussed in Section 3.7.

3.2 Approach to Development of Alternatives

Three major factors combine to determine the type of land treatment process that can be used on a given site:

1. Soil permeability
2. Wastewater quality
3. Discharge quality criteria

For a given site, the soil permeability can be determined. The other two factors, however, must be considered as variables. The wastewater quality to be applied depends on the preapplication treatment. The discharge criteria are also variable because there is a choice between surface water and groundwater discharge. There is also the possibility of collecting the treated water for other uses in agriculture or industry.

The many variables and options associated with land treatment processes and systems require the use of an organized, systematic approach to selecting alternatives. Many approaches have been considered but only three have been commonly used. The three most common approaches to developing land treatment alternatives are:

1. No constraints approach--There are no prior constraints placed on the study. The entire study area is investigated for potential sites while considering the whole spectrum of land treatment processes and combinations to develop alternatives.

2. Process constrained approach--The study begins with some prior constraints that limit consideration of alternatives to certain land treatment processes. Potential sites are identified within the reduced spectrum of land treatment processes created by the constraints.

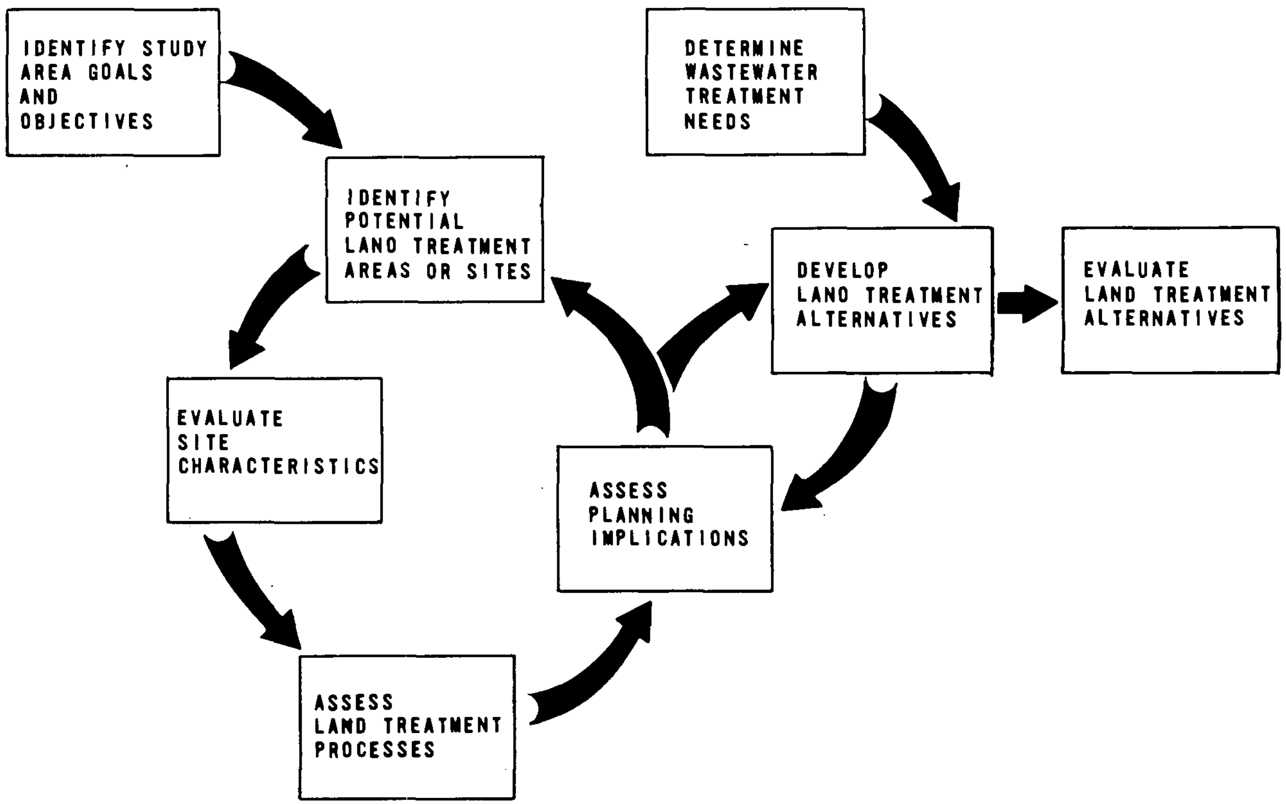
3. Site-constrained approach--A predetermined site (or sites) is available, and treatment processes are evaluated to match the site(s) and the project objectives.

The approach to the development of land treatment alternatives is iterative in nature, as illustrated in Figure 3-1. This iterative process is best achieved in the no-constraints approach. Within the iterative cycle of site identification, site evaluation, process assessment, and planning implications assessment, there are so many degrees of freedom available that several cycles or iterations may be necessary to define and develop an alternative. When the number of sites or processes is predetermined, as in the process-constrained or site-constrained approach, fewer alternative systems can be developed.

A variation of the no-constraints approach is the use of an inductive analytical planning process. Regional goals and objectives are

FIGURE 3-1

PLANNING SEQUENCE FOR THE DEVELOPMENT OF LAND TREATMENT ALTERNATIVES



initially identified and the ability of land treatment to help achieve these and other benefits is assessed. Included are the possibilities for reclamation and resource recovery such as recycling of nutrients through the production of cash crops, preservation of agriculture and open space, and the implementation of other land use planning objectives. Thus, land treatment may be viewed as a means to an end rather than an end in itself.

3.3 Evaluation of Unit Processes

To evaluate the applicability of land treatment processes, the treatment objectives and wastewater quality must be known. The preliminary design of land treatment processes can then be accomplished using average flowrates and hydraulic loading rates. In this section, hydraulic loading rates are discussed for each land treatment process. Guidance is then provided for preliminary planning purposes on land area requirements, preapplication treatment, storage, and recovery of renovated water.

3.3.1 Land Treatment Processes

The first step in evaluating land treatment unit processes is to identify the processes that may be suitable for the requirements and conditions of the study area. The description of treatment process capabilities and objectives in Chapter 2 will provide a useful background for this purpose. The types of factors that should generally be considered at this stage include:

- The ability of each process to meet treatment requirements
- The disposition of applied wastewater in relation to water needs
- The predominant characteristics of the study area that may dictate certain land treatment processes
- The desired secondary objectives, such as increased irrigation water supply

3.3.1.1 Slow Rate Process

For the slow rate process, the hydraulic loading rate can be determined initially from the use of the water balance:

$$\text{Precipitation} + \text{Wastewater applied} = \text{Evapotranspiration} + \text{Percolation} + \text{Runoff} \qquad (3\text{-}1)$$

Effluent runoff is usually not desirable for the slow rate process. For planning, the relationship between precipitation and evapotranspiration on a mean annual basis can be taken from Figure 3-2. If the precipitation and evapotranspiration balance, an estimate of wastewater application rates can be made from the soil permeability rates as presented in Figure 3-3. For example, a slow permeability soil could be loaded at 1.0 to 3 in./wk (2.5 to 7.6 cm/wk) from Figure 3-3. If evapotranspiration exceeds precipitation, the effluent applied can be increased to equal the sum of net evapotranspiration and soil permeability. For example, in central Texas, where net evapotranspiration is 36 in./yr (90 cm/yr), the application rate could be increased by 0.7 in./wk (1.8 cm/wk) on an annual average to a total of 1.7 to 3.7 in./wk (4.3 to 9.4 cm/wk). Application rates beyond 4 in./wk (10 cm/wk) are normally defined as rapid infiltration and involve different considerations.

The shaded area in Figure 3-3 represents the range of average, long-term infiltration rates when considering only soil permeability derived from clear water. The range of values shown in Figure 3-3 as "Range of Application Rates in Practice" is indicative of the many factors that must be considered in selecting the final application rate. Such considerations include crop water needs and tolerances, nutrient balance, and reductions in application rates for crop harvesting or to account for algae in the wastewater.

The hydraulic loading is also affected by the climate and crop selection. The climate will affect the growing season and will dictate the period of application and the amount of storage required. Crop water tolerances and nutrient requirements can directly affect hydraulic loading rates. The following factors affect the selection of crops:

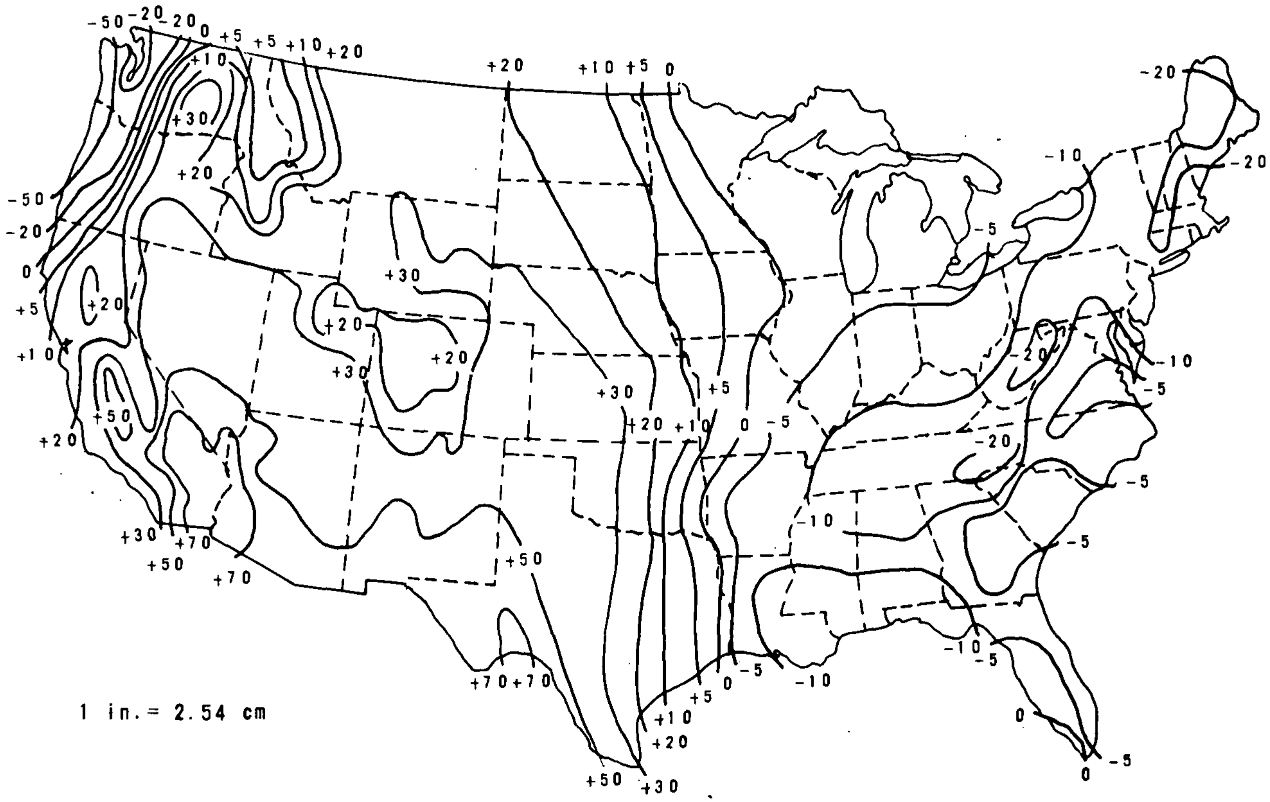
1. Suitability to local climate and soil conditions
2. Consumptive water use and water tolerance
3. Nutrient uptake and sensitivity to wastewater constituents
4. Economic value and marketability
5. Length of growing season
6. Ease of management
7. Public health regulations

3.3.1.2 Rapid Infiltration Process

Rapid infiltration systems are designed on the basis of hydraulic capacity of the soil and the underlying geology. The relationship shown in Figure 3-3 can be used for approximation of hydraulic loading rates,

FIGURE 3-2

POTENTIAL EVAPOTRANSPIRATION VERSUS MEAN ANNUAL PRECIPITATION [1]
Inches

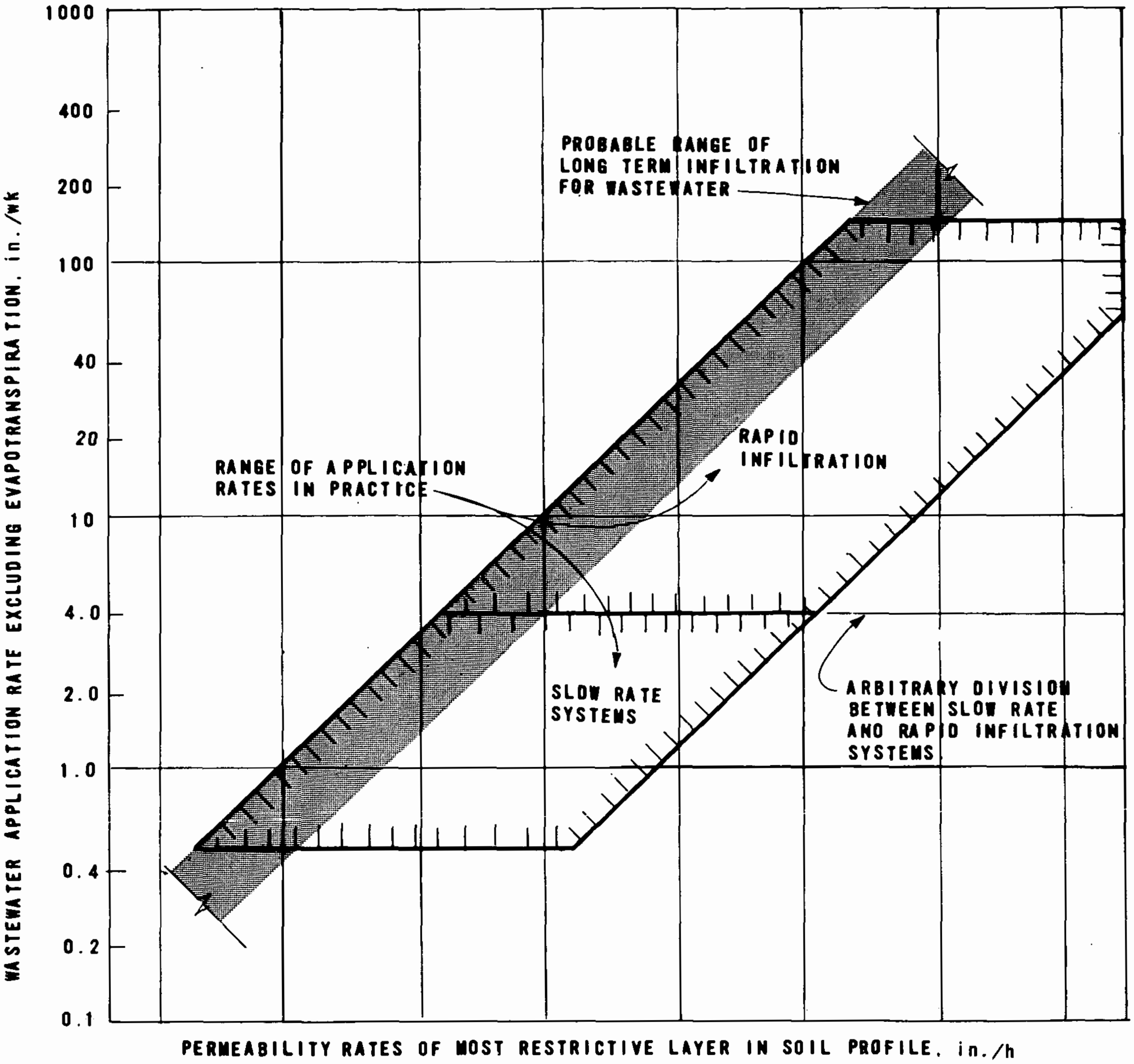


+ POTENTIAL EVAPOTRANSPIRATION MORE THAN MEAN ANNUAL PRECIPITATION

− POTENTIAL EVAPOTRANSPIRATION LESS THAN MEAN ANNUAL PRECIPITATION

FIGURE 3-3

SOIL PERMEABILITY VERSUS RANGES OF APPLICATION RATES
FOR SLOW RATE AND RAPID INFILTRATION TREATMENT



PERMEABILITY* SOIL CONSERVATION SERVICE DESCRIPTIVE TERMS						
VERY SLOW	SLOW	MODERATE-LY SLOW	MODERATE	MODERATE-LY RAPID	RAPID	VERY RAPID
< 0.06	0.06-0.20	0.20-0.60	0.60-2.0	2.0-6.0	6.0-20.0	> 20.0

* MEASURED WITH CLEAR WATER

1 in./wk = 2.54 cm/wk

if the permeability of the most restrictive layer in the soil profile is known. Application rates in the low end of the range should be chosen if any of a number of conditions exist which may be adverse. These include: (1) wide variations in soil types and permeability, (2) shallow soil profiles, and (3) shallow or perched water tables. Reductions in application rates may also be necessary if the system is to be managed to optimize denitification. The cycle of wastewater application and resting must be defined.

3.3.1.3 Overland Flow Process

For overland flow, the application rate depends primarily on the expected treatment performance and the level of preapplication treatment. If primary effluent is used, an application rate in the range of 2.5 to 6 in./wk (6.4 to 15 cm/wk) is usually necessary to produce the effluent quality shown in Table 2-2. The lower end of this range should be considered where: (1) terrace slopes will be greater than about 6%, (2) terraces are less than 150 ft (45 m) long, or (3) climatic conditions are poor. The upper end of the scale can be used when evapotranspiration rates are high, or when a moderate amount of percolation can be expected to take place. In cases where overland flow is to be used as a polishing process or for advanced treatment following preapplication treatment, application rates as high as 6 to 16 in./wk (15 to 40 cm/wk) may be used. These rates have been used in demonstration systems with slopes of 2 to 3% that are 120 ft (36 m) long.

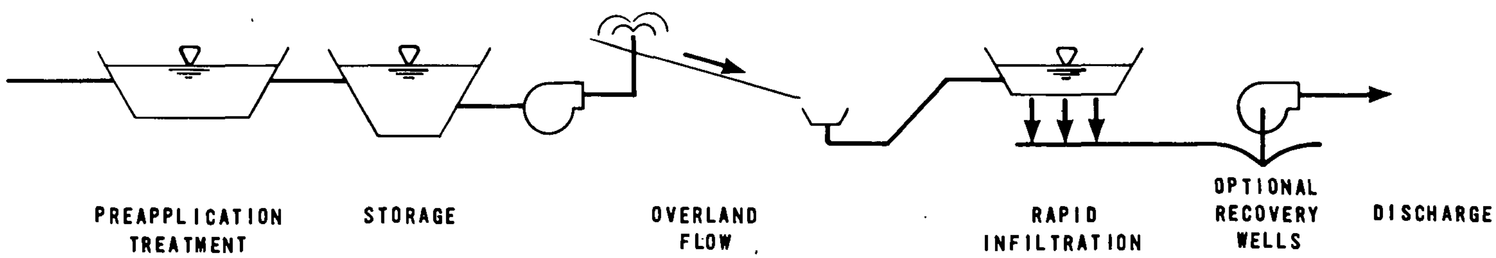
3.3.1.4 Combinations

Combinations of land treatment processes in series can be considered. Examples of two such systems are shown schematically in Figure 3-4. In the first example, rapid infiltration is used after overland flow to further reduce concentrations of BOD, suspended solids, and phosphorus. Because of the increased reliability and overall treatment capability, the application rates for the overland flow process could be higher than normal.

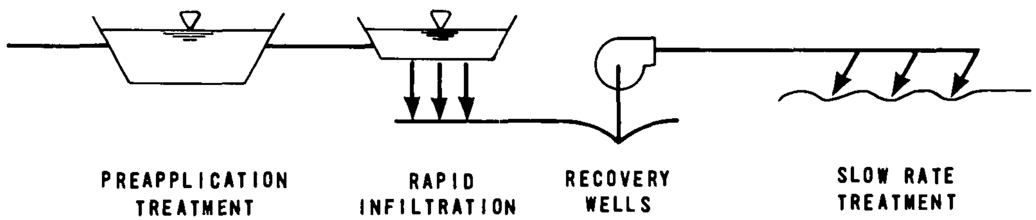
In the second example, the rapid infiltration process precedes slow rate treatment. The recovered renovated water should meet even the most restrictive requirements for use on food crops. The unsaturated zone can be used for storage of renovated water to be withdrawn on a schedule consistent with crop needs.

FIGURE 3-4

EXAMPLES OF COMBINED SYSTEMS



a) OVERLAND FLOW FOLLOWED BY RAPID INFILTRATION



b) RAPID INFILTRATION FOLLOWED BY SLOW RATE TREATMENT

3.3.2 Land Area Requirements

The total land area required for a land treatment system consists of the actual land to which wastewater is applied and the additional area required for buffer zones, storage reservoirs, access roads, pumping stations, preapplication treatment, and maintenance and administration buildings. In addition, it may be necessary to set aside some land for future expansion or emergencies.

The total land area requirement can be estimated for preliminary planning using the nomograph in Figure 3-5. To use the nomograph, first draw a line through appropriate points on the design-flow and application-rate axes to the pivot line. Draw a second line from the intersection of the first line with the pivot line through the appropriate point on the nonoperating time axis. (Nonoperating time is the period during the year when the system is shut down for weather or other reasons.) The calculated total area is then noted at the intersection of that axis with the second line. This total area includes land for application, roads, storage, and buildings. The total area with a 200 ft (61 m) wide buffer zone allowance is read from the right-hand side of the axis; the total area with no allowance for buffer zones is read from the left-hand side.

3.3.3 Preapplication Treatment

Preapplication treatment of wastewater may be necessary for a variety of reasons, including (1) maintaining a reliable distribution system, (2) allowing storage of wastewater without creating nuisance conditions, (3) obtaining a higher level of wastewater constituent removal, (4) reducing soil clogging, and (5) reducing possible health risks. A summary of preapplication treatment practice is presented in Table 2-1.

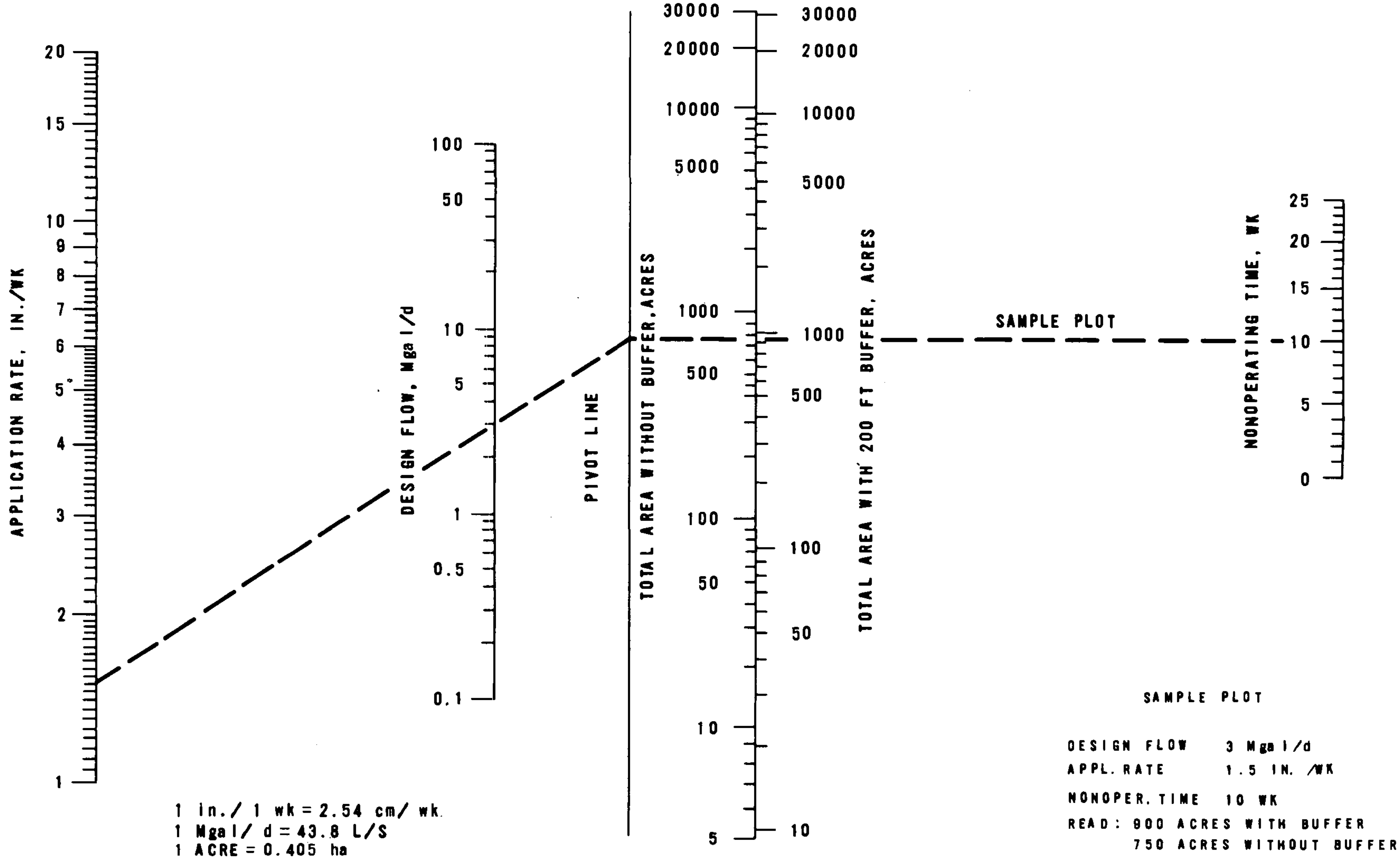
3.3.4 Storage

Storage is provided primarily for nonoperating periods and periods of reduced application rates resulting from climatic constraints. In most situations, however, where this requirement is small, storage may still be necessary for system backup, flow equalization, and proper agricultural management including periods for harvesting. In the planning stage, it will usually be important to determine the approximate volume required for each land treatment alternative so that storage costs can be estimated.

It has been shown that slow rate and overland flow irrigation systems can usually operate successfully below 32°F (0°C), and 25°F (-4°C) is

FIGURE 3-5

TOTAL LAND REQUIREMENT (INCLUDES LAND FOR APPLICATION, ROADS, STORAGE, AND BUILDINGS)



sometimes used as a lower limit. A conservative method for predicting the number of days that are too cold for operation is to assume that application is suspended on all days in which the mean temperature is below 32°F (0°C). This method has the advantage of using readily accessible data.

The National Climatic Center in Asheville, North Carolina, has conducted an extensive study of climate and weather variations throughout the United States. A computer program has been developed to use weather station data in estimating the amount of wastewater storage required at a location because of climatic constraints [2]. For planning and preliminary feasibility assessment, a value for storage days can be found using Figure 3-6. The map gives the number of nonapplication days for which storage would normally be required for a 20 year return period on the basis of climatic factors alone. Additional storage time may be required if reduced winter loading rates are used for overland flow (see 5.1.4.1).

Rapid infiltration basins which are intermittently flooded can often be operated year-round regardless of climatic conditions. The only storage that might be required is that for system backup or extremely severe climatic conditions. During extended periods of cold weather, an ice layer may form on the surface of the bed. However, at Lake George, New York, and at Fort Devens, Massachusetts, this has not proved to be a problem. The application of the wastewater merely floats the ice and infiltration continues. This condition should prevail whenever the soil is porous and well drained; otherwise, precautions are advised.

3.3.5 Recovery of Renovated Water

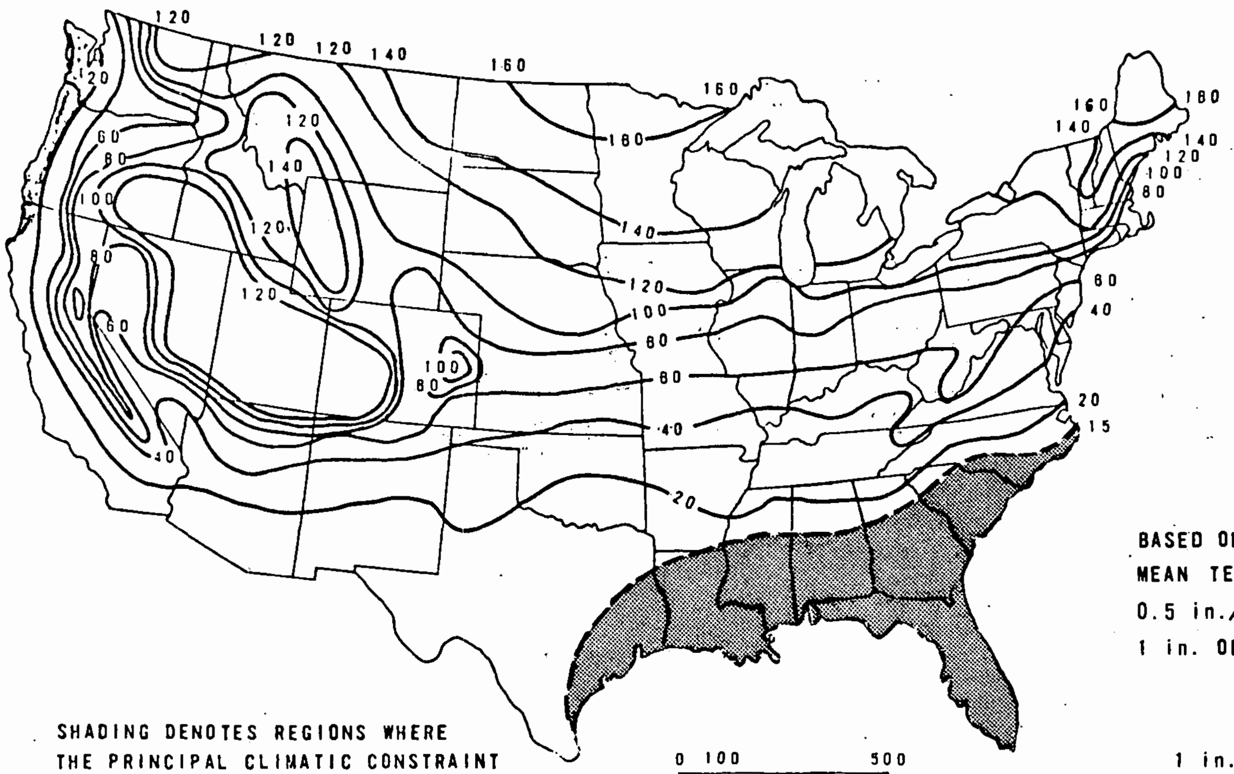
Recovery of the applied wastewater after renovation for reuse or further treatment is often a part of the overall land treatment process. The means to recover renovated water include (1) surface runoff collection, (2) underdrains, (3) recovery wells, and (4) tailwater return. The applicability of these systems to the treatment processes is summarized in Table 3-1. These recovery methods are described in various situations in Chapter 5.

3.4 Wastewater Quality

Knowledge of the quality of the wastewater to be treated is needed in planning to properly assess preapplication treatment needs or special management needs. The major constituents in typical untreated domestic wastewater are presented in Table 3-2. Preapplication treatment using primary sedimentation will reduce BOD and suspended solids (SS), but will not greatly affect nitrogen or phosphorus concentrations. Treatment in oxidation ponds, aerated lagoons, or other biological

FIGURE 3-6

ESTIMATED WASTEWATER STORAGE DAYS BASED ONLY ON CLIMATIC FACTORS [2]



BASED ON 32°F (0°C) MEAN TEMPERATURE
0.5 in./d PRECIPITATION,
1 in. OF SNOWCOVER

SHADING DENOTES REGIONS WHERE THE PRINCIPAL CLIMATIC CONSTRAINT TO APPLICATION OF WASTEWATER IS PROLONGED WET SPELLS

0 100 500
SCALE MILES

1 in. = 2.54 cm

treatment processes further reduces the BOD and SS, and may reduce nitrogen or phosphorus.

TABLE 3-1

APPLICABILITY OF RECOVERY SYSTEMS FOR RENOVATED WATER

Recovery system	Slow rate	Rapid infiltration	Overland flow
Surface runoff collection			
Effluent	NA	NA	Collect[a]
Stormwater	Sediment control	NA	Erosion control
Underdrains	Groundwater control and effluent recovery	Groundwater control and effluent recovery	NA
Recovery wells	Usually NA	Groundwater control and effluent recovery	NA
Tailwater			
Sprinkler application	NA	NA	NA
Surface application	25-50% of applied flow	NA	NA

NA = not applicable.

a. Disinfect if required before discharge; provide for short-term recycling of wastewater after extended periods of shutdown, if effluent requirements are stringent.

TABLE 3-2

IMPORTANT CONSTITUENTS IN TYPICAL DOMESTIC WASTEWATER [3] mg/L

	Type of wastewater		
Constituent	Strong	Medium	Weak
BOD	300	200	100
Suspended solids	350	200	100
Nitrogen (total as N)	85	40	20
Organic	35	15	8
Ammonia	50	25	12
Nitrate	0	0	0
Phosphorus (total as P)	20	10	6
Organic	5	3	2
Inorganic	15	7	4

Land treatment processes are capable of removing large amounts of BOD and SS as well as nutrients, trace elements, and microorganisms.

Hydraulic loading rates were discussed in the previous section and will usually govern site area. However, in some cases constituent loading rates may dictate land area needs. For preliminary planning purposes, the BOD loading rate guidelines in Table 3-3 can be used to determine whether hydraulic or constituent loadings will control the design. Using hydraulic application rates appropriate for the process and the BOD concentrations of the wastewater, BOD loadings can be computed and compared with the values in Table 3-3.

TABLE 3-3

TYPICAL BOD LOADING RATES
lb/acre·d

	Slow rate	Rapid infiltration	Overland flow
Typical range for municipal wastewater[a]	0.2-5	20-160	5-50

a. Loading rates represent total annual loading divided by the number of days in the operating season.

Exceeding the typical values in Table 3-3 will not necessarily be detrimental to the system. The planner or engineer should be aware that special management may be required above these values and provide appropriate safeguards. Loading rates are discussed in detail for each process in Section 5.1.

For trace elements, the concentrations in wastewater vary tremendously with location and percentage of industrial flows. Ranges of values in untreated wastewater, primary effluent, and secondary effluent are presented in Table 3-4. Also included in Table 3-4 are the EPA drinking water standards for these constituents for comparison. Concentrations of trace elements in the wastewater after preapplication treatment which are equal to or less than those recommended for drinking water should represent no management concern. If one or more values is expected to exceed these recommendations, the more detailed discussion of trace element loadings should be consulted in Section 5.1.

3.5 Regional Site Characteristics

Compared to other forms of wastewater treatment, land treatment systems and processes are very site specific. The objective of characterizing physical features of the region is to provide the basic information necessary to make a preliminary assessment of land treatment processes and systems within the study area. The physical regional features that are considered important include: topography, soils, geology, climate, surface water hydrology and quality, and groundwater hydrology and quality. In this section, these topics, along with sources of data, are discussed as they relate to the land treatment processes described in Chapter 2.

TABLE 3-4

CONCENTRATION OF TRACE ELEMENTS IN VARIOUS U.S. WASTEWATERS mg/L

Element	Untreated wastewater[a]	Primary effluents[a]	Secondary effluents[a]	EPA recommended drinking water standards[b]
Arsenic	0.003	0.002	<0.005-0.01	0.05
Cadmium	0.004-0.14	0.004-0.028	0.0002-<0.02	0.01
Chromium	0.02-0.700	<0.001-0.30	<0.010-0.17	0.05
Copper	0.02-3.36	0.024-0.13	0.05-0.22	1.0
Iron	0.9-3.54	0.41-0.83	0.04-3.89	0.3
Lead	0.05-1.27	0.016-0.11	0.0005-<0.20	0.05
Manganese	0.11-0.14	0.032-0.16	0.021-0.38	0.05
Mercury	0.002-0.044	0.009-0.035	0.0005-0.0015	0.002
Nickel	0.002-0.105	0.063-0.20	<0.10-0.149	No standard
Zinc	0.030-8.31	0.015-0.75	0.047-0.35	5.0

a. The concentrations presented encompass the range of values reported in references [4, 5, 6, 7, 8, 9, 10, 11].

b. Reference [12].

3.5.1 Site Identification

The complexity of site identification depends on the size of the study area and the nature of the land use. One approach is to start with land use plans and identify undeveloped land. A tool that can be used is the map overlay technique. Map overlays can help the planner or engineer to organize and study the combined effects of land use, slope, relief, and soil permeability. Criteria can be set on these four factors, and areas that satisfy the criteria can then be located. If this procedure is used as a preliminary step in site identification, the criteria should be reassessed during each successive iteration. Otherwise, strict adherence to such criteria may result in overlooking either sites or land treatment opportunities.

Information required to make a map overlay includes:

Source	Information
USGS quad sheets	Base map with topography
Land use maps	Existing and future land use
Soil maps	Soil permeability and slope

3.5.2 Site Selection

The process of characterizing, evaluating, and selecting sites is usually iterative in nature. The first screening of sites may be done using overlays or considering only land use and soil permeability. Subsequent evaluations include factors such as those presented in Table 3-5.

Once the full array of site characteristics is assembled, and sites have been screened for acceptability, the selection process can include numerical rating systems. The relative effect of each characteristic can be determined by assigning weighting values. The resulting ratings should include input from as many qualified planners and engineers as possible to reduce bias.

3.5.3 Topography

Three main topographic features that affect the suitability of a site for land treatment of wastewater are: slope, relief, and susceptibility to flooding. These features play a major role in the preliminary identification and evaluation of potential sites. A less important topographic feature--aspect--may also affect site suitability. The amount of solar radiation a site receives is related to the aspect, or direction, of the slope. This will affect the consumptive water use of crops, vegetation, or woodland being considered. The type of climate will determine the impact that aspect has on site suitability.

The USGS publishes topographic maps for most areas in the United States. These maps usually have scales of 1:24 000 (7.5 minute series) or 1:62 500 (15 minute series), and they are suitable for determining the slope and elevation of a region for a project in the planning stage.

Examination of topographic features should not be limited to the potential site. Adjacent topography should be evaluated for its effects on the site, particularly with respect to drainage and areas of potential erosion. Adjacent land characteristics to be identified are those that may potentially (1) add stormwater runoff to the site, (2) back up water onto the site, (3) provide relief drainage, or (4) cause the appearance of groundwater seeps.

TABLE 3-5

SITE SELECTION GUIDELINES

Characteristic	Land treatment process affected	Effect
Soil permeability	Overland flow	High permeability soils are more suitable to other processes.
	Rapid infiltration and slow rate	Application rates increase with permeability.
Potential groundwater pollution	Rapid infiltration and slow rate	Affected by the (1) proximity of the site to a potential potable aquifer, (2) presence of an aquiclude, (3) direction of groundwater flow, and (4) degree of groundwater recovery by wells or underdrains.
Groundwater storage and recovery	Rapid infiltration	Capability for storing percolated water and recovery by wells or underdrains is based on aquifer depth, permeability, aquiclude continuity, effective treatment depth, and ability to contain the recharge mound within the desired area.
Existing land uses	All processes	Involves the occurrence and nature of conflicting land use.
Future land use	All processes	Future urban development may affect the ability to expand the system.
Size of site	All processes	If there are a number of small parcels, it is often difficult to control the needed area and implement the plan.
Flooding hazard	All processes	May exclude or limit site use.
Slope	All processes	Steep slopes may (1) increase capital expenditures for earthwork, and (2) increase the erosion hazard during wet weather.
	Rapid infiltration	Steep slopes often affect groundwater flow pattern.
	Overland flow	Steep slopes reduce the travel time over the treatment area and treatment efficiency. Flat land may require extensive earthwork to create slopes.

3.5.3.1 Slope

Excessive slope is an undesirable characteristic for land application because (1) it increases the amount of runoff and erosion that will occur, (2) it may lead to unstable soil conditions when the soil is saturated, and (3) it makes crop cultivation difficult or, in some cases, impossible. Criteria for maximum slope will depend, in part, on both the amount of land with moderate slopes (less than 10%) that is available and the land treatment process. Successful agriculturally related systems using slopes of 15% or more and silviculture type systems on wooded slopes of up to 40% have been reported [13].

The system configuration and earthwork requirements, particularly for overland flow and rapid infiltration treatment, are important factors that will determine the maximum slopes permissible for a potential site. If rolling terrain is to be used for cultivated agriculture, the slope should not exceed about 15%. Grass and forage crops can be adapted to steeper slopes. Relatively flat land is normally required for surface irrigation, although contour furrows have been used on slopes as steep as 5%.

For rapid infiltration, the primary topographic concern is that lateral water movement be controlled so that percolation rates of lower basins are not affected. At Westby, Wisconsin, basins have been terraced into a 5% sloping hillside, but there are no underdrains, and the lateral movement of water from the upper basins reportedly affects the percolation rates in the lower basins.

For overland flow, the primary requirement is that the existing topography be such that terrace slopes of 2 to 8% can be formed economically. The cost and impact of the earthwork required are the major constraints.

3.5.3.2 Relief

Relief is the relative elevation or elevation difference between one part of the land treatment system and another. Relief and terrain are interrelated as they affect the economics of pumping wastewater. The pumping cost is the principal annual operation cost when large elevation differences exist between the wastewater source and the land treatment site, reuse location, or discharge point. This cost must be weighed against the cost of constructing gravity conveyance to sites that may have greater distances between system components but favorable relief characteristics.

For silviculture (where sprinkler irrigation of forest land is considered), more liberal relief and slope tolerances are possible because the nature of the root system, forest litter, and vegetation offer resistance to direct surface runoff and resulting erosion.

3.5.3.3 Susceptibility to Flooding

Location of land treatment systems within a flood plain can be either an asset or a liability, depending on the approach taken to planning and design. Flood prone areas may be undesirable because of the highly variable drainage characteristics usually encountered and potential flood damage to the physical components of the treatment system. On the other hand, flood plains, alluvial deposits, and delta formations may be the only deep soils available in the area. With careful design and choice of application techniques, a land treatment system can be an integral part of a flood plain management plan. The flooding hazard of a potential site should be evaluated with respect to both the severity of floods that could occur and the extent of the area flooded.

The extent of flood protection built into a land treatment system will depend on local conditions. In some cases, it may be preferred to allow the site to flood as needed and provide the protection through offsite storage. Further, flood plains are generally unacceptable for construction of dwellings or commercial buildings, offering an opportunity for imaginative uses of land treatment systems. It should be noted that crops can be grown in flood plains if the infrequency of floods makes it economical to farm.

Descriptions of severe floods that have occurred in the United States, and summaries of all notable floods of each year, are published as USGS Water Supply Papers. Maps of certain localities showing the area inundated in past floods are published as Hydrologic Investigation Atlases by the USGS. More recent maps of flood prone areas have been produced by the USGS in many areas of the country as part of the "Uniform National Program for Managing Flood Losses." The maps are based on standard 7.5 minute (1:24 000) topographic sheets; and, by means of overprint in black and white, they identify those areas that have a 1 in 100 chance of being inundated in any given year. Additionally, other detailed flood information is usually available from local offices of the U.S. Corps of Engineers and the flood control districts that deal with such problems firsthand.

3.5.4 Soils

The soil at a potential site should be identified in terms of its hydraulic, physical, and chemical characteristics. Important physical characteristics include texture, structure, and soil depth. Important hydraulic characteristics are infiltration rate and permeability. Chemical characteristics that may be important include pH, cation exchange capacity, nutrient levels, and the adsorption and filtration capabilities for various inorganic ions.

Information on soil properties can be obtained from several sources, but the SCS soil surveys are the primary source. Well logs can also offer additional data on soils and geology. Soil surveys will normally provide soil maps delineating the apparent boundaries of soil series with their surface texture. A written description of each soil series provides limited information on chemical properties, engineering applications, interpretive and management information, slopes, drainage, erosion potentials, and general suitability for most kinds of crops grown in the particular area. Additional information on soil characteristics and information regarding the availability of soil surveys can be obtained directly from the SCS. The SCS serves as the coordinating agency for the National Cooperative Soil Survey, and as such, cooperates with other government agencies, universities, and agricultural extension services in obtaining and distributing soil survey information.

3.5.4.1 Soil Physical Characteristics

The physical properties of texture and structure are important because of their effect on hydraulic properties. Soil textural classes are defined on the basis of the relative percentage of the three classes of particle size--sand, silt, and clay. Sand particles range in size from 2.0 mm to 0.05 mm; silt particles range from 0.05 mm to 0.002 mm; and particles smaller than 0.002 mm are clay. From the particle size distribution, the textural class can be determined using the textural triangle shown in Figure 3-7. Terms commonly used to describe soil texture and the relationship to textural class names as established by the SCS are listed in Table 3-6.

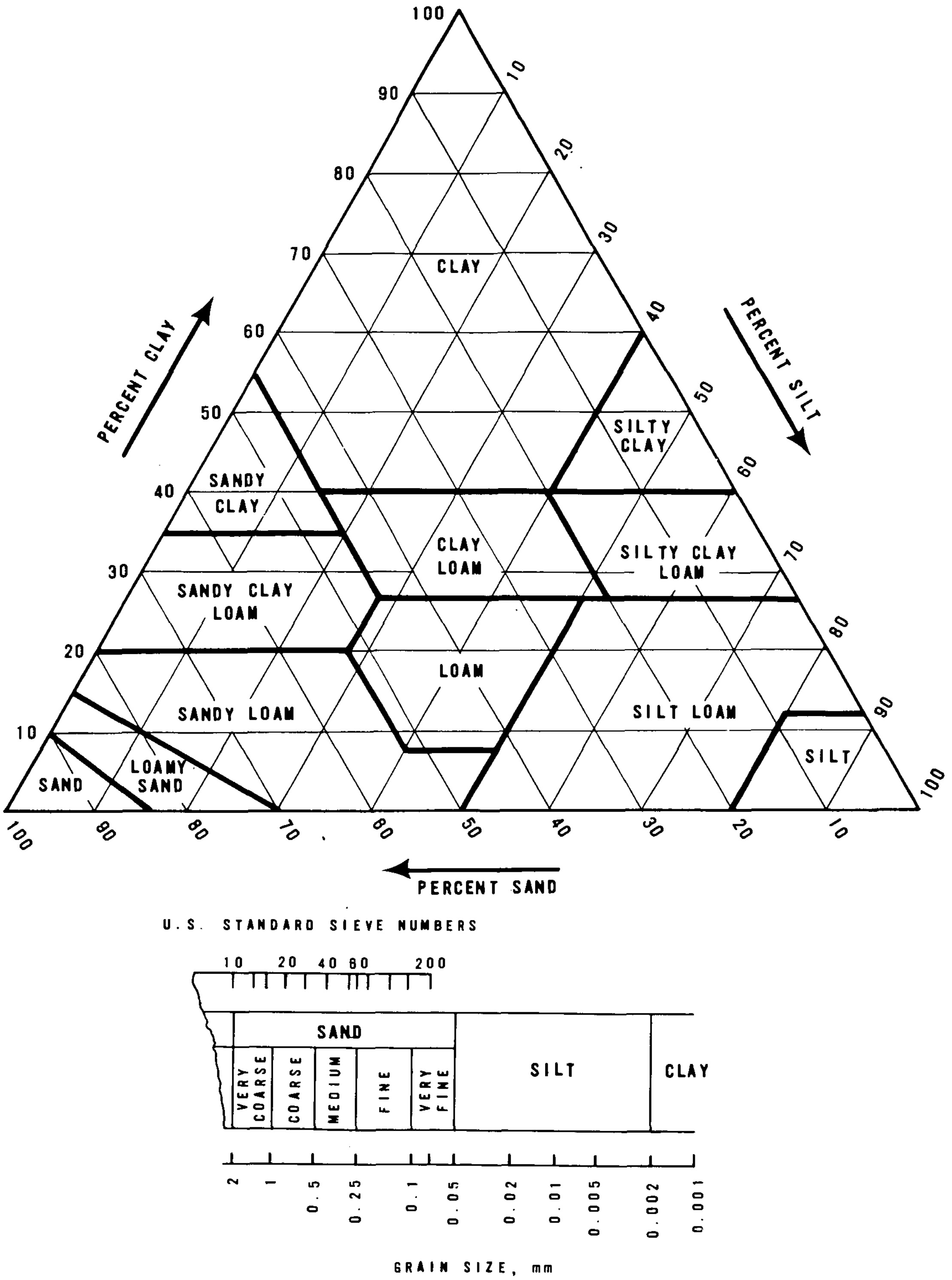
Fine-textured soils do not drain well and retain large percentages of water for long periods of time. As a result, crop management is more difficult than with more freely drained soils such as loamy soils. Fine-textured soils are generally best suited to overland flow systems. Medium-textured soils exhibit the best balance for wastewater renovation and drainage. Loamy (medium texture) soils are generally best suited for slow rate systems (crop irrigation).

Coarse-textured soils (sandy soils) can accept large quantities of water and do not retain moisture very long. This feature is important for crops that cannot withstand prolonged submergence or saturated root zones. Soil structure refers to the aggregation of individual soil particles. If these aggregates resist disintegration when the soil is wetted or tilled, it is well structured. The large pores in well-structured soils conduct water and air, making well-structured soils desirable for infiltration.

Adequate soil depth is important for root development, for retention of wastewater components on soil particles, and for bacterial action.

FIGURE 3-7

PROPORTIONS OF SAND, SILT, AND CLAY IN
THE BASIC SOIL-TEXTURAL CLASSES [14]



Plant roots can extract water from depths ranging from 1 to 9 ft (0.3 to 2.7 m) or more. Retention of wastewater components, such as phosphorus, heavy metals, and viruses, is a function of residence time of wastewater in the soil and the degree of contact between soil colloids and the wastewater components.

TABLE 3-6

SOIL TEXTURAL CLASSES AND GENERAL TERMINOLOGY USED IN SOIL DESCRIPTIONS [15]

General terms		Basic soil textural class names
Common name	Texture	
Sandy soils	Coarse	Sand Loamy sand
	Moderately coarse	Sandy loam Fine sandy loam
Loamy soils	Medium	Very fine sandy loam Loam Silt loam Silt
	Moderately fine	Clay loam Sandy clay loam Silty clay loam
Clayey soils	Fine	Sandy clay Silty clay Clay

The type of land treatment system being considered will determine whether soil depth is adequate. The minimum soil depth for most systems that rely on infiltration (rapid infiltration and slow rate) is about 3 to 5 ft (1.0 to 1.5 m). Soil depths of 1 to 2 ft (0.3 to 0.6 m) can support grass or turf. Overland flow systems require sufficient soil depth to form slopes that are uniform and to maintain a vegetative cover.

3.5.4.2 Soil Hydraulic Properties

Drainage of water within the soil depends on texture, structure, and the absence of subsurface constraints to the flow of water. An example of a vertical constraint would be an impermeable clay, hardpan, or rock strata underlaying a sandy soil. The lateral transmissibility and percolation rates may limit the application rate unless they are equal to or higher than the infiltration rate. For high rate systems that depend largely on vertical water movement, the permeability of the most restricting layer in the upper several feet of soil will usually determine the maximum hydraulic loading.

The most recent permeability class definitions developed by the SCS are shown in Table 3-7. Soil permeabilities other than the values shown in Table 3-7 (for the respective permeability class) may appear in soil literature depending on the age of the document and local variations in interpretation. The soil permeability ranges normally associated with each land treatment process are compared along with the corresponding permeability and textural class in Table 3-8.

TABLE 3-7

PERMEABILITY CLASSES FOR SATURATED SOIL [15]

Soil permeability, in./h	Class
<0.06	Very slow
0.06 to 0.2	Slow
0.2 to 0.6	Moderately slow
0.6 to 2.0	Moderate
2.0 to 6.0	Moderately rapid
6.0 to 20	Rapid
>20	Very rapid

1 in./h = 2.54 cm/h

TABLE 3-8

TYPICAL SOIL PERMEABILITIES AND TEXTURAL CLASSES FOR LAND TREATMENT PROCESSES

	Principal processes			Other processes	
	Slow rate	Rapid infiltration	Overland flow	Wetlands	Subsurface
Soil permeability range, in./h	0.06-20	2.0	0.2	0.06-2.0	0.2-20.0
Permeability class range	Moderately slow to moderately rapid	Rapid	Slow	Slow to moderate	Moderately slow to rapid
Textural class range	Clay loams to sandy loams	Sands and sandy loams	Clays and clay loams	Clay loams to silt loams	Clay loams to sands
Unified Soil Classification [16]	GM-d, SM-d, ML, OL, MH, PT	GW, GP, SW, SP	GM-u, GC, SM-u, SC, CL, OL, CH, OH	....	....

3.5.4.3 Soil Chemical Characteristics

The balance of chemical constituents in soil is important to plant growth and wastewater renovation. The mechanisms of retention of certain constituents by the soil are discussed in Appendixes A through E. Chemical properties of the soil should be known by the engineer prior to design for the purpose of determining changes in soil chemistry that could occur during operation. Some of the indicators of soil conditions are pH, salinity, cation exchange capacity (CEC), exchangeable sodium percentage (ESP), percent base saturation, nutrients, and metals. Detailed discussion of these chemical characteristics is deferred to Appendix F.

3.5.5 Geology

Geologic formations and discontinuities that might cause unexpected flow patterns of applied wastwater to the groundwater should be identified in the planning stages of a land treatment system. If the underlying rock is fractured or crevassed like limestone, percolating wastewater may shortcircuit to the groundwater, thus receiving less than proper treatment because of reduced residence time in the soil. Similarly, perched water tables above the normal groundwater can result from impermeable or semipermeable layers of rock, clay, or hardpan, thus reducing the effective renovative depth. Permanent groundwater should be distinguished from localized perched groundwater conditions. Both the reason for and the direction of movement of a perched groundwater are important geohydrologic factors of a site.

Geologic discontinuities, such as faults and intrusions, should be evaluated for their effect on groundwater occurrence, influence on quantity, and direction of movement. The USGS and many state geological surveys have completed studies and maps indicating the effects of geologic formations on groundwater occurrence and movement. Water well logs can also provide local, detailed information. A groundwater geologist familiar with local conditions can provide valuable information by identifying geologic features that may affect groundwater movement at a particular site.

3.5.6 Climate

An evaluation of climatic factors, such as precipitation, evapotranspiration, temperature, and wind, is used in the determination of the (1) water balance, (2) length of the growing season, (3) number of days when the system cannot be operated, (4) the storage capacity requirement, and (5) the amount of stormwater runoff to be expected.

3.5.6.1 Climatic Data and Its Use

Sufficient climatic data are generally available for most locations from three publications of the National Oceanic and Atmospheric Administration (NOAA - formerly the U.S. Weather Bureau).

The Monthly Summary of Climatic Data provides basic data, such as total precipitation, maximum and minimum temperatures, and relative humidity, for each day of the month for every weather station in a given area. Evaporation data are also given where available.

The Climatic Summary of the United States provides 10 year summaries of data for the same stations in the same given areas. This form of the data is convenient for use in most of the evaluations that must be made and includes:

- Total precipitation for each month of the 10 year period
- Total snowfall for each month of the period
- Mean number of days with precipitation exceeding 0.10 and 0.50 in. (0.25 and 1.3 cm) for each month
- Mean temperature for each month of the period
- Mean daily maximum and minimum temperatures for each month
- Mean number of days per month with temperature less than or equal to 32°F (0°C), and greater than or equal to 90°F (32.5°C)

Local Climatological Data, an annual summary with comparative data, is published for a relatively small number of major weather stations. Among the most useful data contained in the publication are the normals, means, and extremes which are based on all data for that station, on record to date. To use such data, correlation may be required with a station reasonably close to the site.

Climatic data should be subjected to a frequency analysis to determine the expected worst conditions for a given return period. The data analyses are summarized in Table 3-9.

3.5.6.2 Climatic Considerations for Crops

The consumptive use by plants is in direct relation to the climate of the area. Consumptive use or evapotranspiration is the total water used in transpiration, stored in plant tissue, and evaporated from adjacent soil [17]. The consumptive use varies with the type of crop, humidity, air temperature, length of growing season, and wind velocity. The amount of water lost by evapotranspiration can be estimated from the pan evaporation data supplied by NOAA in the vicinity of the site or from theoretical methods (see Appendix F).

TABLE 3-9

SUMMARY OF CLIMATIC ANALYSES

Factor	Data required	Analysis	Use
Precipitation	Annual average, maximum, minimum	Frequency analysis, in./yr	Water balance
Rainfall storm	Intensity, duration	Frequency analysis, in./d	Runoff estimate
Temperature	Days with average below freezing	Frost free period, d	Storage, treatment efficiency, crop growing season
Wind	Velocity and direction	--	Cessation of sprinkling

1 in. = 2.54 cm

The length of the growing season affects the amount of water used by the crop. The length of the growing season for perennial crops is generally the period beginning when the maximum daily temperature stays above the freezing point for an extended period of days, and continues throughout the season despite later freezes [17]. This period is related to latitude and hours of sunlight as well as to the net flow of energy or radiation into and out of the soil. A limited growing season will require long periods of storage or alternative methods of disposal in winter.

3.5.7 Surface Water Hydrology and Quality

3.5.7.1 Hydrology

Surface water hydrology is of interest in land treatment processes mostly because of the runoff of stormwater. Considerations relating to surface runoff control apply to both slow rate and overland flow. Rapid infiltration processes are designed for no runoff.

The control of stormwater runoff both onto and off a land treatment site must be considered. First, the facilities constructed as part of the treatment system must be protected against erosion and washout from extreme storm events. For example, where earthen ditches and/or terraces are used, erosion control from stormwater runoff must be provided. The degree of control of runoff to prevent the destruction of the physical system should be based on the economics of replacing equipment and structures. There is no standard extreme storm event in the design of drainage and runoff collection systems, although a 10 year return event is suggested as a minimum.

3.5.7.2 Quality

The need to control surface runoff resulting from stormwater depends mainly on the expected quality of the runoff relative to the normal discharge requirements to a local body of water. Runoff quality resulting from storms at land treatment sites is essentially unknown for most constituents. However, to give some perspective to the magnitude of nitrogen and phosphorus concentrations in runoff from various agricultural and rural areas, and as an approach to solving the problem, selected data from agricultural stormwater runoff studies are given in Table 3-10.

It is important to note that the research work reported in Table 3-10 was aimed primarily at fertilizing practice and cultivation versus noncultivation as related to nutrient losses. Nevertheless, these data suggest that it is advisable to provide some form of sediment removal at land treatment sites before allowing the remaining runoff water to escape. Based on the experimental work in Wisconsin [21], this would greatly reduce the nutrient losses from the site. Methods used to minimize sediment and nutrient loss include (1) contour planting versus straight-row planting, and (2) incorporation of plant residues to increase organic matter in the soil. In each research study, many additional factors that affect erosion losses were presented, and the interested reader should consult the literature.

More recently, Loehr [22] has compiled runoff quality data from various nonpoint sources. Ranges of values for concentrations of constituents in agricultural runoff resulting from precipitation and the potential yield per unit area of these constituents are listed in Table 3-11.

Runoff quality estimates derived from data in Table 3-11 are to be considered preliminary in nature because of variations in sampling methods, analytical methods, field conditions, and meteorological constraints. The order of magnitude of the characteristics and the differences between sources are more significant than the values. Adherence to established agricultural practices for erosion control and environmental protection will limit adverse runoff impacts.

TABLE 3-10

AVERAGE VALUES OF NITROGEN AND PHOSPHORUS MEASURED IN AGRICULTURAL STORMWATER RUNOFF STUDIES

Location and site description	Management practice	Total nitrogen, mg/L	Total phosphorus, mg/L
North Carolina [18]	Heavily fertilized, uncultivated	4.60 4.60	0.10 0.10
	Lightly fertilized, uncultivated	1.60	0.08
Ithaca, N.Y. (corn, beans, wheat) [19]	Highly fertilized	6.17[a]	0.26[b]
	Moderately fertilized	1.70[a]	0.12
Ontario (marsh) [20]	Fertilized and cultivated	1.88[c]	0.67
	Unfertilized and uncultivated	0.05[c]	0.17
Wisconsin (pilot plots, oat stubble) [21]	Fertilized plowed surface		
	1. In sediment	81.8[d]	0.88
	2. In water	2.8	0.49
		84.6	1.37
	Unfertilized plowed surface		
	1. In sediment	75.2	0.33
	2. In water	0.7	0.1
		75.9	0.43

a. Ammonia plus nitrate nitrogen only.

b. Inorganic phosphorus only.

c. Nitrate plus nitrite nitrogen only.

d. Organic nitrogen from soil sediment accounted for 90+% of all nitrogen. Runoff occurred from 1 h of rain at 2.5 in./h, 24 h after a similar rain event.

1 in./h = 2.54 cm/h

3.5.8 Groundwater Hydrology and Quality

Collection and analysis of available data on groundwater hydrology and quality are essential to planning and feasibility studies. Desirable information includes soil surveys, geologic and groundwater resources surveys, well drilling logs, groundwater level measurements, and chemical analysis of the groundwater. Numerous federal, state, county, and city agencies have this type of information as well as universities, professional and technical societies, and private concerns with groundwater-related interests. Particularly good sources are the USGS at the federal level, state water resources departments, and county water conservation and flood control districts.

TABLE 3-11

SUMMARY OF AGRICULTURAL NONPOINT SOURCES CHARACTERISTICS [22]

Source	Concentration, mg/L				Area yield rate, lb/acre·yr			
	BOD	NO_3-N	Total N	Total P	BOD	NO_3-N	Total N	Total P
Preciptation	12-13	0.14-1.1	1.2-0.04	0.02-0.04	.....	1.3-3.7	5-9	0.04-0.05
Forested land	....	0.1-1.3	0.3-1.8	0.01-0.11	.....	0.6-7.9	3-12	0.03-0.8
Rangeland	....	....	....	....	.....	0.6	....	0.07
Agricultural cropland	7	0.4	9	0.02-1.7	.....	....	0.1-12	0.05-2.6
Land receiving manure	....	....	....	....	.....	....	3.6-12	0.7-2.6
Irrigation tile drainage, western United States								
Surface flow	....	0.4-1.5	0.6-2.2	0.2-0.4	.....	....	3-24	0.9-4.0
Subsurface drainage	....	1.8-19	2.1-19	0.1-0.3	.....	74	38-166	3-9
Cropland tile drainage	....	....	10-25	0.02-0.7	.....	....	0.3-12	0.009-0.3
Seepage from stacked manure	10 300-13 800	....	1 800-2 350	190-280	.....	....	....	....
Feedlot runoff	1 000-11 000	10-23	920-2 100	290-360	1 390	....	890-1 430	9-550

Note: Data do not reflect the extreme ranges caused by improper waste management or extreme storm conditions.

1 lb/acre·yr = 1.12 kg/ha·yr

3.5.8.1 Hydrology

A knowledge of the regional groundwater conditions is particularly important for potential rapid infiltration and slow rate sites. Overland flow will not usually require an extensive hydrogeologic investigation. Sufficient removal of pollutants in the applied wastewater before reaching a permanent groundwater resource is the primary concern. The depth to groundwater and its seasonal fluctuation are a measure of the aeration zone and the degree of renovation that will take place.

When several layers of stratified groundwater underlie a particular site, the occurrence of the vertical leakage between layers should be evaluated. Direction and rate of groundwater flow and aquifer permeability together with groundwater depth are useful in predicting the effect of applied wastewater on the groundwater regime. The extent of recharge mounding, interconnection of aquifers, perched water tables, the potential for surfacing groundwater, and the design of monitoring and withdrawal wells are dependent on groundwater flow data.

Much of the data required for groundwater evaluation may be determined through use of existing wells. Wells that could be used for monitoring should be listed and their relative location described. Historical data on quality, water levels, and quantities pumped from the operation of existing wells may be of value. Such data include seasonal groundwater-level variations, as well as variations over a period of years. The USGS maintains a network of about 15 800 observation wells to monitor water levels nationwide. Records of about 3 500 of these wells are published in Water-Supply Paper Series, "Groundwater - Levels in the United States." Many local, regional, and state agencies compile drillers' boring logs that are also valuable for defining groundwater hydrology.

3.5.8.2 Groundwater Quality

Land treatment of wastewater can provide an alternative to discharge of conventionally treated wastewater. However, the adverse impact of percolated wastewater on the quality of the groundwater must also be considered. Existing groundwater quality should be determined and compared to quality standards for its current or intended use. Groundwater classifications are discussed in Section 5.1.1. The expected quality of the renovated wastewater can then be compared to determine which constituents in the renovated water might be limiting. The USGS "Groundwater Data Network" monitors water quality in observation wells across the country. In addition, the USGS undertakes project investigations or areal groundwater studies in cooperation with local, state, or other federal agencies to appraise groundwater quality. Such reports may provide a large part of the needed groundwater data.

3.6 Other Planning Considerations

Land treatment systems make use of existing natural conditions; therefore, a thorough knowledge of all aspects of any given site is necessary for a successful design. Most features common to all sites or projects have been discussed briefly in the preceding sections. There are also governmental features or planning factors that may be indirectly related to land treatment studies. Some of these factors are presented in this section, including:

- Water rights
- Governmental programs
- Land use

- Environmental setting
- Social and economic aspects

3.6.1 Water Rights

On the basis of water rights considerations, the implementation of a land treatment system may involve a change in water use from nonconsumptive (passing flow through a treatment plant with subsequent discharge) to consumptive. This change can interfere with the water rights of downstream or senior claims to the water as the source of flow is depleted when the discharge is not returned to its original channel [23].

Water rights problems tend to arise in water-short or fully allocated areas, yet the existence of a market for reclaimed water in these areas will aid in the cost effectiveness and acceptability of land treatment. On a national level, these areas are shown in Figure 3-8.

Most riparian (land ownership) rights are in effect east of the Mississippi River, and most appropriative (permit system) rights are in effect west of the Mississippi River, as shown in Figure 3-8 [24]. A legal distinction is made between discharges to a receiving water in a well-defined channel or basin (natural watercourse), superficial waters not in a channel or basin (surface waters), and underground waters not in a well-defined channel or basin (percolating or groundwaters) [24]. A guide in determining whether certain land treatment alternatives may involve water rights problems is presented in Table 3-12. The intention here is not to imply that some alternatives will have problems and others will not, but merely to guide the planner or engineer through the preliminary screening of alternatives.

3.6.1.1 Riparian Rights

According to the Riparian Doctrine, anyone owning land adjacent to, or underlying, a natural watercourse has the right to use, but not consume, the water. Within this theory have arisen two subtheories ("natural flow allocation" and "reasonable use") that affect the manner in which a riparian right can be executed. In natural flow, the landowner can diminish neither the quantity nor the quality of the water before returning it to the watercourse. Beyond minimum consumptive uses, such as drinking, bathing, or cooking, this right is very restrictive, and it gave rise to the reasonable use theory. Water under natural flow can be withdrawn for a "natural," riparian, or nonriparian use. Reasonable use requires that the water be used for a legal and beneficial purpose. Because the water right under riparian theory is closely aligned with the concept of land ownership, the rights to water ownership pass with sale of the land [25].

FIGURE 3-8

DOMINANT WATER RIGHTS DOCTRINES AND AREAS OF WATER SURPLUS OR DEFICIENCY [24]

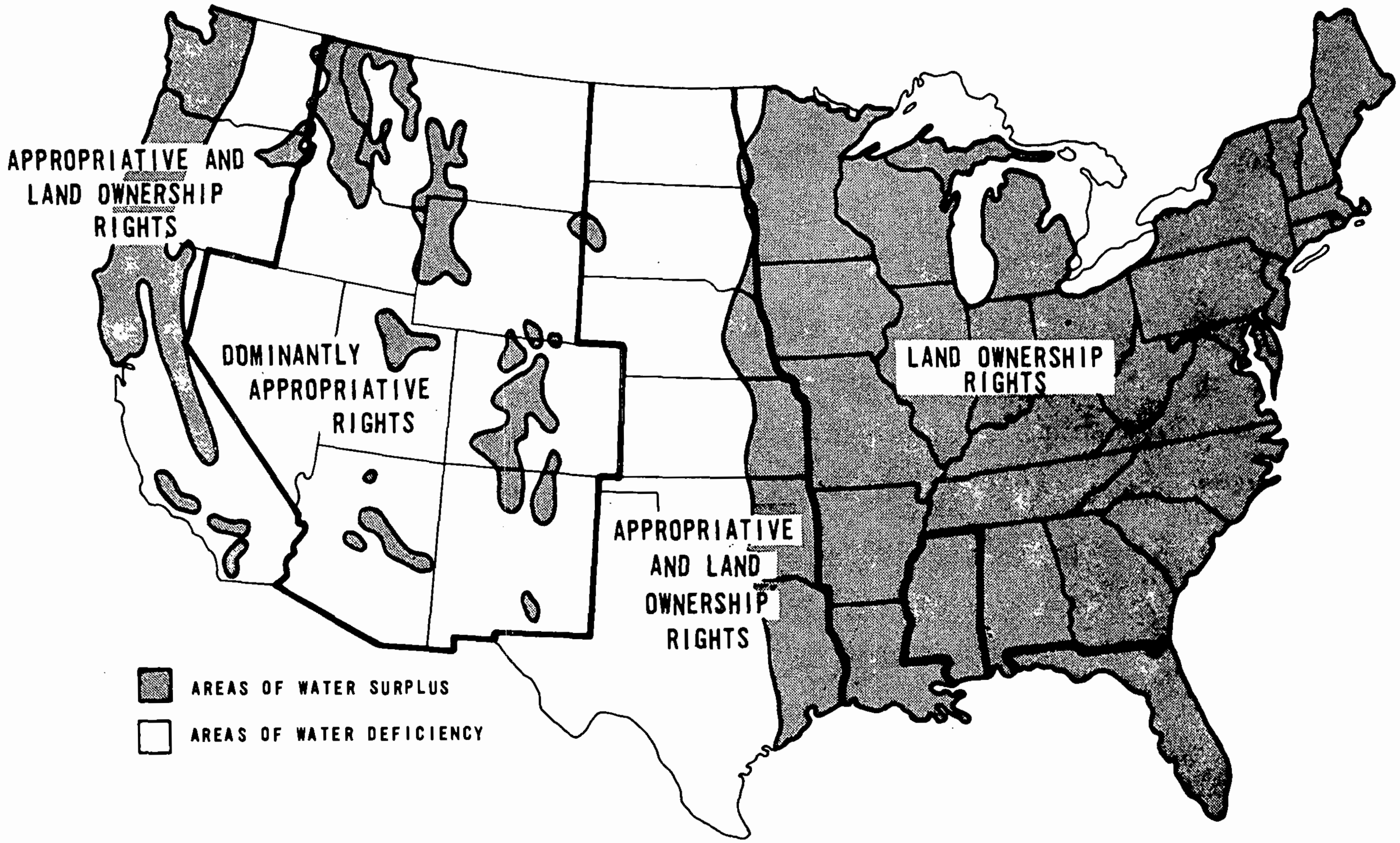


TABLE 3-12

POTENTIAL WATER RIGHTS PROBLEMS FOR LAND TREATMENT ALTERNATIVES[a]

Water definition and water rights theory	Land treatment process		
	Slow rate	Rapid infiltration	Overland flow
Natural watercourses			
Riparian	Unlikely	Unlikely	Unlikely
Appropriative	Likely[b]	Likely[b]	Depends on location of discharge from collection ditch
Combination	Likely[b]	Likely[b]	Depends on location of discharge from collection ditch
Surface waters			
Riparian	Unlikely	Unlikely	Likely[c]
Appropriative	Unlikely	Unlikely	Likely[c]
Combination	Unlikely	Unlikely	Likely[c]
Percolating or groundwaters			
Riparian	Unlikely	Possible	Unlikely
Appropriative	Likely	Likely	Unlikely
Combination	Likely	Likely	Unlikely

a. For existing conditions and alternatives formulation stage of the planning process only. It is also assumed that the appropriative situations are water-short or over-appropriated.

b. If effluent was formerly discharged to stream.

c. If collection/discharge ditch crosses other properties to natural watercourse.

3.6.1.2 Appropriative Rights

Appropriative rights tended to be enacted by statute and defined in the courts on a case-by-case basis. As a result, wide variations exist among the 19 western states that recognize such rights. In general, the basic principles of appropriative rights theory are: (1) first in time, first in right for the water, and (2) subsequent appropriations cannot diminish the quantity or quality of a senior right. Usually, permits are required to establish the right to appropriative water, and the water thus appropriated must also be put to a beneficial use. Rights to appropriated water are not connected with land ownership. They may be bought, sold, exchanged, or transferred wholly or in part [26].

3.6.1.3 Combination Rights

Many states recognize a combination of riparian and appropriative rights. This dual-rights system has developed in states that have water-short and water-surplus areas within their borders. In such cases, the appropriative theory is usually the predominant one [24].

3.6.1.4 Types of Waters

For legal purposes, states have divided waters into three types: natural watercourses, surface water, and percolating (groundwaters). These classifications are arbitrary and are not based on any scientific or empirical rating system but their definition does affect the type of legal problems that may be encountered in land treatment.

3.6.1.4.1 Natural Watercourse

A natural watercourse is one in which water flows in a defined channel either on or below the earth's surface. This definition includes lakes and estuaries and intermittent as well as perennial streams.

The major legal problem that could be expected in both riparian and appropriative states would involve the diversion of what was a direct discharge with the subsequent reduction in flow to the natural watercourse. If the watercourse in question is near or at over-appropriation, junior water users who feel that a reduction in flow may impair their reasonable use of the water may seek administrative or judicial relief.

In a riparian state, the diversion of a discharge that was not originally a part of a stream should not be cause for legal action by downstream users under natural flow theory.

For appropriative rights states, the risk of legal action against the diversion is easier to analyze. If the conditions of the stream are such that the diversion would threaten the quantity or quality of the appropriated water of a downstream user, the damaged party has cause for legal action against the diverter. This action may be injunctive, in which the diverter is prevented from affecting the diversion, or monetary, in which the diverter would be required to compensate for damages caused by his diversion. If the stream in dispute is not already over-appropriated (as is the case in many western streams), or if the area is not water short, it is unlikely that damages could be proved as a result of the diversion.

3.6.1.4.2 Surface Water

A surface water is the legal term for water not contained in a well-defined basin or channel, i.e., rainfall or snowmelt directly on a parcel of land. Such waters belong to the landowner, but he cannot collect and discharge them across adjoining properties without the consent of the owners of those lands. For surface water rights, there is little difference between riparian and appropriative states.

If any of the land treatment alternatives being considered by the planner or engineer require that the renovated water cross another's property, the granting of a drainage or utility easement across the land to the natural watercourse or final user is a necessity in all cases. The cost of such an easement must be considered in the cost-effectiveness analysis.

3.6.1.4.3 Percolating Waters (Groundwaters)

Problems with water rights could arise from two areas: (1) the rise in groundwater caused by the land treatment method may damage adjoining lands, or there may be some interference with the subsurface flow patterns; and (2) if trace contaminants appear in wells of other water rights holders, they may perceive a damage as a result of altered water quality.

In riparian states, the claim of damages would require that a landowner prove that he overlies the same source of the groundwater as the owner/operator. If the alleged damages are not caused by negligent operation of the treatment site, or in a way that is deliberately harmful to the adjoining landowners, it is doubtful that they have sufficient cause for legal action.

For appropriative theory states, the question of an increase in the level or volume of a groundwater should cause no problems because no one's appropriative right would be threatened.

3.6.1.5 Other Water Rights Considerations

In some states, basin authorities or water/irrigation districts have regulations against the transfer of water outside their jurisdictional boundaries. In the western states particularly, the right to divert or use water does not carry with it the right to store such water.

The right to water salvaged from imported water that has run off irrigated lands is also not automatic. The rights in both cases must be specifically obtained or at least must be assured by precedent legal action.

3.6.1.6 Sources of Information

The data contained in this section may be sufficient for a small system, but for larger systems and in problem areas, the watermaster or water rights engineer at the state or local level should be consulted. Some states either have no records or carry unenforceable rights in their records [27], so that further investigation will be necessary if doubt remains. An excellent reference is the National Water Commission publication, A Summary-Digest of State Water Laws available from the Commission [28]. Although summaries of precedent rulings are not guarantees, they may clarify the situation if similar cases can be found [23, 24, 27, 29, 30]. Lastly, if problems arise, the assistance of a water rights attorney is warranted.

3.6.1.7 Resolving Water Rights Problems

To resolve water rights problems, the planner should first attempt to define the water rights setting that could affect the fate of any renovated water and then be aware of the quantity and priority of all rights in the district or basin. The next step is to define the water rights constraints for all alternatives. Once the candidate systems have been selected, the point of discharge, availability and quantity of discharge, and modifications to existing practices should be examined. If problems are likely with any of the feasible alternatives, a water rights attorney should be consulted to define more closely the legal constraints on the alternatives and to define the owner/operator's rights and responsibilities. If the owner's rights to the renovated water can be established, he can now trade those rights with any potentially damaged senior rights or use the revenues from sale of the water to offset possible damage claims.

3.6.2 Governmental Programs

The most important federal programs that should be considered in land treatment, in addition to the EPA Construction Grants Program, are the Soil Conservation Service, Bureau of Reclamation, and U.S. Army Corps of Engineers with their reclamation/irrigation programs being of greatest interest. However, despite the national policy of wastewater reclamation [31] and the National Water Commission's recommendation to exchange sewage effluent with potable water now being used for irrigation, previously subsidized water resources programs often result in such low water prices that renovated wastewater cannot be competitively marketed [27].

In western states, reclamation/irrigation projects are presently financed by interest-free loans to farmers or irrigation districts and can be repaid in 40 years with the first payment due 10 years from project completion for a 50 year total payback period. In eastern states, up to 50% of the cost of supplying irrigation water is borne by the federal government; the remainder is repaid over 40 years at low interest (currently around 5%) [27].

In cases where treated wastewater reuse and sale is desired, the potential markets for irrigation sales and industrial cooling or process water should be evaluated. If the irrigation reuse is not able to compete with existing federal programs, potential industrial users should be contacted. They may be interested because they are not eligible for federally subsidized water projects, and may be prevented from expanding or relocating because of a lack of usable water.

3.6.3 Land Use

The planner should be cognizant of the full spectrum of land uses in the study area. Further, he must be aware of the community goals and objectives expressed by the proposed distribution of land use in the area's general plan. With this knowledge, the planner can develop the opportunities for land treatment sites that will help achieve these land use goals and objectives. Further, the site location, type of system, and related facilities can be planned to optimize conformance to the proposed environmental and social setting.

As a general guide, the type of land uses that are encountered are residential, commercial, industrial, recreational, urban open space, agricultural, wilderness, and greenbelt preserves. In urban areas, residential, commercial, and industrial uses are the most difficult to develop compatible plans for, whereas recreational and urban open space uses are the easiest. Agricultural, wilderness, and greenbelt preserve uses are most easily incorporated into land treatment site planning [32].

A variety of data sources may be used to evaluate present and planned land uses for the study area. Most city, county, and regional planning agencies have land use plans that indicate present land use policy. Often, the plans for future land use are current, but actual land use is out of date. In this case, satellite earth-imagery photographs may be helpful. By using LANDSAT (Land Satellite) or ERTS (Earth Resources Technology Satellite) photographs, not only present land uses but also a number of very useful physical phenomena, such as the extent of the flood plain, location of unmapped faults, and point sources of pollution, can often be determined [33]. Although the techniques for photointerpretation are a subject beyond the scope of this manual, true color, false color infrared, and color infrared prints of the study area as obtained from the USGS, can provide valuable, up-to-date information [33].

When completed, the Land Use Data and Analysis (LUDA) Program of the USGS will be an invaluable planning tool. LUDA will provide a comprehensive collection and analysis of land use and land cover data on a nationwide basis. Individual land use/cover maps will be released following compilation. Periodic revision of the data is planned.

Once the land uses have been identified, the study area should be divided into population density areas for comparison with the land uses. The preferred sites tend to lie in areas that have the lowest population densities (5 persons per acre or less) [32]. This will have the positive side effect of minimizing the number of relocations (with their attendant costs and legal problems) that may be required. Those sites with the lowest population density and with compatible land use should be ranked high in the evaluation process for preliminary screening.

The zoning for each candidate site should be checked. Zoning laws are the means by which a community maintains local control over what kinds of land uses are allowed. They are also the means by which the tax assessment rates are set [34]. If a site appears to be excellent in all other respects but zoning conflicts exist, use permits or waivers may be obtained through the agency having zoning authority.

In addition to minimum population density, the size of land parcels in the study area will strongly affect the final site selection. The fewest number of land parcels needed to develop a site will result in the least number of property acquisitions or lease contracts and the relocation of the least number of families. Assessors plats are the usual source of this type of information.

3.6.4 Environmental Setting

Most public projects require an assessment of their impacts to the environment. Although the environmental impact statement (EIS) procedure is lengthy and described in numerous sources, a brief description of certain key topics is presented.

3.6.4.1 Vegetation and Wildlife

The important relationships are between the ecological communities. Once these are defined as closely as possible, the task of evaluating how the overall ecosystem may adjust to project-created stress becomes easier to accomplish, and the results are easier to relate to decision-makers [35].

If the interrelationships of the various plant and animal ecosystems cannot be defined sufficiently to evaluate the stress, the following information, as a minimum, should be obtained:

- The habitats of rare or endangered species [36]
- Locations of unique or rare native ecological communities [36]
- Preferred routes of migratory animals or birds
- Locations of feeding, watering, nesting, and mating areas--especially of those animals that have a low tolerance for human activity
- Areas whose ecosystems would be substantially altered by periodically applied water or a raised groundwater table
- Plant communities with high water tolerance to the land treatment alternatives

Some of the needed data may be available in the community or regional land use or comprehensive plans. Other excellent sources are state fish and game departments or the U.S. Bureau of Sports Fisheries and Wildlife. Colleges and universities usually have data on the flora and fauna of a region in their biology and zoology departments. Conservation groups, such as the Sierra Club, Audubon Society, Isaac Walton League, and Ducks Unlimited, either have access to these data or know where they can be obtained. Many communities have a naturalist who has intimate knowledge of unrecorded data. If possible, these people should be consulted before and during the definition of the vegetation and wildlife setting.

In the evaluation of the sites for the initial and final screenings, vegetation and wildlife considerations can be significant. Encroachment on the habitats of rare, endangered, or threatened species could eliminate the site from further consideration. If an entire study area has been designated as a potential habitat, a field survey is required for direct observations by qualified biologists/zoologists. In the absence of direct observations, these professionals can usually render judgments on the possibility of the species being found at the various sites.

3.6.4.2 Historical and Archaeological Sites

Because land treatment systems involve large areas of land, the possibility of encountering an historical or archaeological site within the project study area must be carefully considered. Pursuant to the National Historic Preservation Act (PL 89-655) of 1966, many states have begun programs of indentifying historic or archaeologic features or structures. Some states have also developed purchase and preservation programs [37]. Reports on the plans are excellent sources of data for regional considerations. Other data can be found in local universities or college history or geology departments. Aid should be solicited from the local historical or archaeological organizations and their individual members.

3.6.5 Social and Economic Aspects

The social and economic aspects, including relocation, aesthetics, and general public acceptability, are the most difficult for the project planner/engineer to define and evaluate. Gathering factual and statistical data about the study area will be one of the first tasks. One excellent source is the Census Bureau. Also, the Economic Development Administration may provide community economic profile reports. Regional and local planning authorities have generally compiled data for land use, recreation, and employment/population projections. The best sources, however, will be the public advisory group and the feedback obtained at the public participation workshops and the required public hearings [35, 36, 38, 39, 40].

If substantial purchase of land is proposed, relocation may be required of residences, farm buildings, and possibly commercial buildings. Relocation has both social and economic impacts and the magnitude must be fully assessed. An additional consideration is the proximity of schools, churches, and cemeteries, for which relocation may not be socially acceptable [41].

What will be the public reaction to land treatment and reuse of renovated water alternatives? Although the recycling of animal wastes is encouraged and accepted, people are more concerned about the application of human wastes to the land. They generally have misgivings about potential public health, odor, property values, and nuisance problems in connection with land treatment, yet these problems should not arise in a well-planned, well-engineered, and well-managed system [34]. The aesthetic effects can be enhanced by proper planning and the use of buffer zones, trees, shrubs, and careful operation to minimize odor potential, uncontrolled growth of weeds, and standing water.

The other aspect of the public acceptability evaluation--reaction to reuse of renovated water--may depend on the contemplated use of that resource. A recent study conducted in 10 southern California communities indicates that the public is ready for large-scale reuse of renovated water for purposes that do not involve body contact uses of renovated water. In this same study, government officials were surveyed nationally, and they rated "public acceptability" lower than the general public rated it in 12 of the 13 potential reuse categories. These officials were generally the most conservative of the four groups surveyed (general public, water resources experts, industry, and government officials). The results are summarized in Figure 3-9 [42].

These poll results should not be considered indicative of the kind of acceptance that may be encountered elsewhere as Southern California has had positive experiences with wastewater reclamation. It was noteworthy that local officials, who deal with the public on a daily basis, rated public acceptability lower than other government officials. However, as reclamation, conservation of resources, and water shortages become more prevalent, public acceptance to wastewater renovation and reuse should improve.

The project planner or engineer should realize that if land treatment and water renovation is unknown in the study area, it may represent a major change in considering wastewater management and a public education program may be necessary. Unless people understand what is proposed and how it can benefit them, any change will be resisted [43]. A public advisory board can aid in the acceptance of the land treatment alternatives. The problem of "representative" members in the advisory board is not a new one. A typical range of interests for the group might include the following:

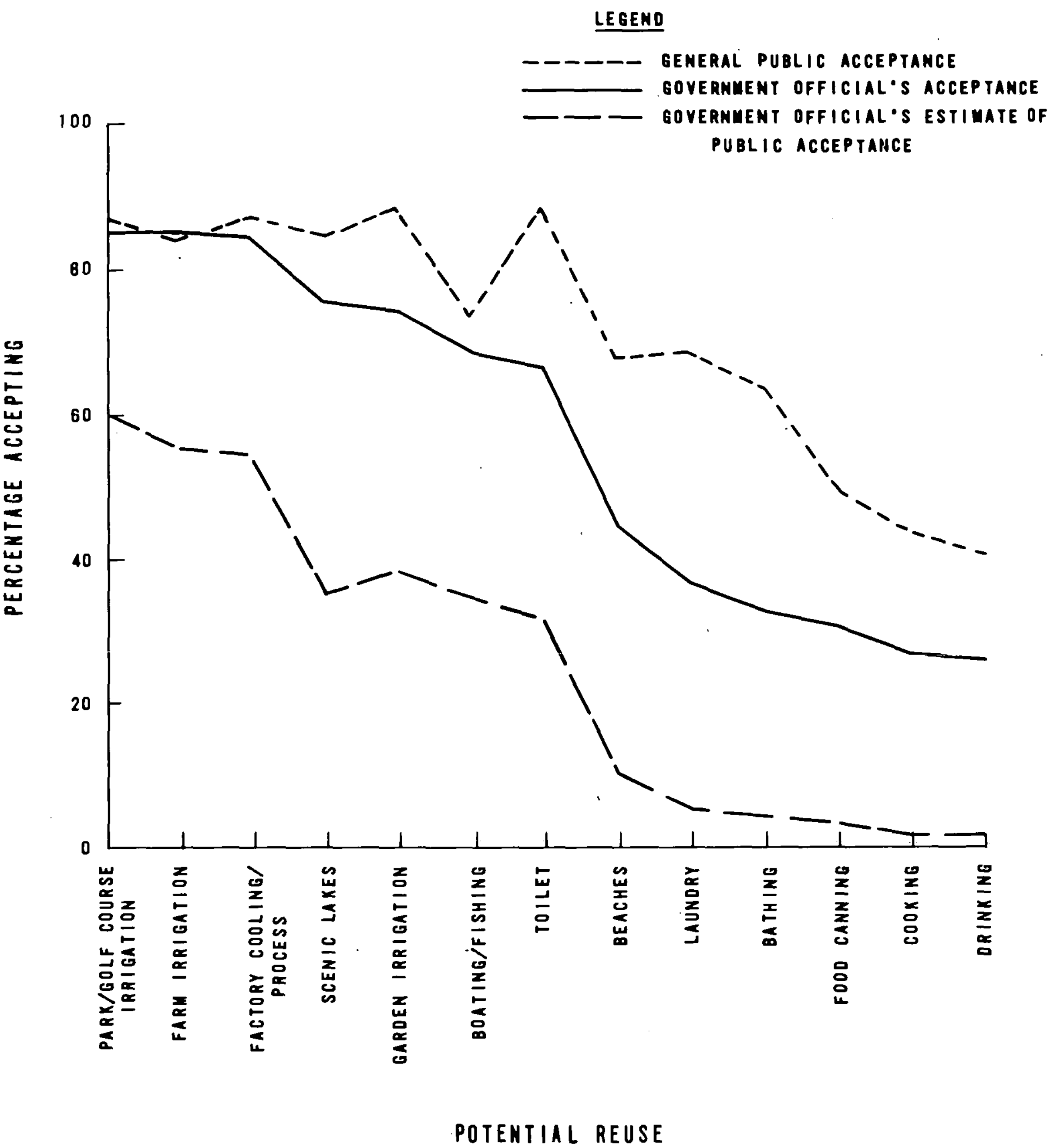
- Farmers representing irrigation districts
- Property owners in areas that have a high potential for system siting
- Civic groups interested in community development
- Conservation groups

There are essentially two types of public participation programs: reactive and participative. In reactive programs, the major events in the planning process (e.g., alternative sites for consideration) are presented to the public. The reactions to the information presented and the remarks of the participants are incoporated into the final screening and selection process [44].

Participative planning differs in the number of meetings and the alternative selection process. A number of public hearings are held in

FIGURE 3-9

PUBLIC ACCEPTABILITY OF RENOVATED WATER REUSE [42]



which the alternatives are presented, and the advantages and disadvantages are listed. The next meeting presents any new alternatives or advantages/disadvantages from the previous meetings. Any rejected alternatives are shown and the reasons for their rejection are outlined. This process is repeated until a final selection is made [38]. The unique involvement of the public at all stages of the alternative development will generate more useful informed feedback and public support [44].

The definition of the social and economic setting and the evaluation of public acceptance will be the result of working within the study area framework and constant interaction between the participants.

3.7 Evaluation of Alternative Systems

The number of alternatives to be evaluated in detail will depend on factors specific to each project, and may involve one or more choices of the treatment process, the site location, or the recovery/reuse options for the renovated water. On the other hand, the topography and soil conditions within a given project area may restrict land treatment to one feasible process, or a very limited number of potential sites may be available. A careful preliminary investigation and screening process is necessary to identify a number of alternatives without sacrificing an objective approach.

For the purposes of this manual, the EPA cost-effectiveness analysis procedures documented in 40 CFR 35, Appendix A, are closely followed [45]. These procedures must be used in selecting municipal wastewater management systems submitted for construction grant funding under PL 92-500. For other planning situations, the EPA document provides a complete evaluation procedure that can be adapted to fit particular objectives. General references on engineering economic evaluations [46] and benefit/cost analysis in water resources planning [47] can provide additional background information for methods of evaluating alternatives. The EPA procedures require an evaluation of both monetary and nonmonetary factors. The most cost-effective alternative is described as follows [45]:

> The most cost-effective alternative shall be the waste treatment management system determined from the analysis to have the lowest present worth and/or equivalent annual value without overriding adverse nonmonetary costs and to realize at least identical minimum benefits in terms of applicable Federal, State, and local standards for effluent quality, water quality, water reuse and/or land and subsurface disposal.

In the following sections, both monetary cost factors and nonmonetary aspects of land treatment systems are discussed. Detailed cost evaluation procedures are not described, but methods of comparing overall costs and nonmonetary factors for land treatment and conventional systems are discussed.

3.7.1 Cost Estimating

Factors that influence both capital and operation and maintenance costs are discussed in the following paragraphs. Only a few cost figures are actually presented, but references are made to specific sources of cost information. Because the cost effectiveness of land treatment is sensitive to land cost, a separate discussion for estimating this item is included. Methods of evaluating revenues and a discussion of tradeoffs that are unique to land treatment cost analysis are also discussed.

3.7.1.1 Capital Costs

Curves for capital costs are available in <u>Costs of Wastewater Treatment by Land Application</u> [48]. The Stage II curves are recommended in conducting cost estimates. Although the base date for these curves was February 1973, they should not be arbitrarily updated by conventional cost indexes. A comparison of unit costs for key items, such as earthwork and continuous-move sprinkling equipment, may provide a more reasonable estimate of the increase in current local prices over the prices of February 1973 [49].

Components that might be used for preapplication treatment include primary sedimentation and aerated lagoons. Their capital costs can be determined from published cost curves for conventional treatment systems [50, 51], recent construction bids, and current price quotations, as necessary. Additional cost estimating data have been published for aerated lagoons because they are commonly used in conjunction with land treatment systems [48, 52]. Costs should include sludge handling as well as liquid processing components.

A checklist of the items requiring a capital cost estimate is provided in Table 3-13. These should be completed for each alternative system.

Salvage values at the end of the planning period for structures and equipment should be based on expected service life. Appendix A of 40 CFR 35 [45] specifies service lives to be used in Section 201 facilities planning (under PL 92-500) as follows:

Land	Permanent
Structures	30 to 50 years
Process equipment	15 to 30 years
Auxiliary equipment	10 to 15 years

TABLE 3-13

CHECKLIST OF CAPITAL COSTS FOR ALTERNATIVE LAND TREATMENT SYSTEMS [48]

Alternative No. Type of system	Average flow _____ mgd Analysis date _______	
	Total cost, $	Amortized cost, \$/yr[a]
Preapplication treatment		
______________________	______	
______________________	______	
Transmission		
______________________	______	
______________________	______	
Storage ______________ Mgal	______	
Field preparation		
______________________	______	
______________________	______	
Recovery		
______________________	______	
______________________	______	
Additional costs		
______________________	______	
______________________	______	
______________________	______	
______________________	______	
SUBTOTAL	______	
Service and interest factor at ______________%	______	
SUBTOTAL	______	
Land[b] at ______________/acre	______	________
TOTAL	______	________

a. Check salvage values, Table 3-14 and preceding text.

b. Section 3.7.1.3.

Additional guidelines for service life of irrigation system components are given in Table 3-14.

TABLE 3-14

SUGGESTED SERVICE LIFE FOR COMPONENTS OF AN IRRIGATION SYSTEM [53][a]

	Service life	
	Hours[b]	Years
Well can casing	......	20
Pump plant housing	......	20
Pump, turbine		
Bowl (about 50% of cost of pump unit)	16,000	8
Column, etc.	32,000	16
Pump, centrifugal	32,000	16
Power transmission		
Gear head	30,000	15
V-belt	6,000	3
Flat belt, rubber and fabric	10,000	5
Flat belt, leather	20,000	10
Power units		
Electric motor	50,000	25
Diesel engine	28,000	14
Gasoline or distillate		
Air cooled	8,000	4
Water cooled	18,000	9
Propane engine	28,000	14
Open farm ditches (permanent)	......	20
Concrete structures	......	20
Concrete pipe systems	......	20
Wood flumes	......	8
Pipe, surface, gated	......	10
Pipe, water works class	......	40
Pipe, steel, coated, underground	......	20
Pipe, aluminum, sprinkler use	......	15
Pipe, steel, coated, surface use only	......	10
Pipe, steel galvanized, surface only	......	15
Pipe, wood buried	......	20
Sprinkler heads	......	8
Solid set sprinkler system	......	20
Center pivot sprinkler system	......	10-14
Side roll traveling system	......	15-20
Traveling gun sprinkler system	......	10
Traveling gun hose system	......	4
Land grading[c]	......	None
Reservoirs[d]	......	None

a. Certain irrigation equipment may have a lesser life when used in a wastewater treatment system.

b. These hours may be used for year-round operation. The comparable period in years was based on a seasonal use of 2 000 h/yr.

c. Some sources depreciate land leveling in 7 to 15 years. However, if proper annual maintenance is practiced, figure only interest on the leveling costs. Use interest on capital invested in water right purchase.

d. Except where silting from watershed above will fill reservoir in an estimated period of years.

3.7.1.2 Annual Operation and Maintenance Costs

Operation and maintenance costs include labor, materials and supplies, and power costs. They may be assumed constant for the planning period though many of the costs will vary throughout the period, particularly those which are flow-dependent, such as power costs for aeration and pumping, and chemical costs. If flows are expected to increase substantially during the planning period, varying operation and maintenance costs should be analyzed on a year-by-year basis (life-cycle cost) or by reducing the total future value of the increasing annual costs to an equivalent annuity amount.

Preapplication treatment will require operation and maintenance labor, materials including chemicals, and power costs. These costs can be determined from cost estimating sources for conventional treatment processes [50, 51, 54]. Additional operation cost data on aerated lagoons can be obtained from other sources [48]. Operation and maintenance costs for the remaining categories can be found in reference [48]. A checklist has also been prepared for operation and maintenance cost estimating purposes and is shown in Table 3-15.

3.7.1.3 Land Costs

3.7.1.3.1 Fee-Simple Purchase

The land category includes the cost of acquiring land for application sites, buffer zones, service roads, storage reservoirs, preapplication treatment facilities, administrative and laboratory buildings, and other miscellaneous facilities. Easements for transmission pipelines may also be included in this category.

Land for preapplication treatment facilities and other permanent structures is usually purchased outright if it is not already under control of the wastewater management agency. Several options are potentially available for acquisition or control of the land used for the treatment process. These include outright purchase (fee-simple acquisition), long-term lease or easement, and purchase with leaseback of the land with no direct involvement in the management of the land. A separate option of simply negotiating contracts with private landowners to sell or deliver wastewater for application would eliminate land acquisition as a capital cost. According to a recent survey, fee-simple land acquisition is preferred by most states, communities, and federal agencies [55].

Purchase of the land provides the highest degree of control over the application sites and ensures uninterrupted land availability for both

TABLE 3-15

CHECKLIST OF ANNUAL OPERATION AND MAINTENANCE COSTS FOR ALTERNATIVE LAND TREATMENT SYSTEMS [48]

Alternative No. ______ Average flow ______ Mgal/d
Type of system ______ Analysis date ______

	Annual cost, $			
	Labor	Power	Material	Total
Preapplication treatment				
______	______	______	______	______
______	______	______	______	______
Transmission				
______	______	______	______	______
______	______	______	______	______
Storage ______ Mgal	______	______	______	______
Distribution				
______	______	______	______	______
______	______	______	______	______
Recovery				
______	______	______	______	______
______	______	______	______	______
Additional costs				
______	______	______	______	______
______	______	______	______	______
______	______	______	______	______
______	______	______	______	______
______	______	______	______	______
Revenues[a]				
______	______	______	______	______
SUBTOTAL	______	______	______	______
Land lease ______				______
TOTAL	______	______	______	______

a. Section 3.7.1.4.

short-term and long-term planning. In many cases, purchase will be more economical than leasing or easements. For this option, land acquisition is treated as a simple capital expenditure.

For projects eligible for PL 92-500 construction grant funding, purchase of land to be used as an integral part of the treatment process is eligible. Purchase and leaseback of land for agricultural or other use involving application of wastewater would require an initial capital expenditure and annual revenues, or negative operation costs, as discussed in a later section.

Assuming that land is purchased, the capital cost is determined simply by multiplying the total area required by the prevailing market value. Methods of estimating the total area required have been discussed in Section 3.3. Because the final alternatives usually include specific sites, the prevailing market value can be estimated from information supplied by a local source, such as the tax assessor's office. In a few cases, the wastewater management agency may already control sufficient land and acquisition is therefore eliminated as a capital cost factor.

The costs of relocating residences and other buildings must be included in the estimate of initial costs and are highly dependent on the location. Agencies such as the U.S. Army Corps of Engineers, U.S. Bureau of Reclamation, and state highway departments can assist in the estimates. For federally funded projects, the acquisition of land and relocation of residents must be conducted in accordance with the Uniform Relocation Assistance and Land Acquisition Policies Act of 1970. In one case, relocation costs for moving approximately 200 familes averaged about $5 000 per family, plus about $300 000 for administration of the program [42].

EPA guidelines require that the salvage value of land be assumed equal to the initial purchase price. Land values may, in fact, appreciate considerably during the planning period, particularly if relatively undeveloped land is purchased initially.

3.7.1.3.2 Leasing

The cost of leasing land for application purposes is included as an operation cost for those alternatives in which fee-simple acquisition is not a viable or an economic option. However, long-term leases are eligible for PL 92-500 construction grant financing, if they can be shown to be more cost-effective.

It has been estimated that leasing/easements will be cost-effective only for several hundred projects nationwide. Most of these projects would be in arid or semiarid areas where effluent has a high value and land has a low value. In these areas, some landowners may be willing to either pay for wastewater effluents, accept wastewater effluents free of charge, or make leasing arrangements at a nominal charge. To be eligible for grant funding, the lease or easement should include the conditions shown in Table 3-16.

TABLE 3-16

REQUIREMENTS FOR LAND LEASING FOR PL 92-500 GRANT FUNDING [56]

- Limit the purpose of the lease or easement to land application and activities incident to land application.
- Describe explicitly the property use desired.
- Waive the landowner's right to restoration of the property at the termination of the lease/easement.
- Recognizing the serious risk of premature lease termination, provide for full recovery of damages by the grantee in such an event with recovery of the paid federal share or, alternatively, retention of the federal share to be used solely for the eligible costs of the expansion or modification of the treatment works associated with the project. The damages would include the difference between the total present worth of treatment works changes resulting from premature termination and the costs resulting from expiration of the lease. The damages would also include any additional losses or costs due to unplanned disruption of wastewater treatment.
- Provide for payment of the lease/easement in a lump sum for the full value of the entire term.
- Provide for leases/easements for a minimum of twenty (20) years, or the useful life of the treatment plant, whichever is longer, with an option of renewal for an additional term, as deemed appropriate.

3.7.1.4 Revenues

Revenues can accrue from crop sales, sale of renovated water, sale of treated effluent for land application, or leaseback of purchased land for farming or other purposes. In the evaluation and comparison of alternatives, revenue estimates can be viewed as offsetting or negative annual costs, but with a higher degree of uncertainty than with estimating capital and operating costs. Crop returns may be anticipated from slow rate processes in which the wastewater management agency controls the land and manages the farming, while overland flow and rapid infiltration processes generally will not produce significant crop revenues. In either case, revenues can be expected to offset only a portion of the total operating cost. Prevailing market values for crops can usually be obtained from state university cooperative extension services, but yield estimates must be made for the proposed conditions of application. These estimates are preliminary and can be based on typical yields for the local area. In a few cases, however, optimization of proposed application rates based on crop yield, revenues, and costs may be investigated during the development of alternatives. Economic models for such a procedure have been published [57, 58].

Relatively little information on crop revenues is available from agencies that actually manage their own farming operations. The most widely reported operation is the one at Muskegon, Michigan (Section 7.6). During 1975, the first full year of operation, total crop revenues amounted to about 44% of the total operating expenses, including a farming management contract fee [59]. The revenues increased an estimated 60% in 1976. The farm operated by San Angelo, Texas, where slow rate application of wastewater is used (Section 7.5), is also reported to be profitable.

For alternatives that propose purchase of land by the wastewater agency and subsequent leaseback to farmers with an agreement to use wastewater for application, a second source of income, is the estimated lease payment. In Bakersfield, California, this type of arrangement brings revenues to the city that are approximately 20% of total treatment operating expenses [60].

Another major source of income may be the renovated water recovered from land treatment systems, particularly runoff from overland flow systems or pumped withdrawal following rapid infiltration systems. Possible markets for the renovated water must be investigated on a case-by-case basis. Methods of assessing the relative value of renovated wastewater for various uses and levels of effluent quality are discussed in reference [61]. Potential reuse categories and possible user costs that would have to be borne as a result of using renovated wastewater rather than normal supplies are discussed in a separate study [62].

For some projects, the quality and quantity of renovated water from all alternatives may not be sufficiently different to affect the marketability of the effluent. For those situations, revenues from the sale of renovated water may not be a meaningful evaluative factor for comparison purposes.

3.7.1.5 Cost Tradeoffs

There are many considerations that can improve the cost-effectiveness of an alternative without changing overall treatment performance. Some of the more important tradeoffs that should be considered in analyzing the alternatives are summarized in Table 3-17.

3.7.2 Nonmonetary Considerations

To complete the cost-effective analysis as previously defined, a range of nonmonetary factors should be evaluated for each alternative. This evaluation also serves as a basis for unavoidable adverse impacts of the selected plan and for outlining mitigation measures for these impacts. Nonmonetary factors, as listed in Table 3-18, may be divided into four categories: (1) treatment performance and reliability, (2) environmental impacts, (3) resource commitments, and (4) implementation and legal constraints.

TABLE 3-17

COST TRADEOFF CONSIDERATIONS FOR LAND TREATMENT SYSTEMS[a]

Option A	versus	Option B
• Land leveling for surface flooding		Sprinkler systems
• Cash crop revenues and operating costs		Forage and cover (requires less land)
• High drawdown rates from storage requiring high volume pumps (minimizes storage volume)		Low dradown rates requiring smaller pumps and distribution facilities (requires more land)
• Existing vegetation		Land preparation for high-nitrogen uptake vegetation
• Double cropping		Perennial crops
• One 8 hour daily shift, no weekend application (requires larger pumps, pipes)		Round-the-clock and weekend operation (higher operating cost)
• Automatic systems (high capital)		Nonautomatic systems (high operation and maintenance)

a. The list is intended to show some of the more obvious options. Many other possibilities will arise in the alternative development process.

TABLE 3-18

NONMONETARY FACTORS FOR EVALUATION OF ALTERNATIVES

1. Treatment performance and reliability
    - Ability to meet effluent quality/water quality goals
    - Process reliability and control
    - Process flexibility
2. Environmental impacts
    - Archaeological, historical, geological sites
    - Plant and animal communities
    - Surface and groundwaters
    - Soils
    - Air quality and odors
    - Noise and traffic
    - Public health
    - Land use
    - Social issues
    - Economic issues
    - Secondary (induced-growth) effects
3. Resource commitments
    - Land
    - Energy
    - Chemicals
4. Implementation and legal constraints
    - Implementation authority
    - Water rights
    - Existing regulations and plans

Table 3-18 can serve as a comprehensive guideline for comparing alternatives, but it must be recognized that each planning situation is unique. Some factors may be relatively insignificant in one situation, while others may be critical. The approach used to compare each factor for various alternatives may be selected by the planner/engineer or may be dictated by requirements of the study. For example, the Urban Studies Program specifically discourages the use of numerical ratings in the Impact Assessment and Evaluation Appendix of its reports [35]. The planner/engineer must be aware of particular requirements for evaluating environmental or other factors for a specific type of project.

3.7.2.1 Treatment Performance and Reliability

Alternatives that are not capable of meeting minimum effluent quality or water quality criteria, and those that provide significantly higher quality but at unacceptable cost, will normally be eliminated during the preliminary screening process. Thus, the expected effluent quality from all alternatives may be relatively similar and may not provide a basis for comparison. However, there are some differences in effluent quality from the various land treatment processes, as pointed out in Chapter 2. These differences should be noted when two or more processes of land treatment are being compared. There may also be differences in performance when conventional and land treatment alternatives are compared. For example, a comparison of expected effluent quality from two conventional systems, three land treatment systems, and four advanced wastewater treatment systems is presented in Table 3-19.

Well planned and operated land treatment systems are reliable [64]. Factors that affect the reliability of land treatment systems include climatic conditions, natural disasters, and equipment breakdown. Future resource availability should also be evaluated, particularly when comparing land treatment systems with systems that consume a higher quantity of power and/or chemicals.

The flexibility of any treatment system, and all its components, to adapt to changing conditions should be evaluated. Conditions that might change include effluent quality standards, wastewater characteristics, growth rate or growth beyond the planning period, surrounding land use, and technological advances. Of particular concern in land treatment systems is the future availability of land. Prudent design will avoid situations on which no land is available for future expansion.

3.7.2.2 Environmental Impacts

Information on characterizing various aspects of the environmental setting was presented in Section 3.6. With this background, the primary and secondary impacts of each of the alternative plans may be assessed.

TABLE 3-19

COMPARISON OF EFFLUENT QUALITY FOR CONVENTIONAL, LAND TREATMENT, AND ADVANCED WASTEWATER TREATMENT SYSTEMS [63]

System	Effluent constituent, mg/L					
	BOD	SS	NH_3-N	NO_3-N	Total N	P
Conventional treatment						
Aerated lagoon	35	40	10	20	30	8
Activated sludge	20	25	20	10	30	8
Land treatment						
Slow rate	1	1	0.5	2.5	3	0.1
Overland flow	5	5	0.5	2.5	3	5
Rapid infiltration	5	1	....	10	10	2
Advanced wastewater treatment[a]						
1	12	15	1	29	30	8
2	15	16	....	....	3	8
3	5	5	20	10	30	0.5
4	5	5	....	....	3	0.5

a. The advanced wastewater treatment systems are as follows:

1 = biological nitrification
2 = biological nitrification-denitrification
3 = tertiary, two-stage lime coagulation, and filtration
4 = tertiary, two-stage lime coagulation, filtration, and selective ion exchange

3.7.2.3 Resource Commitments

The use and conservation of resources--land, energy, and chemicals--will be indirectly included in the cost analysis, but the noneconomic impacts should be evaluated as well. The amount of land committed to wastewater treatment and renovation will be larger for land treatment systems than for conventional treatment systems. The extent to which this is a negative or positive impact involves evaluation of several factors discussed in the preceding section, including project land use, and social and economic issues. It must be recognized that the use of the land is necessarily a long-term commitment. However, the land used for an application site is not destroyed or irrevocably altered. When operations cease, it again becomes available for other land uses.

Energy requirements should be compared independently of the cost analysis. Land treatment energy requirements will depend significantly on the distance and elevation required for transmission, as well as on other pumping requirements. Conventional or advanced wastewater treatment processes may require relatively high energy inputs, in part because of the energy required for additional sludge handling and disposal. Relative comparisons of energy requirements for a number of treatment strategies have been published [54, 64].

Chemical requirements should be evaluated primarily on the basis of future availability, which will depend, in part, on the location of the project. If disinfection or supplemental fertilization is needed, chemicals may be needed for a land treatment system. In advanced wastewater treatment, there are many additional processes that require chemicals. Land treatment alternatives involving cultivation and harvesting of crops can be viewed as conserving nutrients, whereas most advanced wastewater treatment methods for nutrient removal tie up or release nutrients in a relatively unusable form.

3.7.3 Plan Selection

The approach taken to summarize and present the results of the evaluation will depend on the specific planning situation. Monetary costs for each alternative should be expressed on the basis of total present worth or equivalent annual cost. Nonmonetary factors should be presented on a numerical scale or expressed in qualitative terms. To the greatest extent possible, the summary should permit comparison of land treatment and conventional treatment systems on an equivalent basis.

The actual selection process may involve the wastewater management agency, the engineer/planner, technical or nontechnical advisory groups, input from citizens or special interest groups, and other interested governmental bodies. The selected alternative is the most cost effective, reliably meets all water quality goals, and does not have overriding nonmonetary impacts.

Once a plan has been selected tentatively, the final step should be to address any adverse impacts associated with the plan that are unavoidable. Mitigating measures should be outlined to ensure at the planning stage that such impacts can be minimized.

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CHAPTER 4

FIELD INVESTIGATIONS

4.1 Introduction

The primary information for a specific site can usually be found in a USDA-SCS county soil survey. Detailed field investigations are often needed, however, to assess the suitability of a site for land treatment. The intent of this chapter is to outline those tests normally conducted for each type of land treatment process, the reasons for their use, and the conclusions that can be reached from the results. The procedures for conducting these tests are discussed elsewhere: infiltration, permeability, and aquifer tests are discussed in Appendix C; and physical and chemical soil tests are discussed in Appendix F. The significance of various wastewater characteristics to land treatment is also presented. The field tests normally associated with land treatment processes are summarized in Table 4-1.

TABLE 4-1

SUMMARY OF FIELD TESTS FOR LAND TREATMENT PROCESSES

	Processes		
Properties	Slow rate (SR)	Rapid infiltration (RI)	Overland flow (OF)
Wastewater constituents	Nitrogen, phosphorus, SAR[a], EC[a], boron	BOD, SS, nitrogen, phosphorus	BOD, SS, nitrogen, phosphorus
Soil physical properties	Depth of profile Texture and structure	Depth of profile Texture and structure	Depth of profile Texture and structure
Soil hydraulic properties	Infiltration rate Subsurface permeability Aquifer tests (optional)	Infiltration rate Subsurface permeability Aquifer tests	Infiltration rate
Soil chemical properties	pH, CEC, exchange-cations (% of CEC), EC[a], metals[b], phosphorus adsorption (optional)	pH, CEC, phosphorus adsorption	pH, CEC, exchange-able cations (% of CEC)

a. May be applicable to arid and semiarid areas.

b. Background levels of metals such as cadmium, copper, or zinc in the soil should be determined if food chain crops are planned.

4.2 Wastewater Characteristics

The wastewater constituents to be characterized for the various land treatment processes will vary with the climate and the discharge quality requirements. For example, for slow rate systems in humid areas the sodium adsorption ratio (SAR) and electrical conductivity (EC) will be less important than they are in arid areas. The discharge quality requirements for surface water will be provided in the discharge permit. The discharge quality requirements for groundwater can include nitrate nitrogen and trace elements, as presented in Section 5.1.1.

For constituents such as BOD, suspended solids, nitrogen, and phosphorus, the concentrations are used to compute the loading rates. These rates can be compared to soil treatment mechanisms as discussed in Section 5.1. For trace elements, the allowable loadings are also discussed in Section 5.1. For inorganic constituents of importance to slow rate systems, guidelines are presented in Table 4-2.

4.3 Soil Physical and Hydraulic Properties

The physical and hydraulic properties of soils are interrelated. For example, a major reason for establishing the depth of the profile and the texture and structure is to determine the hydraulic capacity. Depth of the soil profile above bedrock is also important in assessing wastewater renovation (slow rate and rapid infiltration processes) and in assessing practical limits on earth moving. Interpretation of soil physical and hydraulic properties is presented in Table 4-3.

4.4 Soil Chemical Properties

Chemical properties are of importance in assessing (1) potential treatment efficiency for infiltration systems, (2) need for soil amendments, and (3) baseline levels of any constituents expected to accumulate in the profile and cause long-term problems.

Both chemical and biological treatment mechanisms are affected by soil pH. Chemical removal mechanisms for phosphorus change with pH (Appendix B). Biological activity is reduced as the pH drops below about 5. The effects on plants are presented in Table 4-4.

The cation exchange capacity (CEC) is a measurable indicator of the potential adsorption capacity for trace elements. The percentage of the CEC occupied by exchangeable sodium (ESP) is important to maintenance of soil permeability.

TABLE 4-2

RELATIONSHIP OF POTENTIAL PROBLEMS TO CONCENTRATIONS OF MAJOR INORGANIC CONSTITUENTS IN IRRIGATION WATERS FOR ARID AND SEMIARID CLIMATES [1]

Problem and related constituent	No problem	Increasing problems	Severe
Salinity[a]			
EC of irrigation water, mmhos/cm	<0.75	0.75-3.0	>3.0
Permeability			
EC of irrigation water, mmhos/cm	>0.5	<0.5	<0.2
SAR (sodium adsorption ratio)[b]	<6.0	6.0-9.0	>9.0
Specific ion toxicity[c]			
From root absorption			
Sodium (evaluate by SAR)	<3	3.0-9.0	>9.0
Chloride, meq/L	<4	4.0-10	>10
Chloride, mg/L	<142	142-355	>355
Boron, mg/L	<0.5	0.5-2.0	2.0-10.0
From foliar absorption (sprinklers)[d]			
Sodium, meq/L	<3.0	>3.0	....
Sodium, mg/L	<69	>69	....
Chloride, meq/L	<3.0	>3.0	....
Chloride, mg/L	<106	>106	....
Miscellaneous[e]			
HCO_3, meq/L	<1.5	1.5-8.5	>8.5
HCO_3, mg/L	<90	90-520	>520
pH	Normal range = 6.5-8.4		

Note: Interpretations are based on possible effects of constituents on crops and/or soils. Suggested values are flexible and should be modified when warranted by local experience or special conditions of crop, soil, and method of irrigation.

a. Assuming water for crop plus water needed for leaching requirement will be applied. Crops vary in tolerance to salinity. Electrical conductivity (EC) mmhos/cm x 640 = approximate total dissolved solids (TDS) in mg/L or ppm; mmhos x 1,000 = micromhos.

b. $$SAR = \frac{Na}{\sqrt{\frac{Ca + Mg}{2}}}$$

where Na = sodium; Ca = calcium; Mg = magnesium, in all meq/L.

c. Most tree crops and woody ornamentals are sensitive to sodium and chloride (use values shown). Most annual crops are not sensitive.

d. Leaf areas wet by sprinklers (rotating heads) may show a leaf burn due to sodium or chloride absorption under low-humidity, high-evaporation conditions. (Evaporation increases ion concentration in water films on leaves between rotations of sprinkler heads.)

e. HCO_3 with overhead sprinkler irrigation may cause a white carbonate deposit to form on fruit and leaves.

TABLE 4-3

INTERPRETATION OF SOIL PHYSICAL AND HYDRAULIC PROPERTIES

Depth of soil profile, ft	
<1-2	Suitable for OF[a]
>2-5	Suitable for SR and OF
5-10	Suitable for all processes
Texture and structure	
Fine texture, poor structure	Suitable for OF
Fine texture, well-structured	Suitable for SR and possibly OF
Coarse texture, well-structured	Suitable for SR and RI
Infiltration rate, in./h	
0.2-6	Suitable for SR
>2.0	Suitable for RI
<0.2	Suitable for OF
Subsurface permeability	
Exceeds or equals infiltration rate	Infiltration rate limiting
Less than infiltration rate	May limit application rate

a. Suitable soil depth must be available for shaping of overland flow slopes. Slow rate process using a grass crop may also be suitable.

1 ft = 0.305 m
1 in. = 2.54 cm

For slow rate systems that emphasize agricultural crop production, soil tests will be conducted for the major nutrients--nitrogen, phosphorus, and potassium; boron; gypsum content; and insoluble calcium ($CaCO_3$). While the latter three are most applicable on arid climates, the remainder are applicable to all locations. These tests should be conducted and the results interpreted for both crop production and land application aspects under the supervision of a qualified soil scientist.

4.5 Other Field Investigations

4.5.1 Soil Borings

When field investigations are conducted during the facilities planning stage for assessing the suitability of the site, it may be necessary to conduct soil borings. Existing well logs can provide additional information if the wells are located within a similar geologic formation. Generally the shallow (up to 10 ft [3 m] deep) soils work can be performed using a soil auger (Figure 4-1) or a backhoe. The soil horizons exposed by a backhoe are illustrated in Figure 4-2. For deeper investigations of soils and groundwater, drill rigs can be used (Figure 4-3). The drill holes can be small diameter with 2 to 4 in. (5 to 10 cm) being typical. The soil removed should be logged and notations made for depths at which groundwater and restricting layers to hydraulic movement are encountered.

TABLE 4-4

INTERPRETATION OF SOIL CHEMICAL TESTS

Test result	Interpretation
pH of saturated soil paste	
<4.2	Too acid for most crops to do well
4.2-5.5	Suitable for acid-tolerant crops
5.5-8.4	Suitable for most crops
>8.4	Too alkaline for most crops, indicates a possible sodium problem
CEC, meq/100 g	
1-10	Sandy soils (limited adsorption)
12-20	Silt loam (moderate adsorption)
>20	Clay and organic soils (high adsorption)
Exchangeable cations, % of CEC (desirable range)	
Sodium	≤5
Calcium	60-70
Potassium	5-10
ESP, % of CEC	
<5	Satisfactory
>10	Reduced permeability in fine-textured soils
>20	Reduced permeability in coarse-textured soils
EC_e, mmhos/cm at 25° of saturation extract	
<2	No salinity problems
2-4	Restricts growth of very salt-sensitive crops
4-8	Restricts growth of many crops
8-16	Restricts growth of all but salt-tolerant crops
>16	Only a few very salt-tolerant crops make satisfactory yields

4.5.2 Groundwater

Knowledge of the existing groundwater quality beneath a site can provide information on quality objectives of treated water. As indicated in Section 5.1.1 the determination of the groundwater case (1, 2, or 3) depends on the use and quality of the groundwater. Wells on adjacent land should have similar quality if located within the same aquifer. Some soil borings can also be used as observation wells for system monitoring.

FIGURE 4-1

CLOSED AND OPEN BARREL AUGERS AND TILING SPADE

4.5.3 Vegetation and Topography

Site inspections are necessary to assess the existing vegetation and topography. The plant species growing in an area can be used as an indication of soil characteristics relating to plant growth. They should not be used as the only means of problem assessment. However, if their occurrence is noted, detailed soil investigations should be conducted to assess the extent of the problem. Some plant species and the probable indication of soil characteristics are given in Table 4-5.

The topography should be mapped prior to final design to allow accurate earthwork computations. Both the existing vegetation and topography should be assessed for costs of clearing and field preparation.

FIGURE 4-2

SOIL PROFILE WITH TWO HORIZONS

FIGURE 4-3

TYPICAL DRILL RIG USED FOR SOIL BORINGS

TABLE 4-5

PROBABLE SOIL CHARACTERISTICS INDICATED BY PLANTS [2][a]

Plant species	Probably indicates
Alpine fir	Poorly drained soil, high water table
Spruce	Poorly drained soil, high water table
Cattails	Poorly drained soil, high water table
Sedges	Poorly drained soil, high water table
Willow	Poorly drained soil, high water table
Dogwood	Poorly drained soil, high water table
Needle and thread grass	Light textured, sandy soil
Western wheat grass	Heavy textured, poorly drained soil
Salt grass	Highly saline soil
Mexican fireweed	Highly saline soil
Grease wood	Highly saline soil, sodium problems
Foxtail	Salt, sodium, high water table
Ponderosa pine	Dry soil
Good sage brush	Good and deep soil

a. Primarily for western states. Similar information for other locations can be found in county soil surveys.

4.6 References

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CHAPTER 5

PROCESS DESIGN

5.1 Land Treatment Process Design

The design of a land treatment process does not lend itself to a step-by-step procedure. The two most important determinations in design are: (1) selection of the site and treatment process, and (2) calculation of the required field area. The iterative nature of site and process selection is described in Chapter 3 and the decisions reached there are assumed to be inputs to this chapter. In this chapter, the process design discussion centers on determining the critical loading rates required to calculate the field area.

This chapter is organized into discussions of (1) the process design for slow rate, rapid infiltration, overland flow, and wetlands application; (2) system components such as preapplication treatment (5.2), storage (5.3), distribution (5.4), and effluent recovery (5.5); (3) vegetation selection and agricultural management; (4) system monitoring, and (5) facilities design guidance. The purpose of the chapter is to focus on design aspects unique to land treatment.

Much background detail is provided to familiarize the environmental engineer with land treatment. Practices common to most engineers will not be discussed. Detailed cost data are not provided, but sources for such information are described in Chapter 3.

The process design procedure for land treatment starts with the required final effluent quality. For each process, the critical loading rate (usually hydraulic or nitrogen) is then determined. The loadings and removals of BOD, SS, phosphorus, trace elements, and microorganisms are also discussed as they may be important in estimating effluent quality or the expected life of the selected site. Extensive discussions of the chemistry and microbiology of nitrogen, phosphorus, pathogens, and metals are presented in the appendixes.

5.1.1 Effluent Quality Criteria

As in conventional process design, it is first necessary to determine the quality required for the treated effluent produced by the system as well as the influent wastewater quality. The wastewater quality is discussed in Chapters 3 and 4. The expected treated water quality from slow rate, rapid infiltration, and overland flow was presented in Table 2-3.

Surface discharge of treated water is expected from overland flow and wetlands systems. Surface discharge from slow rate and rapid infiltration systems can result from the installation of underdrains or wells. The quality criteria for surface discharges are established for the particular watercourse by state and federal agencies.

Subsurface discharge consists of percolate from slow and rapid infiltration systems. Because of the clay soils associated with overland flow and wetland systems, little percolating (usually 5 to 20%) of the applied wastewater occurs. There is little concern for this percolate quality because of the reduction in wastewater constituents after passing through fine textured soils.

The EPA criteria for best practicable waste treatment for alternatives using land application include three cases for groundwater discharge [2]. In each case, the constituent concentration is assumed to be measured in the groundwater at the perimeter of the site.

Case 1 - The groundwater can potentially be used for drinking water supply.

The chemical and pesticide levels in Table 5-1 should not be exceeded in the groundwater. If the existing concentration of an individual parameter exceeds the standards, there should be no further increase in the concentration of that parameter resulting from land application of wastewater.

Case 2 - The groundwater is used for drinking water supply.

The same criteria as Case 1 apply and the bacteriological quality criteria from Table 5-1 also apply in cases where the groundwater is used without disinfection.

Case 3 - Uses other than drinking water supply.

1. Groundwater criteria should be established by the Regional Administrator based on the present or potential use of the groundwater.

The Regional Administrator in conjunction with the appropriate state officials and the grantee shall determine on a site-by-site basis the areas in the vicinity of a specific land application site where the criteria in Case 1, 2, and 3 shall apply. Specifically determined shall be the monitoring requirements appropriate for the project site. This determination shall be made with the objective of protecting the groundwater for use as a drinking water supply and/or other designated uses as appropriate and preventing irrevocable damage to groundwater. Requirements shall include provisions for monitoring the effect on the native groundwater.

Having established the effluent quality requirements for a surface discharge and for the appropriate class of groundwater, the process selection can be made or confirmed. The next step is to determine the needed loading rates to achieve the requirements.

TABLE 5-1

EPA-PROPOSED REGULATIONS ON INTERIM PRIMARY DRINKING WATER STANDARDS, 1975 [2]

Constituent or characteristic	Value	Reason for standard
Physical		
Turbidity, units	1[b]	Aesthetic
Chemical, mg/L		
Arsenic	0.05	Health
Barium	1.0	Health
Cadmium	0.01	Health
Chromium	0.05	Health
Fluoride	1.4-2.4[c]	Health
Lead	0.05	Health
Mercury	0.002	Health
Nitrates as N	10	Health
Selenium	0.01	Health
Silver	0.05	Cosmetic
Bacteriological		
Total coliform, per 100 mL	1	Disease
Pesticides, mg/L		
Endrin	0.0002	Health
Lindane	0.004	Health
Methoxychlor	0.1	Health
Toxaphene	0.005	Health
2,4-D	0.1	Health
2,4,5-TP	0.01	Health

a. The latest revisions to the constituents and concentrations should be used.

b. Five mg/L of suspended solids may be substituted if it can be demonstrated that it does not interfere with disinfection.

c. Dependent on temperature; higher limits for lower temperatures.

5.1.2 Slow Rate Process

The design procedure for the slow rate process is iterative (see Figure 5-1). The field area is first calculated based on hydraulic loading rates and the wastewater flow to be treated. The area is then calculated from the nitrogen loading rate which is determined using a nitrogen balance. The larger area is used in design. Both the acceptable hydraulic and nitrogen loadings depend in part on the vegetation selected (see Section 5.6.1). The discussion of hydraulic and nitrogen loading rates is followed by discussions of removals of BOD and suspended solids, phosphorus, trace elements, and microorganisms.

5.1.2.1 Hydraulic Loading Rates

The hydraulic loading will be limiting in situations where slow permeability soils are used, or nitrogen limits are not critical. The hydraulic loading rates for the design must be within the soil capabilities, as estimated (Figure 3-3) or measured (Chapter 4 and Appendix C).

The hydraulic loading is based on a water balance that includes precipitation, infiltration rate, evapotranspiration (or consumptive use by plants), soil storage capabilities, and subsoil permeability. Generally, the total monthly application should be distributed uniformly, but considerations must be made for planting, harvesting, drying, and other nonapplication periods. The application rate must then be balanced as shown in Equation 5-1.

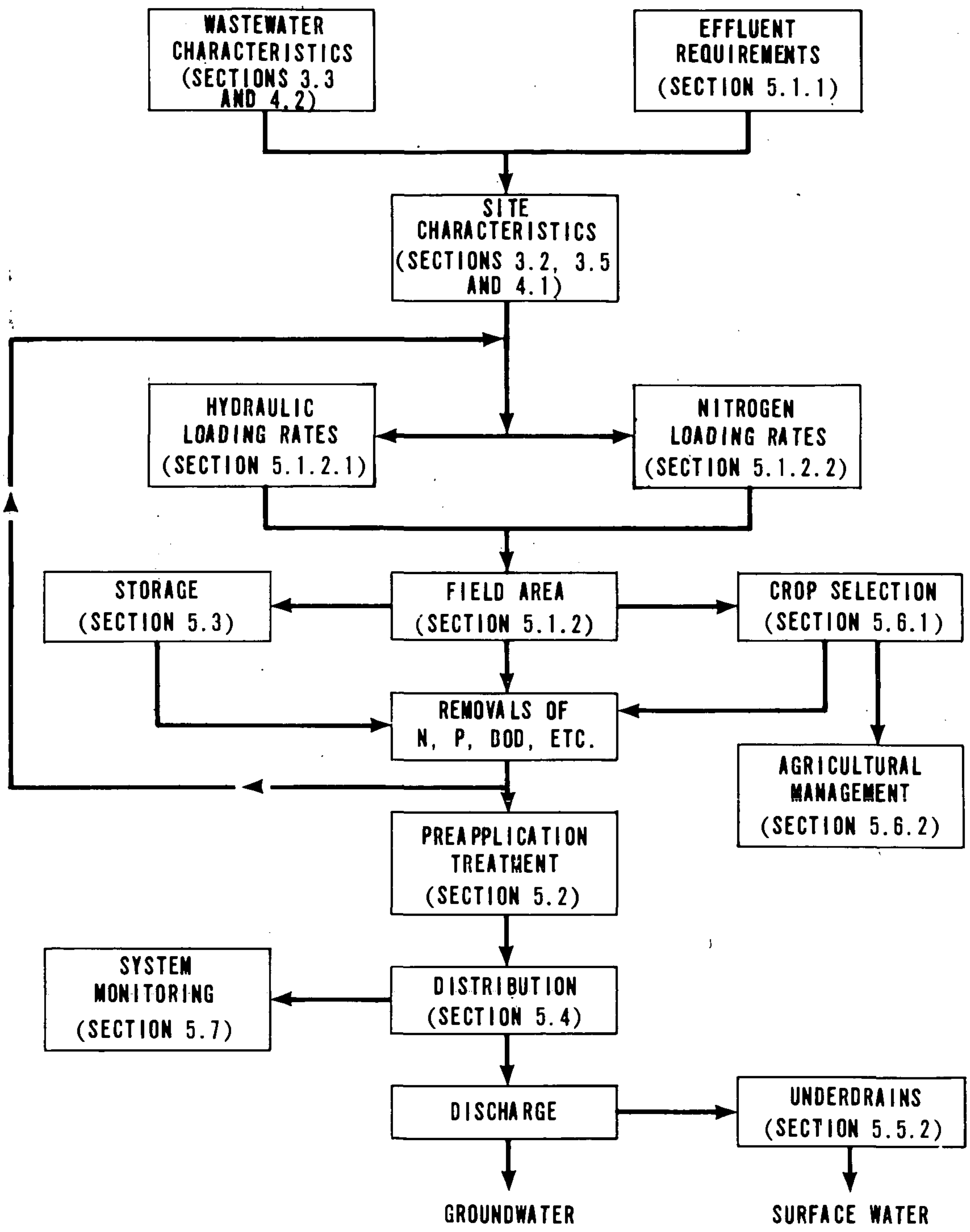
$$L_w + Pr = ET + W_p + R \tag{5-1}$$

where L_w = wastewater hydraulic loading rate, in./wk (cm/wk)
Pr = design precipitation, in./wk (cm/wk)
ET = evapotranspiration (or crop consumptive use of water), in./wk (cm/wk)
W_p = percolating water, in./wk (cm/wk)
R = net runoff, in./wk (cm/wk)

The relationship in Equation 5-1 can be used for a weekly balance, as shown, or for monthly or annual balances. Design precipitation is calculated from a 10 year return frequency analysis of wetter-than-normal conditions using all the available data (Section 3.5.4.1). Evapotranspiration estimates can be obtained from extension specialists, land grant universities, or irrigation specialists. Peak rates for selected crops that affect maximum hydraulic loadings are presented in Section 5.6.1. Expected percolating water can be estimated from soil characteristics and verified with field investigations (Appendix C). For slow rate systems, wastewater is assumed to percolate, so net runoff, R , can be assumed to be negligible.

FIGURE 5-1

SLOW RATE DESIGN PROCEDURE



5.1.2.2 Nitrogen Loading Rates

Nitrogen management for the slow rate process is principally crop uptake with some denitrification. The annual nitrogen balance is:

$$L_n = U + D + 2.7\, W_p C_p \qquad (5\text{-}2)$$

where L_n = wastewater nitrogen loading, lb/acre·yr (kg/ha·yr)
U = crop nitrogen uptake, lb/acre·yr (kg/ha·yr)
D = denitrification, lb/acre·yr (kg/ha·yr)
W_p = percolating water, ft/yr (cm/yr)
C_p = percolate nitrogen concentration, mg/L

Crop nitrogen uptake values are presented in Table 5-2. These values are based on typical yields under commercial fertilization and may increase where conditions of excess nitrogen prevail. Crop nitrogen uptake values in design depend on actual crop yields and local agricultural agents should be contacted. Double cropping of field crops such as corn and barley can increase the total annual nitrogen uptake.

TABLE 5-2

TYPICAL VALUES OF CROP UPTAKE OF NITROGEN[a]
[3, 4, 5, 6]

Crop	Nitrogen uptake, lb/acre·yr
Forage crops	
Alfalfa[b]	200-480
Coastal bermuda grass	350-600
Kentucky bluegrass	180-240
Bromegrass	116-200
Reed canary grass	300-400
Sweet clover[b]	158
Tall fescue	135-290
Quackgrass	210-250
Field crops	
Barley	63
Corn	155-172
Cotton	66-100
Milo maize	81
Soybeans[b]	94-128
Forest crops	
Young deciduous	100
Young evergreen	60
Medium and mature deciduous	30-50
Medium and mature evergreen	20-30

a. For choice of suitable crop and uptake value, contact the local agricultural agent.

b. Legumes will also take nitrogen from the atmosphere.

1 lb/acre·yr = 1.12 kg/ha·yr

Denitrification is difficult to determine under field conditions, but losses generally range from 15 to 25% of the applied nitrogen. Conditions favorable to increased denitrification are summarized in Table 5-3. Volatilization is known to occur (see Appendix A) but is difficult to quantify.

TABLE 5-3

FACTORS FAVORING DENITRIFICATION IN THE SOIL

High organic matter
Fine textured soils
Frequent wetting
High groundwater table
Neutral to slightly alkaline pH
Vegetative cover
Warm temperature

The percolate nitrogen will be limited in concentration to 10 mg/L for design purposes if the flow is to Case 1 or Case 2 groundwater. An alternative approach is to conduct a geohydrologic study (Appendix C) to quantify groundwater flow. If it can then be shown that groundwater quality leaving the site meets Case 1 or Case 2 requirements, then a higher design percolate nitrogen concentration should be allowed. The percolating water is determined from the water balance. It affects the allowable loading of nitrogen considerably, as illustrated in the following example for both arid and humid climates.

EXAMPLE 5-1: ANNUAL NITROGEN BALANCE FOR DESIGN PERCOLATE NITROGEN CONCENTRATION OF 10 mg/L

Conditions

	Humid climate	Arid climate
1. Applied nitrogen concentration, C_n , mg/L	25	25
2. Crop nitrogen uptake, U , lb/acre·yr	300	300
3. Denitrification, as % of applied nitrogen	20	20
4. Precipitation minus evapotranspiration, Pr - ET , ft/yr	1.7	-1.7

The annual water balance, using Equation 5-1, is:

$$L_w + Pr = ET + W_p + R$$

$$\text{or} \quad W_p = L_w + P_r - ET - 0$$

$$W_p = L_w + 1.7 \quad \text{(humid)}$$

$$W_p = L_w - 1.7 \quad \text{(arid)}$$

The amount of percolating water, W_p , resulting from the applied effluent, L_w , has a significant effect on the allowable nitrogen loading, Ln .

The annual nitrogen balance, using Equation 5-2, is:

$$L_n = U + D + 2.7\, W_pC_p \tag{5-2}$$

$$L_n = 300 + 0.2\, L_n + (2.7)(L_w + 1.7)(10) \quad \text{(humid)}$$

$$L_n = 300 + 0.2\, L_n + (2.7)(L_w - 1.7)(10) \quad \text{(arid)}$$

The relationship between the nitrogen loading and the hydraulic loading is:

$$L_n = 2.7\ C_n L_w \quad \text{(U.S. customary)} \tag{5-3}$$

$$L_n = 0.1\ C_n L_w \quad \text{(SI units)} \tag{5-3a}$$

where L_n = wastewater nitrogen loading, lb/acre·yr (kg/ha·yr)
C_n = applied nitrogen concentration, mg/L
L_w = wastewater hydraulic loading, ft/yr (cm/yr)

Therefore, for this example,

$$L_n = (2.7)(25)\ L_w$$
$$= 67.5\ L_w$$
$$\text{or} \quad L_w = 0.015\ L_n$$

With two equations and two unknowns, the nitrogen balance can now be solved:

Humid climate

$$L_n = 300 + 0.2\ L_n + (2.7)(L_w + 1.7)(10)$$
$$L_n = 300 + 0.2\ L_n + (2.7)(0.015\ L_n + 1.7)(10)$$
$$L_n = 300 + 0.2\ L_n + 0.405\ L_n + 45.9$$
$$0.395\ L_n = 345.9$$
$$L_n = 875 \text{ lb/acre·yr}$$

Arid climate

$$L_n = 300 + 0.2\ L_n + (2.7)(L_w - 1.7)(10)$$
$$L_n = 300 + 0.2\ L_n + (2.7)(0.015\ L_n - 1.7)(10)$$
$$L_n = 300 + 0.2\ L_n + 0.405\ L_n - 45.9$$
$$0.395\ L_n = 254.1$$
$$L_n = 643 \text{ lb/acre·yr}$$

Complete solution

	Humid climate	Arid climate
1. Wastewater nitrogen loading, L_n, lb/acre·yr	875	643
2. Wastewater hydraulic loading, L_w, ft/yr	13.1	9.6
3. Percolating water, W_p, ft/yr	14.8	7.9
4. Denitrification, D, lb/acre·yr	175	129
5. Percolate nitrogen loading, $P_n = 2.7\ C_p W_p$, lb/acre·yr	400	213

1 lb/acre·yr = 1.12 kg/ha
1 ft/yr = 0.305 m/yr

5.1.2.3 BOD and SS Removal

As indicated in Table 2-3, the expected BOD concentration of treated water after 5 ft (1.5 m) of percolation is less than 2 mg/L. At Hanover, New Hampshire, percolate BOD ranged from 0.6 to 2.1 mg/L after passage through 5 ft (1.5 m). For a primary effluent with a BOD concentration of about 100 mg/L at a loading rate of 5 lb/acre·d (5.6 kg/ha·d), the average percolate concentration was 1.5 mg/L [7]. For industrial wastewaters, BOD loadings have exceeded 200 lb/acre·d (224 kg/ha·d). Suspended solids removals are expected to be similar to BOD removals although few data are available for existing systems.

5.1.2.4 Phosphorus Removal

Phosphorus retention is extremely effective in slow rate systems as a result of adsorption and chemical precipitation. Phosphorus retention for sites can be enhanced by use of crops such as grass with large phosphorus uptake. Grass also minimizes soil erosion and surface runoff losses. Field determination of levels of free iron oxides, calcium, and aluminum, and soil pH will provide information on the type of chemical reactions that will occur. Determination of the phosphorus sorption capacity of the soils requires laboratory testing with field samples from the proposed areas (see Appendix F).

The estimated phosphorus retention from the empirical model (Appendix B.4.4) can be computed for the loading and soil sorption properties. Systems with strict phosphorus control for recovered water should include routine soil phosphorus monitoring to verify retention in the soil and system performance.

5.1.2.5 Trace Element Removal

An evaluation of the annual applications of trace metals should be made on the basis of wastewater applications and an estimate of wastewater concentrations from field testing or existing data (see Table 3-4). The assessment of trace metal concentrations is especially important in cases where industrial sources are present. The potential toxicity to plants can be assessed by comparing loadings computed to the recommended application values for sensitive crops (Table 5-4). In cases where annual total application or applied concentrations approach levels shown in Table 5-4, system management to maintain soil pH at 6.5 or above by liming may be needed.

5.1.2.6 Microorganism Removal

The minimizing of public health risks is a basic goal for any wastewater treatment system. The potential for public health risks resulting from land application of wastewater varies greatly depending upon specific site details such as:

1. Type of application
2. Public access to the site
3. Preapplication treatment
4. Population density and adjacent land use
5. Type of disposition of vegetative cover
6. Natural occurring and artificial onsite buffer zones
7. Climate

The U.S. Army Medical Department and the EPA have conducted and are continuing to conduct studies at operational land application sites to document microorganism removal and transport mechanisms. These locations include Fort Devens, Massachusetts, Deer Creek, Ohio; Fort Huachuca, Arizona; and Pleasanton, California. In addition, the development of a mathematical model describing aerosol transport is underway by the U.S. Army.

TABLE 5-4

SUGGESTED MAXIMUM APPLICATIONS OF TRACE ELEMENTS TO SOILS WITHOUT FURTHER INVESTIGATION[a]

Element	Mass application to soil, lb/acre	Typical concentration, mg/L[b]
Aluminum	4 080	10
Arsenic	82	0.2
Berylium	82	0.2
Boron	610	1.4[c]
Cadmium	8	0.02
Chromium	82	0.2
Cobalt	41	0.1
Copper	164	0.4
Fluoride	820	1.8
Iron	4 080	10
Lead	4 080	10
Lithium	.....	2.5[d]
Manganese	164	0.4
Molybdenum	8	0.02
Nickel	164	0.4
Selenium	16	0.04
Zinc	1 640	4

a. Values were developed for sensitive crops on soils with low capacities to retain elements in available forms [8, 9].

b. Based on reaching maximum mass application in 20 years at an annual application rate of 8 ft/yr.

c. Boron exhibits toxicity to sensitive plants at values of 0.75 to 1.0 mg/L.

d. Lithium toxicity limit is suggested at 2.5 mg/L concentration for all crops, except citrus which uses a 0.075 mg/L limit. Soil retention is extremely limited.

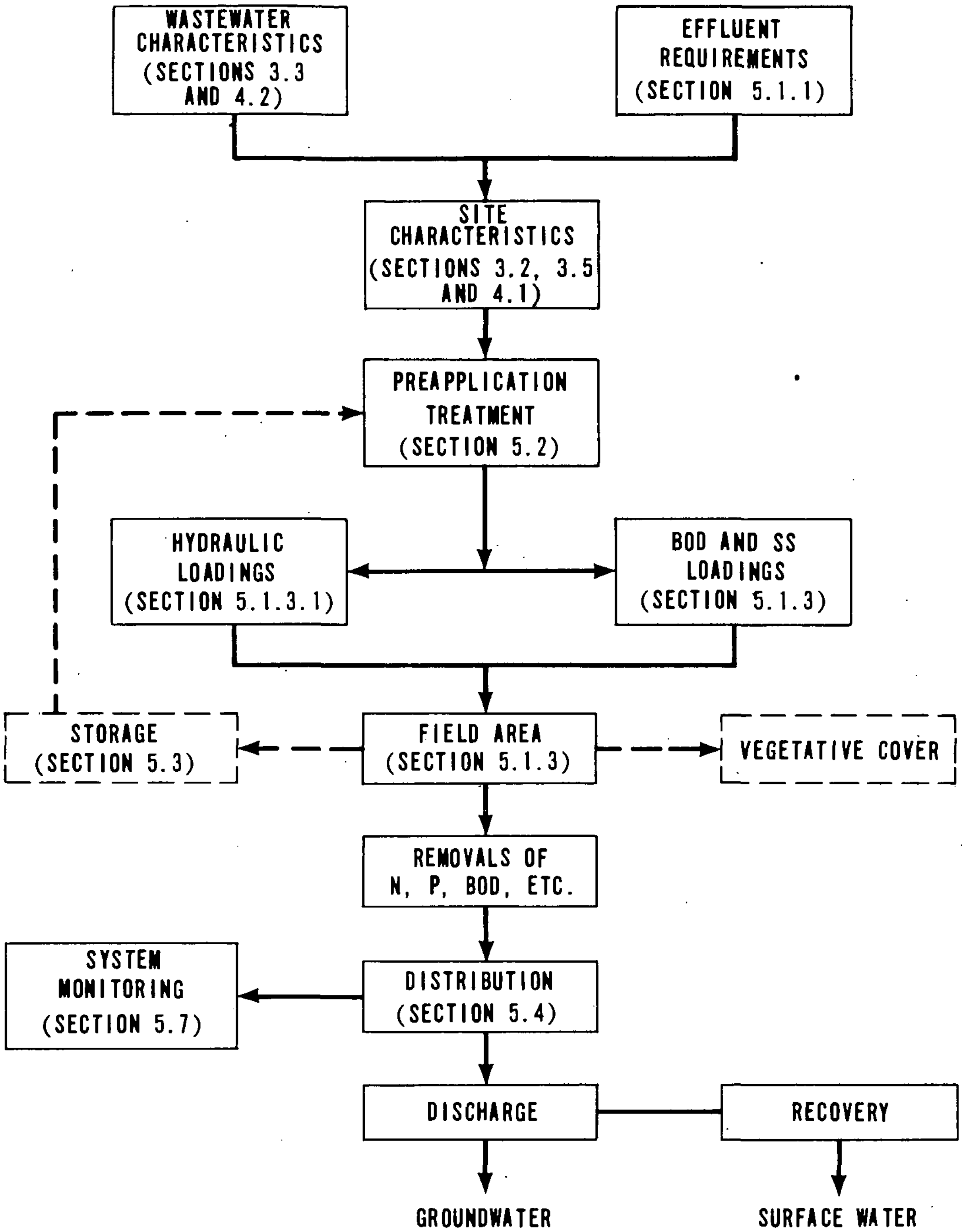
1 lb/acre = 1.12 kg/ha
1 ft =0.305 m

5.1.3 Rapid Infiltration

The design procedure for rapid infiltration is presented in Figure 5-2. The principal differences from slow rate systems are (1) hydraulic applications are greater, so greater reliability of permeability measurements is required; (2) nitrogen removal mechanisms rely less on crop uptake and more on nitrification-denitrification; (3) solids applications are greater; and (4) systems can be adapted to severe climates.

FIGURE 5-2

RAPID INFILTRATION DESIGN PROCEDURE



5.1.3.1 Hydraulic Loading Rates

Hydraulic loadings and subsequent liquid movement through the soil depend on soil permeability, subsurface geological conditions, and constituent loadings. Annual hydraulic loading rates can range from 20 to 400 ft/yr (6 to 120 m/yr). Typical loading rates are shown in Table 5-5.

TABLE 5-5

TYPICAL HYDRAULIC LOADING RATES FOR RI SYSTEMS

Location	Hydraulic loading rate, ft/yr	Soil type	Type of wastewater
Flushing Meadows, Arizona	364	Sand	Secondary
Santee, California	265	Gravel	Secondary
Lake George, New York	140	Sand	Secondary
Calumet, Michigan	110	Sand	Untreated
Hemet, California	108	Sand	Secondary
Hollister, California	97	Sandy loam	Primary
Fort Devens, Massachusetts	94	Sand and gravel	Primary
Westby, Wisconsin	36	Silt loam	Secondary

1 ft/yr = 0.305 m/yr

System design for a rapid infiltration process includes the interrelated factors of hydraulic loading rate per application cycle, soil infiltration capacity, application and resting cycle, solids applied in the wastewater, and subsoil permeability. Although site investigations may show that the infiltration rate is greater than the soil permeability, the infiltration rate, under design conditions with solids applications, will usually decrease and control liquid applications. Figure 3-3 can be used for an initial estimate of the average infiltration rate. For final design values, soil infiltration tests (described in Appendix C) should be conducted. The most limiting layer in the soil profile should be evaluated and that permeability should be used in design.

The operating infiltration rate will vary between two values: one being the initial rate for clean soil and clear water, and the other being a decreased rate for wastewater, with a surface accumulation of organics and other suspended solids. The cycle of application and resting is designed to restore the infiltration rate to nearly its initial value by the end of the resting period. For a specific rapid infiltration site, a design decision has to be made that balances suspended solids application, land area requirements, and resting requirements.

5.1.3.2 Hydraulic Loading Cycles

The existing hydraulic loading and resting cycles of rapid infiltration systems, as given in Table 5-6, demonstrate several design concepts. Most systems are intended to maximize infiltration rates, although Flushing Meadows, Arizona, and Fort Devens, Massachusetts, experimented with the cycle to promote denitrification.

TABLE 5-6

TYPICAL HYDRAULIC LOADING CYCLES [11]

Location	Loading objective	Application period	Resting period	Bed surface
Calumet, Michigan	Maximize infiltration rates	1-2 d	7-14 d	Sand (not cleaned)
Flushing Meadows, Arizona				
Maximum infiltration	Increase ammonium adsorption capacity	2 d	5 d	Sand (cleaned) and grass cover[a]
Summer	Maximize nitrogen removal	2 wk	10 d	Sand (cleaned) and grass cover[a]
Winter	Maximize nitrogen removal	2 wk	20 d	Sand (cleaned) and grass cover[a]
Fort Devens, Massachusetts	Maximize infiltration rates	2 d	14 d	Grass (not cleaned)
Fort Devens, Massachusetts	Maximize nitrogen removal	7 d	14 d	Grass (not cleaned)
Lake George, New York				
Summer	Maximize infiltration rates	9 h	4-5 d	Sand (cleaned)[a]
Winter	Maximize infiltration rates	9 h	5-10 d	Sand (cleaned)[a]
Tel Aviv, Israel	Maximize renovation	5-6 d	10-12 d	Sand[b]
Vineland, New Jersey	Maximize infiltration rates	1-2 d	7-10 d	Sand (disked), solids turned into soil[c]
Westby, Wisconsin	Maximize infiltration rates	2 wk	2 wk	Grassed
Whittier Narrows, California	Maximize infiltration rates	9 h	15 h	Pea gravel

a. Cleaning usually involved physical removal of surface solids.

b. Maintenance of sand cover is unknown.

c. Solids are incorporated into surface sand.

For basin surfaces with grass or vegetation, the need for maintenance is less strict than for bare surfaces. Based on operations at Fort Devens, the grass should be allowed to grow and die without placing heavy mechanical equipment, which compacts the surface, on the infiltration beds. Periodic harrowing of the soil surface or mowing of the grass may be considered depending on aesthetic demands.

In summary, system design for maximum infiltration rates should include adequate drying time based on local climate and solids loadings to restore infiltration rates. If the soil surface is maintained bare of vegetation, the surface should be periodically raked, harrowed, or disked. Nitrogen removal by denitrification will require additional considerations for lesser soil aeration and the effect of lessened opportunity for solids degradation.

5.1.3.3 Nitrogen Loading Rates

Because nitrogen loading rates can exceed crop uptake by an order of magnitude, crop uptake (if a crop is planted) is relatively minor and nitrification, denitrification, and ammonium sorption are generally of greatest importance.

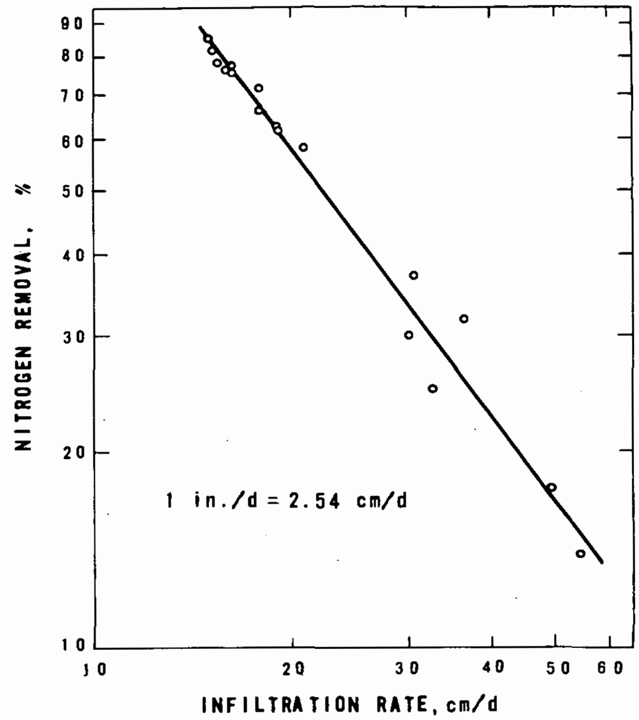
The retention of ammonium by the cation exchange capacity can be excellent. The conversion of ammonium to nitrate occurs rapidly when short, frequent applications are used to promote aerobic conditions in the soil. Longer application cycles, which restrict soil reaeration, favor nitrogen loss by denitrification. Available organic matter in the soil profile as a result of applied BOD also increases the amount of denitrification. The most comprehensive work on nitrogen has been conducted at the Flushing Meadows Project (described in Section 7.8).

At the Flushing Meadows Project, at a 365 ft/yr (111 m/yr) application, the sustained removal of nitrogen was 30% [12]. For lower application rates Lance found that the nitrogen removal increased to over 80% (see Figure 5-3). Although the relationship is strictly valid only for the sandy soil and secondary effluent used at Flushing Meadows, similar relationships should exist for other soils and wastewaters. Infiltration rates in the field can be changed by modifying the depth of flooding, compacting the soil surface, or by applying wastewater containing higher BOD and suspended solids [10].

When nitrification is the objective of rapid infiltration, short application periods followed by relatively long resting periods are used. Rapid infiltration systems will produce a nitrified effluent at nitrogen loadings up to 60 lb/acre·d (67.2 kg/ha·d). Nitrification below 36°F (2°C) and below pH 4.5 is minimal (Appendix A).

FIGURE 5-3

EFFECT OF INFILTRATION RATE ON NITROGEN REMOVAL FOR RAPID INFILTRATION, PHOENIX, ARIZONA [10]



5.1.3.4 BOD and SS Removal

Removals of BOD and suspended solids depend on the soil type and travel distance in the soil. Removal of BOD is primarily accomplished by aerobic bacteria that depend on resting periods to reaerate the soil. Loading rates have some effect on removals but too many other variables such as temperature, resting period, and soil type are involved to allow estimation of removals from loading rates alone. Selected loading rates and concentrations in the treated water are presented in Table 5-7.

5.1.3.5 Phosphorus Removal

The basic mechanisms for phosphorus removal are similar to those described for slow rate systems (Section 5.1.2.4). The coarser textured soils used for rapid infiltration may have less retention capacity for phosphorus. Soil capabilities can be estimated from specific testing (Appendix F; Section F.3.3.2).

TABLE 5-7

BOD AND SUSPENDED SOLIDS DATA FOR SELECTED RAPID INFILTRATION SYSTEMS [13-16]

Location	BOD		Suspended solids		Sampling depths, ft
	Average loading rate, lb/acre·d[a]	Treated water concentration, mg/L	Average loading rate, lb/acre·d	Treated water concentration, mg/L	
Phoenix, Arizona	40	0-1	54	0.8	100
Lake George, New York	47	1.2	...	...	10
Calumet, Michigan	71	11[b]	43	...	11
Hollister, California	158	8	197	...	25
Fort Devens, Massachusetts	78	12	...	...	64

a. Total lb/acre·yr applied divided by the number of days in the operating season (365 days for these cases).

b. Soluble TOC.

1 lb/acre·d = 1.12 kg/ha·d

5.1.3.6 Trace Element Removal

As indicated in Section 5.1.2.5, heavy metals are removed from solution by the adsorptive process and by precipitation and ion exchange in the soil. The concerns about heavy metals in rapid infiltration systems are: (1) the high rates of application, and (2) the potentially low adsorptive potential of the coarse soils. The heavy metal application criteria (Table 5-4), recommended to ensure protection of sensitive plants in slow rate systems, can be safely exceeded for rapid infiltration systems because sensitive agricultural crops are not grown.

5.1.3.7 Microorganism Removal

The mechanisms of microorganism removal include straining, sedimentation, predation and desiccation during preapplication treatment; desiccation and radiation during application; and straining, desiccation, radiation, predation, and hostile environmental factors upon application to the land. Removals of fecal coliforms for selected rapid infiltration systems are presented in Table 5-8.

TABLE 5-8

FECAL COLIFORM REMOVAL IN SELECTED RAPID INFILTRATION SYSTEMS

Location	Soil type	Fecal coliforms, MPN/100 mL: Effluent applied	Fecal coliforms, MPN/100 mL: Renovated water	Sampling depth, ft
Phoenix, Arizona	Sand	1 000 000	0-30	100
Hemet, California	Sand	60 000	11	8
Calumet, Michigan	Sand	--[a]	1-10	10

a. Untreated wastewater.

1 ft = 0.305 m

5.1.4 Overland Flow

Overland flow systems use the land surface as the treatment medium over which a thin sheet of wastewater moves and upon which the biological and chemical processes occur. The design procedure is typically to select a hydraulic loading based on the required treatment performance for BOD in the wastewater. Nitrogen removals or transformations are then assessed based on comparison with existing systems. The design procedure is presented in Figure 5-4.

5.1.4.1 Hydraulic Loading Rates

Hydraulic loading rates, when untreated or primary effluent is applied, can range from 2.5 to 8 in./wk (6.4 to 20 cm/wk) depending on the climate, required treatment performance, and detention time on the slope. Lower values of 3 to 4 in./wk (7.5 to 10 cm/wk) should be considered (1) for slopes greater than 6%, (2) for terraces less than 150 ft (45 m), or (3) because of reduced biological activity during very cold weather. Thomas has reported excellent results using untreated wastewater at about 4 in./wk (10 cm/wk) on 2 to 4% slopes 120 ft (36 m) long [17]. Recently, Thomas has experimented with loadings of 6 and 8 in./wk (15 and 20 cm/wk) with untreated wastewater and primary effluent and has indicated continued excellent removals of BOD, suspended solids, and nitrogen [18].

For lagoon or secondary effluent, loadings of 6 to 16 in./wk (15 to 40 cm/wk) can be considered. Lower values of 7 to 10 in./wk (17.5 to 25 cm/wk) should be considered when the factors (1) through (3) described above, apply. Loading rates and design conditions for four demonstration projects are presented in Table 5-9.

FIGURE 5-4

OVERLAND FLOW DESIGN PROCEDURE

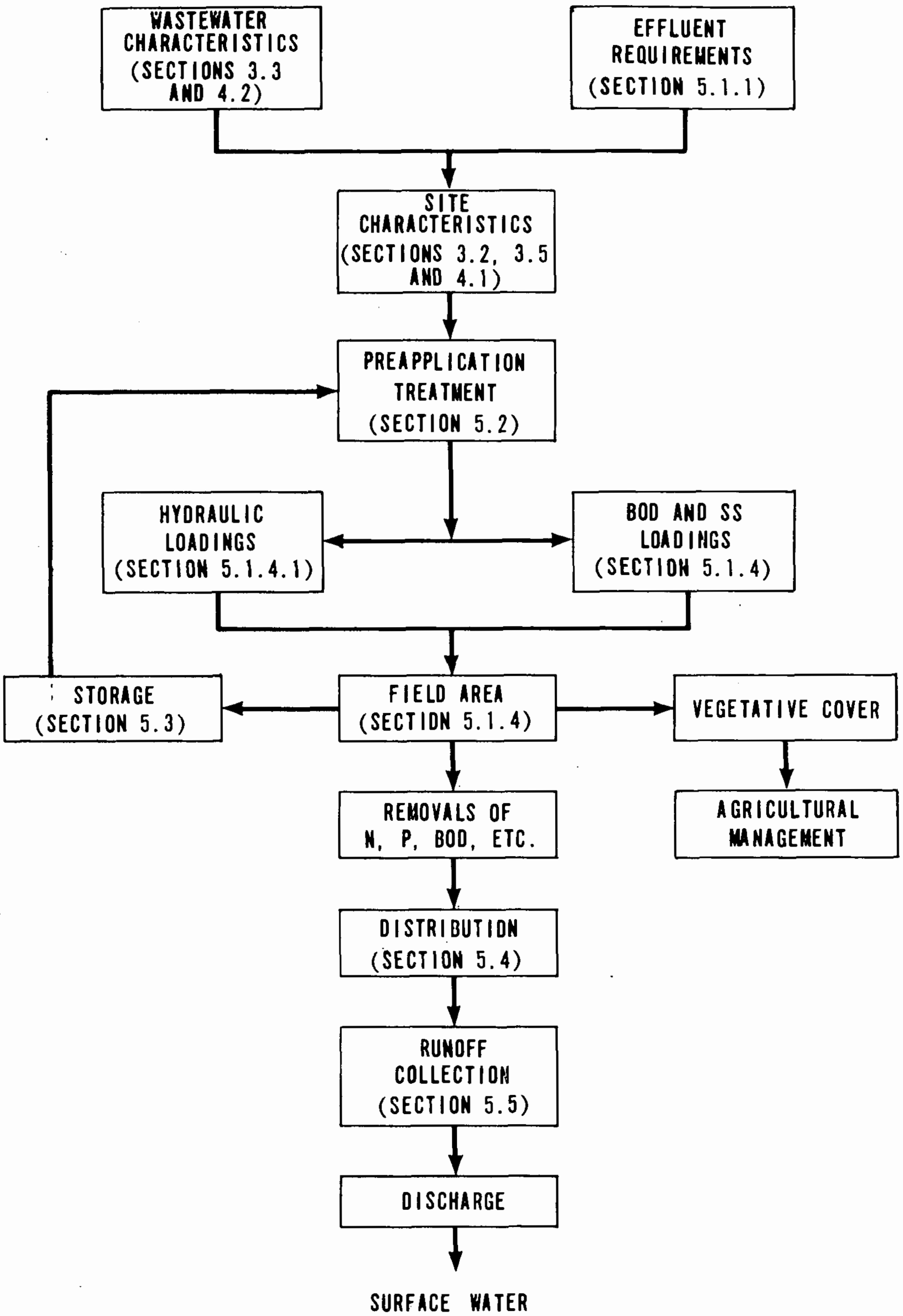


TABLE 5-9

SELECTED HYDRAULIC LOADING RATES FOR OVERLAND FLOW SYSTEMS

	Type of effluent applied	Hydraulic loading rates, in./wk	Degree of slope, %	Slope length, ft
Ada, Oklahoma	Raw comminuted	4-8	2-4	120
Ada, Oklahoma	Trickling filter	10-16	2-4	120
Pauls Valley, Oklahoma	Oxidation pond	10.3	2-3	150
Utica, Mississippi	Oxidation pond	2.5-5	2-8	150

1 in./wk = 2.54 cm/wk
1 ft = 0.305 m

Loading rates and cycles for an overland flow system are designed to maintain active microorganism growth on the soil surface. The operating principles are similar to a conventional trickling filter with intermittent dosing. The rate and length of application should be controlled; anaerobic conditions can result from overstressing the system. The resting period should be long enough to allow the soil surface layer to reaerate, yet short enough to keep the microorganisms in an active state. Experience with existing systems indicates that optimum cycles range from 6 to 8 hours on and 16 to 18 hours off, for 5 to 6 d/wk depending on the time of year. Application periods may be extended during the summer months to allow portions of the system to be taken out of service for crop harvesting.

5.1.4.2 Nitrogen Removal

Nitrogen removal in overland flow systems is excellent. Two important mechanisms responsible for these removals are biological nitrification/denitrification and crop uptake. The overlying water film and organic matter, and the underlying saturated soil form an aerobic-anaerobic double layer necessary for nitrification followed by denitrification. These conditions are similar to those found in rice fields or marshes. The treated runoff quality for nitrogen at Ada, Oklahoma, is shown in Table 5-10.

5.1.4.3 BOD and SS Removal

Removals of BOD, both at Ada, Oklahoma and at Paris, Texas, have improved with system age. At Ada, after about 100 days of operation, the BOD concentration in the runoff stabilized at an average of 8 mg/L for the 4 in./wk (10 cm/wk) rate [17]. At Paris, Texas (described in

Section 7.12), the BOD in the treated runoff improved from an average of 9 mg/L in 1968 to an average of 3.3 mg/L in 1976.

Suspended solids removals are generally less than those for BOD. At Ada, removals averaged 95% and concentrations ranged from 8 to 16 mg/L for the 4 in./wk (10 cm/wk) rate [17]. At Paris, Texas, 245 mg/L of suspended solids is applied and the treated runoff typically contains 25 to 30 mg/L of suspended solids [11].

TABLE 5-10

NITROGEN CONCENTRATIONS IN TREATED RUNOFF
FROM OVERLAND FLOW WHEN USING UNTREATED WASTEWATER [17, 18]
mg/L

	Loading rate, in./wk	
Nitrogen forms	4	8
Total	2.9	3
Organic	1.6	..[a]
Ammonia	0.8	..[a]
Nitrate and nitrite	0.5	..[a]

a. Not measured, but assumed to be similar to 4 in./wk loading rate.

1 in./wk = 2.54 cm/wk

5.1.4.4 Phosphorus Removal

Of the three major land treatment processes, overland flow systems have the most limited potential for phosphorus removal. Because there is very limited percolation of wastewater in overland flow systems, the soil-water contact is limited to the soil surface area. The wastewater flowing over the soil surface does not have extensive contact with the components of the soil that normally fix large amounts of phosphorus. In addition, the residence time on the slope is usually less than 24 hours. However, some phosphorus appears to be removed by the organic layer on the surface of overland flow slopes [19] and the grass will take up 30 to 40 lb/acre·yr (33 to 44 kg/ha·yr).

At Ada, Oklahoma, alum was added to the wastewater prior to application. For an application rate of 4 in./wk (10 cm/wk), the removals of phosphorus are presented in Table 5-11. Similar results were obtained at Vicksburg, Mississippi [20].

TABLE 5-11

PHOSPHORUS CONCENTRATIONS IN TREATED RUNOFF FROM OVERLAND FLOW, ADA, OKLAHOMA [17]

Sample	Concentration of total phosphorus, mg/L	Phosphorus removal, %
Untreated wastewater	9.8	..
Runoff		
No alum	3.7	62
14 mg/L alum	1.6	84
20 mg/L alum	1.5	85

Carlson et al. reported 75% phosphorus removal from secondary effluent in greenhouse studies at 0.5 in./d (1.3 cm/d) applications [21]. At Melbourne, Australia, phosphorus removals of 35% from raw wastewater are reported (Appendix B).

5.1.4.5 Trace Element Removal

Trace element removal by overland flow is relatively good. Hunt and Lee [19] report that rates of removal are greater than 90% for all, and greater than 98% for some heavy metals. It is believed that most of the heavy metals are removed in the surface organic mat.

5.1.4.6 Microorganism Removal

The mechanisms involved in the removal of bacteria by the soil in overland flow systems are similar to those for removal of metals. At the pilot study at Ada, Oklahoma, the overall reduction for total coliforms was about 95%, while fecal coliform reduction was about 90% [17].

5.1.5 Wetlands Application

The designed use of wetlands to receive and satisfactorily treat wastewater effluents is a relatively new concept. At present, the use of wetlands has not been incorporated into large, full-scale treatment systems; however, the potential treatment capacity has been confirmed at many pilot systems and research sites. Hydraulic loadings and general performance criteria are given in Table 5-12.

TABLE 5-12

HYDRAULIC LOADINGS AND GENERAL PERFORMANCE CRITERIA - RESEARCH AND DEMONSTRATION WETLAND SYSTEMS

					Final concentrations, mg/L			
Location	System type	Preapplication treatment	Length of application	Application rate, in./d	BOD	Suspended solids	Total nitrogen	Total phosphorus
New York [22]	Constructed marsh/pond	Aerated	Year-round	1.8	16	43	4	2.2
Wisconsin [23]	Constructed marsh	Primary and secondary	Year-round	...[a]	5-18	14-80	......	10-12
	Natural marsh	Primary and secondary	Year-round	...	5-27	90-154	0.5-2	1-3
California [24]	Constructed marsh	Secondary	Year-round	...	....	......	<7	2.8
New Jersey [25]	Natural tidal marsh	Secondary	Year-round	2.0-5.0	....	......	......	.....
Canadian Northwest [26]	Natural swamp	Lagoon and raw septage	Year-round	....	<10[b]	<40[b]	<8[b]	<1[b]
Minnesota [27]	Constructed peat bed	Secondary	Summer (campground)	2.0-6.0	<5	<5	<10	<0.8
Florida [28, 29]	Natural cypress dome	Secondary	Year-round	1.0	....	......	0.6-3.8	0.2-1.0

a. Application varies to give detention times of 5 hours to 10 days.

b. At point in swamp 2.26 mi (3 640 m) downstream from outfall.

1 in./d = 2.54 cm/d

5.1.5.1 Process Description

There are several types of wetlands (as mentioned in Chapter 2) with varying amounts of organic substrate and vegetative growth and varying degrees of soil moisture. They require low-lying, usually level, saturated land, sometimes partially or intermittently covered with standing water. In wetland application systems, wastewater is renovated by the soil, plants, and microogranisms as it moves through and over the soil profile. Wetland systems are somewhat similar to overland flow systems in that most of the water flows over a relatively impermeable soil surface and the renovation action is more dependent on microbial and plant activity than soil chemistry.

5.1.5.2 Hydraulic Loadings

Items to be considered in selecting the hydraulic loading include:

1. Detention time of applied wastewater

2. Rate of water loss from system by planned overflow or slow seepage

3. System upsets due to washouts by precipitation or wastewater applications

5.1.5.3 Nitrogen Management

For a wetland system, the following mechanisms should be considered as factors having an influence on nitrogen balance:

1. Denitrification

2. Above ground and below ground plant uptake

3. Dilution

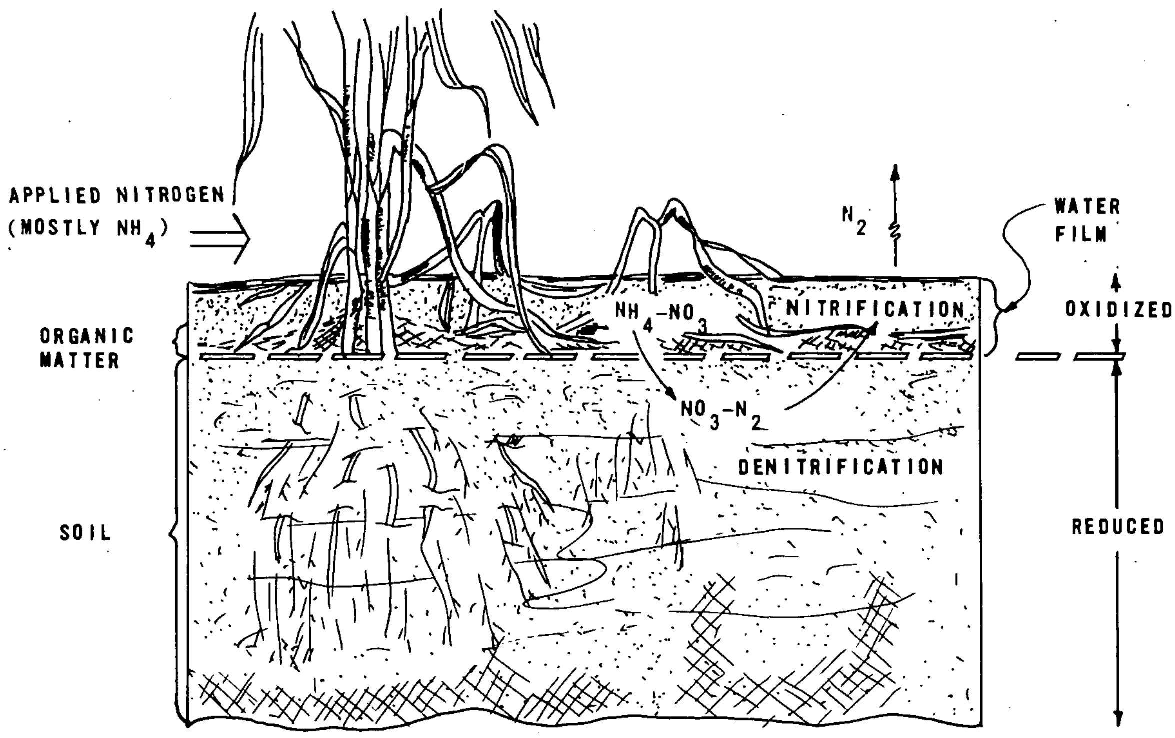
4. Sorption with living or dead material

The biomass productivity of wetland systems has been reported to be 4 to 5 tons/acre (8 to 10 Mg/ha) in Wisconsin marshes, with other reported values up to 6.7 to 8 tons/acre (15 to 18 Mg/ha) [30]. The total plant nitrogen uptake is extremely high since nitrogen content of the plants may be from 2.0 to 2.5%; however, the below ground portion of the plant may contain 4 to 6 times as much biomass as the above ground portion. Thus, the majority of the nitrogen content in the system is released and recycled rather than being available for removal by harvest. The seasonal uptake and release by perennial and annual vegetation is influenced by the nitrogen cycling. In general, natural wetlands can be classified as low in nitrogen availability, because their high organic content can serve as a nitrogen sink. Managed wetlands can facilitate biomass production and harvesting to provide an effective nitrogen removal mechanism.

The schematic diagram presented in Figure 5-5 illustrates the principal nitrogen transformations occurring in a wetland system. The overlying water and underlying organic soil form an aerobic-anaerobic double-layer, thus providing ideal conditions for biological nitrification-denitrification reactions to occur.

FIGURE 5-5

PRINCIPAL NITROGEN TRANSFORMATIONS
IN WETLANDS [20]



5.1.5.4 Climatic Considerations

Climatic considerations for wetland systems are not well defined. Although wetland systems have been utilized in locations from Florida [28] to the Canadian Northwest Territories [26], the mechanisms involved in retention or removal of each wastewater component are uncertain. A wetland system in Wisconsin used for wastewater application is shown in Figure 5-6.

FIGURE 5-6

WETLAND SYSTEM AT BRILLION MARSH, WISCONSIN

5.1.5.5 Phosphorus Management

Within the range of wetland system types, the phosphorus removals can vary considerably. For peatlands, Stanlick reports 99% removal of phosphorus [27]. In Wisconsin, using man-made and natural marshes, Spangler et al. report 30 to 40% removal on a year-round basis [23]. The removal capabilities are generally high during the growing season, because the soils have a high cation exchange capacity (about 105 meq/100 g) and wetland plants account for luxury uptake of phosphorus. Harvest of the above ground plant portions will remove some phosphorus from the system; however, release from the below ground portion will occur during the nongrowing season.

5.1.5.6 Trace Elements Removal

Although extensive research has not been done with wetlands, the results of research from metal accumulation in lake and river sediments are generally applicable. Organic soils may have high cation exchange capacities so retention should be excellent; however, the effects of pH on retention must be considered.

5.1.5.7 Microorganism Removal

The removal of pathogenic microorganisms depends on the pathway for water leaving the site. Systems that have no overflow and function by water seepage through a slightly permeable soil will have excellent removal of all microorganisms due to physical entrapment and die-off mechanisms. Surface overflow systems offer a less positive removal, so natural die-off as a function of detention time, climate, and other environmental factors must be assessed.

5.2 Preapplication Treatment

The design of preapplication treatment facilities involves three steps:

1. Determine the level of treatment required for the selected land treatment process and site conditions

2. Select a treatment system capable of meeting this level

3. Establish design criteria and perform detailed design of the selected treatment system

Only the level of treatment required will be discussed, because the second and third items are standard engineering procedures that are not unique to land treatment.

5.2.1 Determination of Level Required

In general, the level of preapplication treatment required is an internal process decision made by the designer to ensure optimum performance of the land treatment process. Preapplication treatment may be necessary for a variety of reasons including (1) improving distribution system reliability, (2) reducing the potential for nuisance conditions, (3) obtaining a higher overall level of wastewater treatment, (4) reducing soil clogging, and (5) reducing the risk of public health impacts. The need for preapplication treatment to reduce impacts on various system components is summarized in Table 5-13.

TABLE 5-13

NEED FOR PREAPPLICATION TREATMENT

System component	Potential problems	Level of treatment and mitigating measures
Storage	Odors and solids accumulation	Screened[a] - add aerators Primary - add aerators[b] Biological
Distribution system	Solids deposition orifice clogging	Screened - velocity control and larger orifice nozzles Primary - velocity control Biological
Crops	Limited selection	Screened - limit to high tolerance crops (i.e., grass) Primary - limit to feed, seed, and fiber crops Biological - disinfect to suit individual needs
Hydraulic loading	Reduced loading rate for rapid infiltration	Screened - increased basin maintenance Primary - increased basin maintenance or use of vegetation Biological
Public access	Health risk due to contact	Screened - posting and fencing or buffer zones or disinfection Primary - posting and fencing or buffer zones or disinfection Biological - disinfect for public access areas

a. Bar screens and comminution.

b. Only for short-term (less than a month) storage.

5.2.1.1 Impacts During Storage

The primary consideration for preapplication treatment prior to storage is reduction of the potential for nuisance conditions. This may require reduction in the settleable solids and organic content of the wastewater to levels achievable with primary treatment. This will minimize the possibility of nuisance conditions developing in the storage lagoon. Such factors as climate, length of storage, and reservoir design will determine the necessary level of BOD reduction. Storage reservoirs provide additional treatment through further biological action, deposition of solids, and long-term pathogen die-off [31]. Supplemental aeration could be provided in the reservoir to meet excessive oxygen demand. An alternative concept would be to design the storage lagoon to double as a deep facultative lagoon. Disinfection prior to storage should not be necessary as long as public access to the storage lagoon is restricted.

5.2.1.2 Impacts on Transmission

Although transmission of wastewater to the application site will usually not govern the level of preapplication treatment, the method of transmission should be taken into consideration. For systems in which wastewater is to be pumped to the application site and no other preapplication treatment is required, coarse screening, degritting, and comminution should be included to avoid excessive wear on the equipment.

5.2.1.3 Impacts on Distribution Systems

Preapplication treatment considerations for surface distribution techniques include coarse and settleable solids removal to avoid solids deposition in ditches and laterals. The need for disinfection will depend on the possibility of public contact with the distribution system.

Different criteria apply to sprinkler distribution systems. To avoid plugging of nozzles, it has been recommended that the size of the largest particle in the applied wastewater be less than one-third the diameter of sprinkler nozzles [11]. Removal of coarse and settleable solids as well as grit and any oil and grease should be a minimum preapplication treatment level to maintain reliability in systems using sprinkler distribution.

5.2.1.4 Impacts on Slow Rate Application

For slow rate systems, hydraulic or nitrogen loadings will generally govern the system performance. Thus, from the standpoint of process performance and soil matrix impacts, preapplication treatment for reduction of organics and suspended solids is not necessary. Industrial wastewaters with high organic strength have been applied to land successfully, and data are available to indicate that no significant difference in overall performance was obtained when both primary and secondary effluent were applied under similar conditions [32].

Where the method of application is by sprinkling, limits on aerosol and mist drift should be considered. Preapplication treatment such as primary settling, secondary treatment, and disinfection all serve to reduce the bacterial content of the effluent and hence reduce the numbers of aerosolized bacteria. The need for secondary treatment or disinfection must be evaluated on a case-by-case basis taking into account (1) the population density of the area, (2) the degree of public access to the site, (3) the relative size of the application area, (4) the feasibility of providing buffer zones or plantings of trees or shrubs, and (5) the prevailing climatic conditions.

For forage crop irrigation, the need for disinfection can be balanced against the exposure risk to the public or grazing animals from pathogens in municipal wastewater. On the basis of a limited comparison of land treatment with conventional treatment and discharge, it was concluded that the relative risks to the public were essentially the same [33]. Primary treatment followed by surface application to land that is fenced has been considered adequate to minimize health risks. For application to parks, golf courses, and areas of public access, biological treatment followed by disinfection is often practiced.

5.2.1.5 Impacts on Rapid Infiltration Application

The potential for soil clogging is higher for rapid infiltration systems than for slow rate systems due to greater hydraulic loading rates. As a minimum, primary treatment to remove coarse and settleable solids should be included as preapplication treatment. Reduction of solids to secondary levels will increase the allowable hydraulic loading rates for rapid infiltration systems and a balance can be achieved between the degree of preapplication treatment and the hydraulic loading rate. Algae carryover from holding ponds or lagoons will increase the potential for soil clogging. The use of in situ pilot studies may be required to develop soil response relationships between hydraulic loading and wastewater solids and organic levels [11].

5.2.1.6 Impacts on Overland Flow Application

Because overland flow treatment is basically a surface phenomenon, soil clogging is not a problem, and high BOD and suspended solids removals have been achieved with systems applying raw comminuted municipal wastewater [34] and industrial wastewater with 616 mg/L BOD and 263 mg/L of suspended solids [35]. Thus, preapplication treatment for removal of organics and solids would be necessary only to the extent required by other system components. The need for predisinfection would be governed by consideration of the method of distribution. Low-pressure, large-droplet, downward sprinkling nozzles and bubbling orifices or gated pipe distribution should not require predisinfection if public access is controlled. Postdisinfection of the wastewater runoff may be required. Because overland flow systems are less effective for removal of phosphorus than other land treatment methods, preapplication treatment to enhance overall phosphorus removal may be necessary if a low level is required in the collected runoff.

5.2.2 Industrial Pretreatment

Pretreatment of industrial wastewaters discharged into municipal systems may be required for several reasons, including (1) protection of the collection system; and (2) removal of constituents that would have an adverse impact on the treatment system or would pass through the treatment process relatively unchanged, causing unacceptable effluent quality. General guidelines for pretreatment of industrial wastes discharged to municipal systems using conventional secondary treatment have been published by the EPA [36].

Pollutants that are compatible with conventional secondary treatment systems would generally be compatible with land treatment systems. As with conventional systems, pretreatment requirements will be necessary for such constituents as fats, grease, and oils, and sulfides to protect collection systems and treatment components. Pretreatment requirements for conventional biological treatment will also protect land treatment processes.

High levels of sodium decrease the soil permeability. Pretreatment requirements may be necessary for industrial wastes high in sodium, if the SAR of the total wastewater might be increased to unacceptable levels.

Plant toxicity from metals is discussed in Appendix E and recommended maximum concentrations of trace elements in irrigation water have been previously presented in Table 5-4. Concern over accumulation in the food chain is greatest for cadmium, as discussed in Appendix E. The potential for groundwater contamination from trace elements is greatest for rapid infiltration systems, although the ability of soils to remove and accumulate heavy metals from such systems has been demonstrated [13, 37].

5.3 Storage

There is a need for storage in many land treatment systems because of the effect of climate on treatment or an imbalance between wastewater supply and application. Slow rate and overland flow systems may cease operation during adverse climatic conditions whereas rapid infiltration systems can usually continue operation year-round. An alternative to storage may be seasonal discharge to surface waters.

5.3.1 Determining Storage Needs

The National Climatic Center in Asheville, North Carolina, has conducted an extensive study of climatic variations throughout the United States and the effect of these variations on storage requirements for soil treatment systems [38]. Three computer programs, as presented in Table 5-14, have been developed to estimate the storage days required when inclement weather conditions preclude land treatment system operation.

TABLE 5-14

SUMMARY OF COMPUTER PROGRAMS FOR DETERMINING STORAGE FROM CLIMATIC VARIABLES

EPA program	Applicability	Variables	Remarks
EPA-1	Cold climates	Mean temperature, rainfall, snow depth	Uses freeze index
EPA-2	Wet climates	Rainfall	Storage to avoid surface runoff
EPA-3	Moderate climates	Maximum and minimum temperature, rainfall, snow depth	Variation of EPA-1 for partly favorable conditions

Depending on the dominant climatic conditions of a region, one of the three computer programs will be most suitable. The program best suited to a particular region is shown in Figure 5-7. The maximum storage days over the period of record is calculated as well as storage days for recurrence intervals of 5, 10, and 50 years.

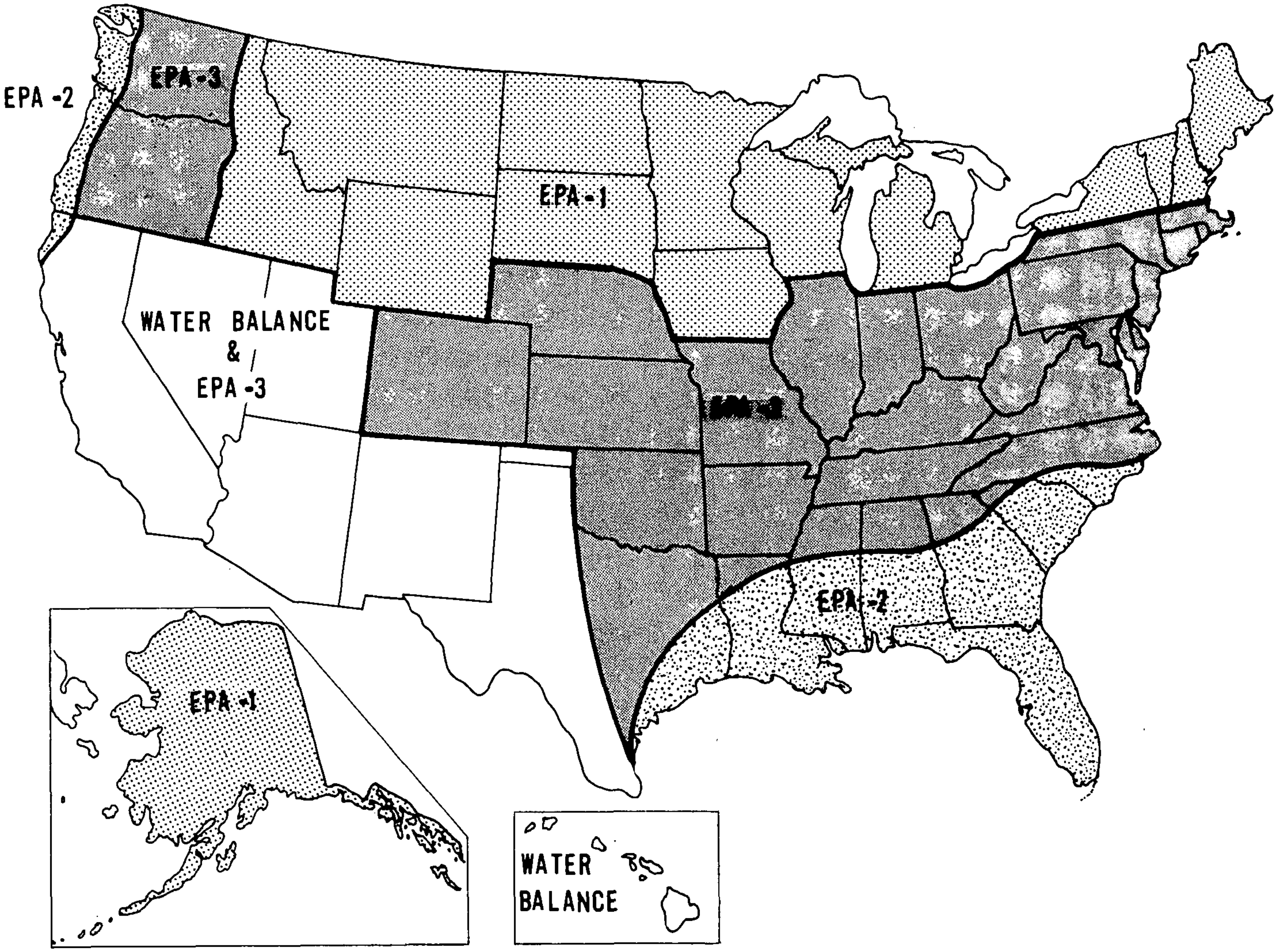
The validity of any one of the computer storage programs will depend on the presence of adequate data. The quality, completeness, and length of record are all important. Assigning threshold values and the confidence level of the output must be considered carefully in order to provide a realistic estimate of required storage. To use these programs, contact the National Climatic Center of the National Oceanic and Atmospheric Administration in Asheville, North Carolina 28801; a fee is required.

5.3.1.1 Determining Storage in Cold Climates

The operation of a slow rate or overland flow system is likely to be affected adversely if severe cold weather prevails for long periods (2 to 4 months). If annual crops are being irrigated, then the growing season will determine the storage requirement. However, if perennials, such as grasses and woodlands, are irrigated, then wastewater application will normally be stopped only by frozen soil conditions. The EPA-1 program computes a "freezing index" which provides a measure of the intensity and duration of cold periods that are likely to occur in the Northeast, the northern half of the Midwest, and parts of the Rocky Mountain area (see Figure 5-7). When the index reaches 200 to 300, the ground is assumed to be frozen and wastewater application is not recommended [39]. Limitations to the use of the freezing index include: lack of soil temperature data, yearly winter temperature variations, and differences in design and operating practices for existing land treatment systems.

FIGURE 5-7

DETERMINATION OF STORAGE BY EPA COMPUTER PROGRAMS AND WATER BALANCE ACCORDING TO CLIMATIC CONSTRAINTS [11, 38]



EPA-1 also computes storage based on the favorable/unfavorable day analysis by assigning threshold values to (1) mean daily temperature, (2) daily precipitation, and (3) snow cover. The storage requirement is increased by one day's flow on days designated as unfavorable according to the threshold values. On a favorable day, storage is reduced by a fraction of one day's flow--the drawdown rate. A range of common threshold values used as input to the EPA-1 program is listed in Table 5-15.

TABLE 5-15

THRESHOLD VALUES FOR THE EPA-1 STORAGE PROGRAM

Parameter	Favorable day threshold values
Mean daily temperature, °F	>25-32
Daily precipitation, in.	<0.5-1.0
Snow cover, in.	<1.0
Drawdown rate, % of average flow	50-25

1°F = 1.8°C + 32
1 in. = 2.54 cm

5.3.1.2 Determining Storage in Moderate Climates

To estimate storage for moderate climatic zones where winter conditions are less severe such as the mid-Atlantic states (see Figure 5-7), EPA-3 is recommended. This program is more flexible than EPA-1 in that minimum and maximum daily temperatures are examined instead of the mean daily temperature. Both temperature thresholds must be reached for a day to be favorable. However, if the maximum daily temperature is exceeded, but the minimum temperature is below the lower threshold, the program assumes that it is warm enough to permit operation during a portion of the day, i.e., a partly favorable day. EPA-3 is organized so that on partly favorable days, storage is automatically increased by some fraction of the daily flow. The precipitation and snow cover thresholds and the drawdown rate act as they do for EPA-1. Weather station data for the above parameters are examined during the months of November through April for the available period of record (20 years minimum).

The drawdown rate, as used in EPA-3, is the amount of water applied on favorable days in addition to the average daily flow. In moderate climates, this parameter can significantly reduce storage requirements but at the expense of increasing land requirements. Availability and cost of the additional land will determine how much wastewater can be applied from storage.

5.3.1.3 Determining Storage in Wet Climates

In wet regions where a high percentage of the annual precipitation is in the form of rain and the mean daily temperature seldom drops below 32°F (0°C), the EPA-2 storage program should be used. EPA-2 accounts for prolonged wet periods where rain can occur almost daily between the months of November and April. Regions where prolonged wet spells limit the application of wastewater are at locations along the Gulf states and the Pacific Northwest Coastal Region. Daily climatological data are examined in an attempt to identify days when the soil is saturated and an application of wastewater would result in unwanted runoff. Any day where runoff occurs is defined as unfavorable and considered a storage day.

The EPA-2 program is an outgrowth of work by Palmer to determine conditions of meteorological drought for agricultural purposes [40]. The rate at which excess soil moisture is depleted from the soil (the soil drying rate) is approximated by the EPA-2 program to account for the long-term effects of extremely heavy rainfall. The program can be modified to suit different soil conditions.

5.3.1.4 Determining Storage in Warm Climates

In warm climates, such as the semiarid and arid southwest United States, the climatic constraints to application of wastewater are usually very small (1 to 5 days). In these situations total storage may depend on the balance between water supply and application rate and the amount of certain wastewater constituents, e.g., nitrogen that can be applied to the soil without exceeding groundwater quality restrictions.

An irrigation water balance, which also considers nitrogen loading, can be used to estimate cumulated storage in warm climates for slow rate systems. The water balance consists of the elements in the following equation:

$$\begin{matrix}\text{Design} \\ \text{precipitation}\end{matrix} + \begin{matrix}\text{Wastewater} \\ \text{applied}\end{matrix} = \text{Evapotranspiration} + \text{Percolation} + \text{Runoff} \qquad (5\text{-}4)$$

A monthly evaluation of water balance is suggested due to seasonal variations in component factors. Of all the factors, precipitation is the most unpredictable. A range of values that might be encountered can be established on the basis of a frequency analysis of wetter-than-normal years (the wettest in 10, 20, or 25 years may be reasonable).

When using the water balance to estimate storage, the recurrence interval for precipitation and evapotranspiration should be the same.

An example of a monthly irrigation water balance to determine storage requirements for a 1 Mgal/d (43.8 L/s) slow rate system is shown in Table 5-16. This water balance assumed (1) precipitation and evapotranspiration data for the wettest year in 25, (2) nitrogen is limiting and separate calculations show that 120 acres (48 ha) are required, (3) a perennial grass is grown and irrigated year-round, (4) tailwater runoff from surface application is contained and reapplied, and (5) the storage reservoir is empty at the beginning of the water year.

The maximum storage would be 22.7 in. (58 cm) in the month of March, calculated for an application area of 120 acres (48 ha), yielding the required storage volume of 227 acre-ft (280 000 m^3) or 74 days flow at 1 Mgal/d (43.8 L/s).

5.3.1.5 Irrigation and Consumptive Use Requirements

In mild climates, storage requirements could be governed by the management of crops that are to be grown. The irrigation and consumptive use requirements shown in Table 5-17 for the Bakersfield, California, slow rate system illustrate how storage is affected by crop selection. It should be pointed out that only a portion of the applied wastewater is actually consumed by the crops (or lost by evapotranspiration). Some of the wastewater will be lost by seepage from irrigation ditches, from surface runoff, and by deep percolation below the root zone in the field. This is reflected in Table 5-17, where irrigation requirements are shown to be greater than consumptive use values. The U.S. Department of Agriculture has estimated that on the average about 47% of the irrigation water enters the soil and is held in the root zone where it is available to crops. It also points out that it is possible to attain irrigation efficiencies of 70 to 75% by proper selection, design, and operation of the irrigation system, including provision for tailwater return [42].

There are several months listed in Table 5-17 when there is no irrigation requirement although consumptive use is indicated. In these cases, it is assumed that crop water needs are being supplied by effective growing season precipitation and carryover soil moisture from winter rains, or pre-irrigation.

TABLE 5-16

EXAMPLE OF STORAGE DETERMINATION FROM A WATER BALANCE FOR IRRIGATION [41]
Inches

Month (1)	Evapotrans-piration[a] (2)	Allowable percolation[b] (3)	Water losses[c] (2)+(3) =(4)	Precipitation[a] (5)	Water deficit[d] (4)-(5) =(6)	Wastewater available[e] (7)	Change in storage[f] (7)-(6) =(8)	Total storage (9)
Oct	2.3	10.0	12.3	1.6	10.7	9.3	-1.4	0
Nov	1.0	10.0	11.0	2.4	8.6	9.3	0.7	0.7
Dec	0.5	5.0	5.5	2.7	2.8	9.3	6.5	7.2
Jan	0.2	5.0	5.2	3.0	2.2	9.3	7.1	14.3
Feb	0.3	5.0	5.3	2.8	2.5	9.3	6.8	21.1
Mar	1.1	10.0	11.1	3.4	7.7	9.3	1.6	22.7
Apr	3.0	10.0	13.0	3.0	10.0	9.3	-0.7	22.0
May	3.5	10.0	13.5	2.1	11.4	9.3	-2.1	19.9
Jun	4.8	10.0	14.8	1.0	13.8	9.3	-4.5	15.4
Jul	6.0	10.0	16.0	0.5	15.5	9.3	-6.2	9.2
Aug	5.7	10.0	15.7	1.1	14.6	9.3	-5.3	3.9
Sep	3.9	10.0	13.9	2.0	11.9	9.3	-2.6	1.3
Total annual	32.3	105.0	137.3	25.6	111.7	111.8		

a. Precipitation and evapotranspiration data are entered into columns 5 and 2, respectively.

b. On the basis of the nutrient balance to satisfy groundwater quality standards, the design allowable percolation rate is 10 in./month from March through November and 5 in./month for the remaining months (column 3).

c. The water losses (column 4) are found by summing evapotranspiration and percolation.

d. The water deficit (column 6) is the difference between the water losses and the precipitation.

e. The wastewater available per month (column 7) is

$$\text{Wastewater available} = \frac{1 \text{ Mgal/d} \times 30.4 \text{ d/month} \times 36.8 \text{ acre-in./Mgal}}{120 \text{ acres}}$$

f. The monthly change in storage (column 8) is the difference between the wastewater available (column 7) and the monthly water deficit (column 6).

1 in. - 2.54 cm
1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha

TABLE 5-17

IRRIGATION AND CONSUMPTIVE USE REQUIREMENTS FOR SELECTED CROPS AT BAKERSFIELD, CALIFORNIA [43, 44]
Depth of Water in Inches

Month	Pastures or alfalfa[a]		Double crop barley and grain sorghum		Cotton[c]		Sugar beets[d]	
	Consumptive use	Irrigation requirements	Consumptive use	Irrigation requirements	Consumptive use	Irrigation requirements	Consumptive use	Irrigation requirements
Jan	0.9	1.2	1.0	....	....	..	....	....
Feb	2.0	2.7	2.0	....	....	15[e]	....	....
Mar	3.8	5.1	3.8	6.0	....	..	....	5.0
Apr	5.2	7.0	5.2	6.0	0.6	..	1.0	9.0
May	7.0	9.4	2.6	....	1.2	..	2.5	5.0
Jun	8.6	11.5	....	10.0	3.6	5	5.0	9.0
Jul	9.4	12.6	4.5	7.0	7.2	12	7.0	7.5
Aug	8.7	11.7	8.0	12.0	8.4	12	8.0	4.5
Sep	5.8	7.8	6.0	9.0	6.0	..	....	....
Oct	4.3	5.8	3.0	....	2.5	..	....	....
Nov	2.0	2.7	....	....	....	..	....	6.0[f]
Dec	1.0	1.3	1.0	10.0	....	..	....	....
Total	58.7	78.8	37.1	60.0	29.5	44	23.5	46.0

a. Estimated maximum consumptive use (evapotranspiration) of water by mature crops with nearly complete groundcover throughout the year.

b. Barley planted in November-December, harvested in June. Grain sorghum planted June 20-July 10, harvested in November-December.

c. Rooting depth of mature cotton: 6 ft. Planting dates: March 15 to April 20. Harvest: October, November, and December.

d. Rooting depth: 5 to 6 ft. Planting date: January. Harvest: July 15 to September 10.

e. Pre-irrigation should wet soil to 5 to 6 ft depth prior to planting.

f. Pre-irrigation is used to ensure germination and emergence. First crop irrigations are heavy in order to provide deep moisture.

1 in. = 2.54 cm
1 ft = 0.305 m

5.3.2 Storage Reservoir Design

Most agricultural reservoirs are constructed of simple homogeneous (uniform materials) earth embankments, the design of which conforms to the principles of small dam design. Depending on the magnitude of the project, state regulations may govern the design. In California for example, any reservoir with embankments higher than 6 ft (1.8 m) and a capacity in excess of 50 acre-ft (61 800 m^3) is subject to state regulations on design and construction of dams, and plans must be reviewed and approved by the appropriate agency [45]. Design criteria and information sources are included in the U.S. Bureau of Reclamation publication, Design of Small Dams [46]. In many cases, it will be necessary that a competent soils engineer be consulted for proper soils analyses and structural design of foundations and embankments.

5.4 Distribution

The most common distribution techniques for land application fall within two major categories--surface and sprinkler--the selection of which depends on the objectives of the project and the limitations imposed by physical conditions such as topography, type of soil, crop requirements, and level of preapplication treatment.

Surface distribution employs gravity flow from piping systems or open ditches to flood the application area with several inches of water. Surface distribution is more suited to soils with moderate to low intake rates. Control of runoff is usually more of a consideration for surface distribution than for sprinkling, as applications of 2 in. (5 cm) or less by surface methods are difficult to apply uniformly. Graded land is essential to proper performance of a surface system.

Sprinkler distribution simulates rainfall and is less susceptible to topographic constraints than surface methods. It is particularly suited to irrigation of both highly permeable and highly impermeable soils. Sprinkler distribution may be used to irrigate most crops and, when properly designed, provides a more uniform distribution of water and greater flexibility in range of application rates than is available with surface distribution. Limitations to sprinkling include adverse wind conditions, clogging of nozzles with solids, and preapplication treatment requirements. Sprinkling also involves a significant utilization of equipment and its capital costs are significantly higher than those for surface distribution.

For all types of distribution systems, the maximum flow requirement of a given system and field area is referred to as the system capacity, which is computed by the formula:

$$Q = \frac{CAD}{FH} \tag{5-5}$$

where Q = discharge capacity, gal/min (L/s)
C = constant, 453 (28.1)
A = field area, acres (ha)
D = gross depth of application, in. (cm)
F = number of days to complete one cycle
H = number of operating hours

The system capacity is useful for determining mainline sizes, pump capacities, storage requirements, and operating time requirements. If several sites are involved with different loading requirements, system capacities must be computed separately within the same period of time to determine total system capacity.

5.4.1 Surface Systems

Surface distribution methods include ridge and furrow irrigation, surface flooding (border strip) irrigation, infiltration basins, and, overland flow. The distinguishing physical features of these methods are illustrated in Figure 5-8. Variations of methods employed in crop irrigation and the suitability of each to conditions of use are summarized in Table 5-18. Similar criteria for the surface application methods not normally associated with crop irrigation are summarized in Table 5-19.

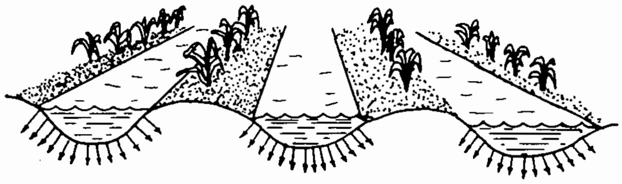
5.4.1.1 Ridge and Furrow Irrigation

Ridge and furrow irrigation consists of running irrigation streams along small channels (furrows) bordered by raised beds (ridges) upon which crops are grown. Furrows may be level or graded, straight or contoured. A similar method is corrugation irrigation, which consists of furrows excavated from the surface without creating raised beds. To simplify this presentation, only straight, graded ridge and furrow irrigation will be referred to hereafter, as its design considerations are applicable to all these methods.

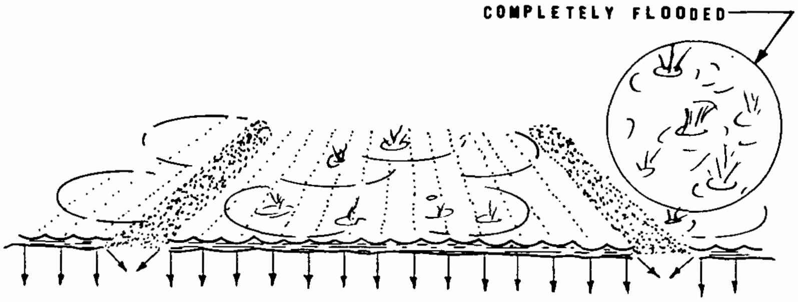
Intake characteristics for furrow irrigation are distinguished from those for border and sprinkler irrigation because the water only partially covers a given field area, and moves both downward and outward. Intake characteristics are best determined by inflow-outflow

FIGURE 5-8

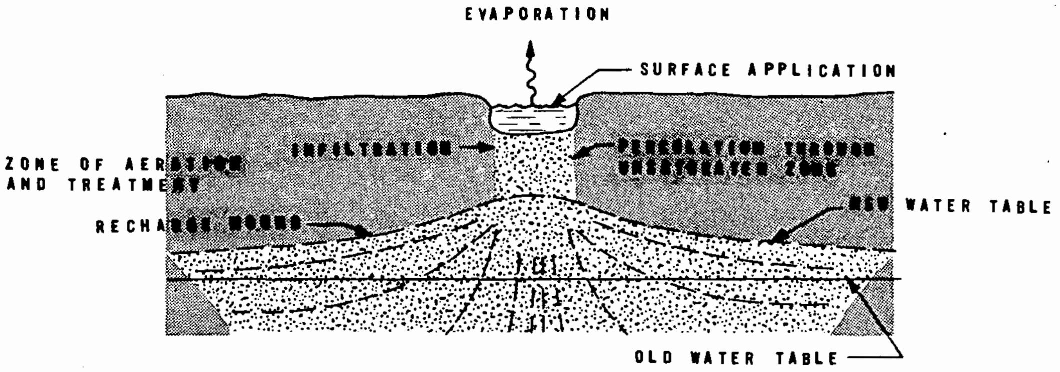
SURFACE DISTRIBUTION METHODS



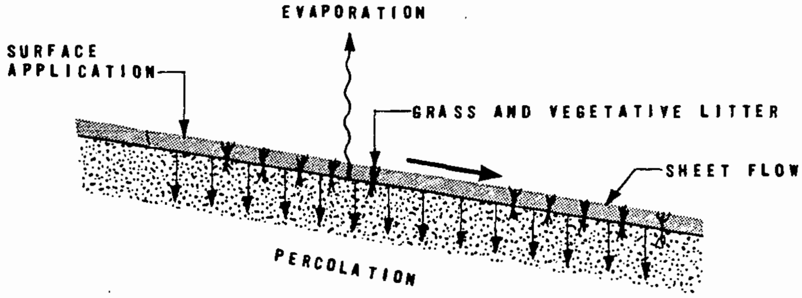
(a) RIDGE AND FURROW IRRIGATION



(b) FLOODING (BORDER STRIP) IRRIGATION



(c) RAPID INFILTRATION



(d) OVERLAND FLOW

TABLE 5-18

SURFACE IRRIGATION METHODS AND CONDITIONS OF USE [47]

Irrigation method	Suitabilities and conditions of use				Remarks
	Crops	Topography	Water quantity	Soils	
Flooding					
Small rectangular basins	Grain, field crops, orchards, rice	Relatively flat land; area within each basin should be leveled.	Can be adapted to streams of various sizes	Suitable for soils of high or low intake rates; should not be used on soils that tend to puddle	High installation costs. Considerable labor required for irrigating. When used for close-spaced crops, a high percentage of land is used for levees and distribution ditches. High efficiencies of water use possible.
Large rectangular basins	Grain, field crops, rice	Flat land; must be graded to uniform plane	Large flows of water	Soils of fine texture with low intake rates	Lower installation costs and less labor required for irrigation than small basins. Substantial levees needed.
Contour checks	Orchards, grain, rice, forage crops	Irregular land, slopes less than 2%	Flows greater than 1 ft^3/s	Soils of medium to heavy texture that do not crack on drying	Little land grading required. Checks can be continuously flooded (rice), water ponded (orchards), or intermittently flooded (pastures).
Narrow borders up to 16 ft wide	Pasture, grain, alfalfa, vineyards, orchards	Uniform slopes less than 7%	Moderately large flows	Soils of medium to heavy texture	Borders should be in direction of maximum slope. Accurate cross-leveling required between guide levees.
Wide borders up to 100 ft wide	Grain, alfalfa, orchards	Land graded to uniform plane with maximum slope less than 0.5%	Large flows, up to 20 ft^3/s	Deep soils of medium to fine texture	Very careful land grading necessary. Minimum of labor required for irrigation. Little interference with use of farm machinery.

TABLE 5-18
(Concluded)

Irrigation method	Suitabilities and conditions of use: Crops	Topography	Water quantity	Soils	Remarks
Benched terraces	Grain, field crops,	Slope up to 20%	Streams of small to medium size	Soils must be sufficiently deep that grading operations will not impair crop growth	Care must be taken in constructing benches and providing adequate drainage channel for excess water. Irrigation water must be properly managed. Misuse of water can result in serious soil erosion.
<u>Furrow</u>					
Straight furrows	Vegetables, row crops, orchards, vineyards	Uniform slopes not exceeding 2% for cutivated crops	Flows up to 12 ft^3/s	Can be used on all soils if length of furrows is adjusted to type of soil	Best suited for crops that cannot be flooded. High irrigation efficiency possible. Well adapted to mechanized farming.
Graded contour furrows	Vegetables, field crops, orchards, vineyards	Undulating land with slopes up to 8%	Flows up to 3 ft^3/s	Soils of medium to fine texture that do not crack on drying	Rodent control is essential. Erosion hazard from heavy rains or water breaking out of furrows. High labor requirement for irrigation.
Corrugations	Close-spaced crops such as grain, pasture, alfalfa	Uniform slopes of up to 10%	Flows up to 1 ft^3/s	Best on soils of medium to fine texture	High water losses possible from deep percolation or surface runoff. Care must be used in limiting size of flow in corrugations to reduce soil erosion. Little land grading required.
Basin furrows	Vegetables, cotton, maize, and other row crops	Relatively flat land	Flows up to 5 ft^3/s	Can be used with most soil types	Similar to small rectangular basins, except crops are planted on ridges.
Zizag furrows	Vineyards, bush berries, orchards	Land graded to uniform slopes of less than 1%	Flows required are usually less than for straight furrows	Used on soils with low intake rates	This method is used to slow the flow of water in furrows to increase water penetration into soil.

1 ft^3/s = 0.028 m^3/s
1 ft = 0.305 m

TABLE 5-19

NONIRRIGATING SURFACE APPLICATION METHODS AND CONDITIONS OF USE

Application method	Suitabilities and conditions of use				Remarks
	Vegetation	Topography	Water supply	Soils	
Rapid infiltration	Perennial grasses	Relatively flat to irregular	May be relatively large, soils permitting	Coarse texture and high infiltration rates	Water applied on intermittent basis to maintain permeability. Often less land preparation than most irrigation systems.
Overland flow	Perennial grasses[a]	Uniformly graded with slopes from 2 to 8%	Moderately large flows with high percentage of runoff	Limited permeability	Land must be smooth to achieve sheet flow without ponding.

a. Suitable for continuously wet-root conditions.

measurements in the field. Design application rates are then based on these results. Furrow intake rates are usually expressed as flowrate (gal/min, L/s) per unit length (100 ft, 100 m) of furrow. Application rates are usually expressed as flowrate per furrow, or furrow stream size.

Other factors of critical importance for design of ridge and furrow irrigation are: furrow stream size [48, 49, 50], furrow length (Table 5-20), furrow slope [47], and furrow spacing (Table 5-21).

TABLE 5-20

SUGGESTED MAXIMUM LENGTHS OF CULTIVATED FURROWS FOR DIFFERENT SOILS, SLOPES, AND DEPTHS OF WATER TO BE APPLIED [47]
Feet

	Avg depth of water applied, in.											
	Clays				Loams				Sands			
Furrow slope, %	3	6	9	12	2	4	6	8	2	3	4	5
0.05	1 000	1 300	1 300	1 300	400	900	1 300	1 300	200	300	500	600
0.1	1 100	1 400	1 500	1 600	600	1 100	1 400	1 500	300	400	600	700
0.2	1 200	1 500	1 700	2 000	700	1 200	1 500	1 700	400	600	800	1 000
0.3	1 300	1 600	2 000	2 600	900	1 300	1 600	1 900	500	700	900	1 300
0.5	1 300	1 600	1 800	2 400	900	1 200	1 500	1 700	400	600	800	1 000
1.0	900	1 300	1 600	1 900	800	1 000	1 200	1 500	300	500	700	800
1.5	800	1 100	1 400	1 600	700	900	1 100	1 300	250	400	600	700
2.0	700	900	1 100	1 300	600	800	1 000	1 100	200	300	500	600

1 ft = 0.305 m
1 in. = 2.54 cm

TABLE 5-21

OPTIMUM FURROW OR CORRUGATION SPACING [49]

Soil condition	Optimum spacing, in.
Coarse sands - uniform profile	12
Coarse sands - over compact subsoils	18
Fine sands to sandy loams - uniform	24
Fine sands to sandy loams - over more compact subsoils	30
Medium sandy-silt loam - uniform	36
Medium sandy-silt loam - over more compact subsoils	40
Silty clay loam - uniform	48
Very heavy clay soils - uniform	36

1 in. = 2.54 cm

The distribution systems most commonly used for ridge and furrow irrigation consist of open ditches with siphon pipes (see Figure 5-9), or gated surface piping system (see Figure 5-10). The open ditch system may be supplied by distribution ditches or canals with turnouts, or by buried pipelines with valved risers. Gated surface piping systems generally consist of aluminum pipe with multiple gated outlets, one per furrow. The pipe is connected to hydrants which are secured to valved risers from underground piping systems.

FIGURE 5-9

PLASTIC SIPHON TUBE FOR FURROW IRRIGATION

5.4.1.2 Surface Flooding Irrigation

Surface flooding irrigation consists of directing a sheet flow of water along border strips, or cultivated strips of land bordered by small levees. This method is particularly suited to close-growing crops such as grasses that can tolerate periodic inundation at the ground surface. The border strips usually have slight, if any, cross slopes, and may be level or graded in the direction of flow. Border strips may also be straight or contoured. For purposes of illustration, only straight, graded border irrigation will be included in the design considerations that follow. Detailed design procedures developed by the SCS for the various types of borders are given in Chapter 4, Section 15 of the SCS Engineering Handbook [51].

FIGURE 5-10

ALUMINUM HYDRANT AND GATED PIPE FOR FURROW IRRIGATION

Application rate for border irrigation is dependent on the soil intake rate and physical features of the strip. Water is applied in the same manner as in ridge and furrow irrigation. However, the stream is normally shut off when it has advanced about 75% of the length of the border. The objective is to have sufficient water remaining on the border after shutoff to irrigate the remaining length of border to the proper depth with very little runoff. Theoretically, it is possible to apply the water nearly uniformly along the border using this technique. However, actually achieving uniform distribution with minimal runoff requires a good deal of skill and experience on the part of the operator. Minimization of runoff is somewhat less critical when tailwater return systems are used.

The widths of border strips are often selected for compatibility with farm implements, but they also depend to a certain extent upon slope, which affects the uniformity of distribution across the strip. A guide for estimating strip widths based on grades in the direction of flow is presented in Table 5-22.

Other design factors for a border strip system are similar to those of ridge and furrow irrigation. These factors include intake characteristics [51], border strip lengths and slopes (Table 5-23 and 5-24). Another factor influencing design is surface roughness, which is a measure of resistance to flow caused by soil and vegetation.

TABLE 5-22

RECOMMENDED MAXIMUM BORDER STRIP WIDTH [51]

Irrigation grade, %	Maximum strip width, ft
0	200
0.0-0.1	120
0.1-0.5	60
0.5-1.0	50
1.0-2.0	40
2.0-4.0	30
4.0-6.0	20

1 ft = 0.305 m

TABLE 5-23

DESIGN STANDARDS FOR BORDER STRIP IRRIGATION, DEEP ROOTED CROPS [47]

Soil type and infiltration rate	Slope, %	Unit flow per foot of strip width, ft^3/s	Avg depth of water applied, in.	Border strip, ft	
				Width,	Length
Sandy, 1+ in./h	0.2-0.4	0.11-0.16	4	40-100	200-300
	0.4-0.6	0.09-0.11	4	30-40	200-300
	0.6-1.0	0.06-0.09	4	20-30	250
Loamy sand, 0.75-1 in./h	0.2-0.4	0.07-0.11	5	40-100	250-500
	0.4-0.6	0.06-0.09	5	25-40	250-500
	0.6-1.0	0.03-0.06	5	25	250
Sandy loam, 0.5-0.75 in./h	0.2-0.4	0.06-0.08	6	40-100	300-800
	0.4-0.6	0.04-0.07	6	20-40	300-600
	0.6-1.0	0.02-0.04	6	20	300
Clay loam, 0.25-0.5 in./h	0.2-0.4	0.03-0.04	7	40-100	600-1 000
	0.4-0.6	0.02-0.03	7	20-40	300-600
	0.6-1.0	0.01-0.02	7	20	300
Clay, 0.10-0.25 in./h	0.2-0.3	0.02-0.04	8	40-100	1 200+

1 ft^3/s = 28.3 L/s
1 in. = 2.54 cm
1 ft = 0.305 m

The distribution systems for surface flooding irrigation are basically the same as for ridge and furrow irrigation. A common practice in system layouts is to locate the vertical risers from buried lines at spacings equal to the border strip widths. Thus, one valve supplies

TABLE 5-24

DESIGN STANDARDS FOR BORDER STRIP IRRIGATION, SHALLOW ROOTED CROPS [47]

Soil profile	Slope, %	Unit flow per foot of strip width, ft^3/s	Avg depth of water applied, in.	Border strip, ft	
				Width	Length
Clay loam, 24 in. deep over permeable subsoil	0.15-0.6	0.06-0.08	2-4	15-60	300-600
	0.6-1.5	0.04-0.07	2-4	15-20	300-600
	1.5-4.0	0.02-0.04	2-4	15-20	300
Clay, 24-in. deep over permeable subsoil	0.15-0.6	0.03-0.04	4-6	15-60	600-1 000
	0.6-1.5	0.02-0.03	4-6	15-20	600-1 000
	1.5-4.0	0.01-0.02	4-6	15-20	600
Loam, 6-18 in. deep over hardpan	1.0-4.0	0.01-4.0	1-3	15-20	300-1 000

1 ft^3/s = 28.3 L/s
1 in. = 2.54 cm
1 ft = 0.305 m

each strip, and is preferably located midway between the borders to provide uniform distribution across the strip (see Figure 5-11). For strips having widths greater than 30 ft (9.1 m), at least two outlets per strip will ensure good distribution uniformity. Use of gated pipe provides much more uniform distribution at the head of border strips and allows the flexibility of easily changing to ridge and furrow irrigation if crop changes are desired.

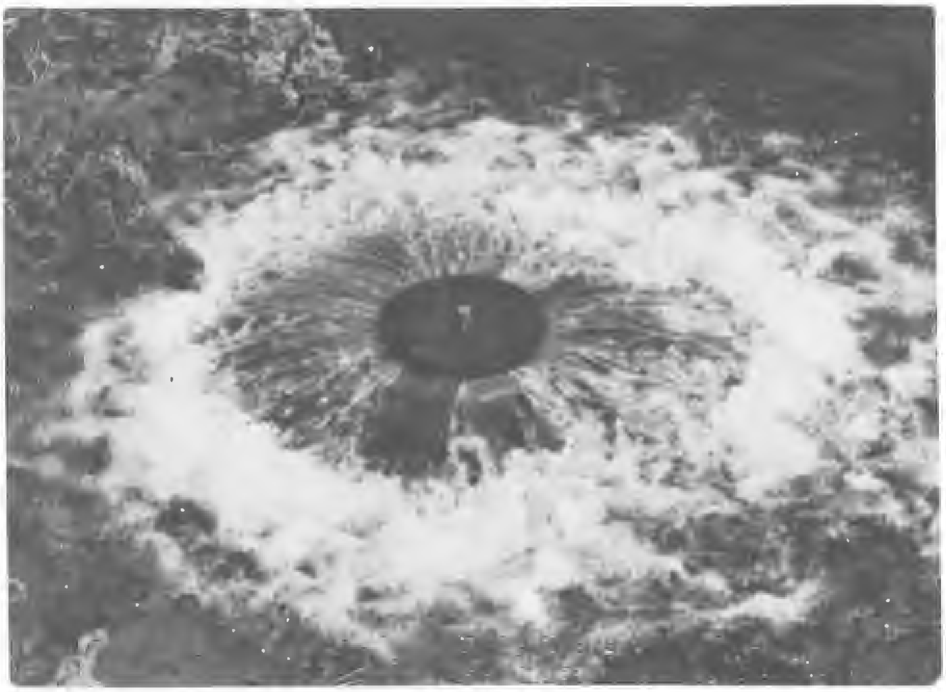
5.4.1.3 Rapid Infiltration Basins

The design of rapid infiltration basins depends on topography; when subsurface flow is to a surface water body, the basin shape and the elevation difference between the basins and the surface water are important. The basins are usually formed by constructing earthen dikes or by excavation.

Control of subsurface flow and recovery of renovated water are essential considerations for proper design of a rapid infiltration systems. If discharge to permanent groundwater is not feasible, a recovery system should be planned to withdraw the renovated water and reuse it for irrigation or recreation or discharge it to surface waters. Methods of recovery include underdrainage systems, pumped withdrawal, and natural drainage to surface waters.

FIGURE 5-11

OUTLET VALVE FOR BORDER STRIP APPLICATION



Where natural subsurface drainage to surface water is planned, the groundwater table must be controlled to prevent groundwater mounding. The aquifer should be able to readily transmit the renovated water away from the infiltration site. Bouwer [52] suggests the following equation for determining the required elevation difference between the water course and the spreading basin.

$$WI = KDH/L \tag{5-6}$$

where W = width of infiltration area, ft (m)

I = hydraulic loading rate, ft/d (m/d)

K = permeability of aquifer, ft/d (m/d)

D = average thickness of aquifer below water table perpendicular to flow direction, ft (m)

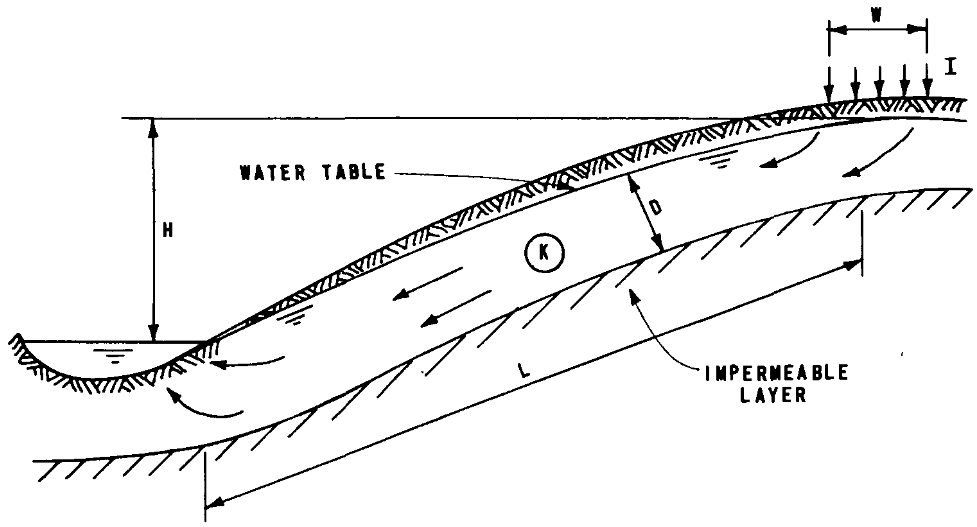
H = elevation difference between water level in stream or lake and maximum allowable water table level below infiltration area, ft (m)

L = distance of lateral flow, ft (m)

The relationships of these parameters are indicated in Figure 5-12. The product WI defines the amount of the applied water for a given section and thereby controls the infiltration basin sizing. Thus, if the amount of applied water is controlled by groundwater considerations, relatively large hydraulic loading rates (I) may be employed by utilizing relatively narrow (W) basins.

FIGURE 5-12

NATURAL DRAINAGE OF RENOVATED WATER INTO SURFACE WATER [52]



Basin sizing includes consideration of the amount of usable land available, the hydraulic loading rate, topography, and management flexibility. Sizing may also be influenced by groundwater considerations as discussed in the previous paragraph. In order to operate a system on a continuous basis, at least two basins will be required, one for flooding and one for drying, unless sufficient storage is available elsewhere in the system. Multiple basins are desirable to provide flexibility in the management of the system.

Basins should be relatively flat to allow uniform distribution of applied water over the surface. Thus, where sloping lands are to be utilized, terraced basins may be required. Cross slopes and longitudinal slopes should be on the order of those used for border irrigation. Basin widths and lengths are controlled by slopes, number of basins desired for management, distribution system hydraulics, and as previously discussed, water table restrictions.

The type of basin surface has been the subject of considerable debate and the relative advantages and disadvantages should be weighed on a case-by-case basis. The surface may consist of bare soil, or it may be covered with vegetation. The advantages of a vegetative cover include maintenance of infiltration rates, removal of suspended solids by filtration, additional nutrient removal if the vegetation is harvested, and possible promotion of denitrification. Among the disadvantages are increased basin maintenance, lower depth of application to avoid drowning the vegetation, and shorter periods of inundation to promote growth. At Flushing Meadows, it was found that a gravel covered surface reduced the infiltration capacity of a basin [53]. This was attributed to the mulching effect of the gravel, which prevented the drying of the underlying soil.

The distribution system for infiltration basins is often similar to that for surface irrigation, although sprinklers have been used. The purpose of the distribution system is to apply water at a rate which will constantly flood the basin throughout the application period at a relatively uniform depth. Effluent weirs may be used to regulate the depth of applied water. The discharge from the weirs is collected and distributed to holding ponds for recirculation, or to other infiltration basins. Water may be conveyed to the basins by pipeline or open channel systems. If equal flow distribution is intended for each basin, the distribution line or channel supplying the outlets to parallel basins should be sized so that hydraulic losses between the outlets will be insignificant. Outlets may be turnout gates from open channels or valved risers from underground piping systems. A basin outlet and splash pad are shown in Figure 5-13.

FIGURE 5-13

RAPID INFILTRATION BASIN INFLUENT STRUCTURE

5.4.1.4 Overland Flow

Overland flow distribution is accomplished by applying wastewater uniformly over relatively impermeable sloped surfaces which are vegetated. Although the most common method of distribution is with sprinklers, surface methods such as gated pipe or bubbling orifice may be used (Section 7.1.1). Gravel may be necessary to dissipate energy and ensure uniform distribution of water from these surface methods.

Slopes must be steep enough to prevent ponding of the runoff, yet mild enough to prevent erosion and provide sufficient detention time for the wastewater on the slopes. Experience at Paris, Texas, has indicated that best results are obtained with slopes between 2 and 6% [54]. A slope of 8%, used at Utica, Mississippi, is shown in Figure 5-14. Slopes must have a uniform cross slope and be free from gullies to prevent channeling and allow uniform distribution over the surface. The network of slopes and terraces that make up an overland flow system may be adapted to natural rolling terrain, as has been done at Napoleon, Ohio [11]. The use of this type of terrain will minimize land preparation costs.

FIGURE 5-14

OVERLAND FLOW SLOPE (8%) AT UTICA, MISSISSIPPI

5.4.1.5 Distribution System Design

Water is normally conveyed to surface distribution systems by canals (lined and unlined) or pipelines whose design standards are published by the American Society of Agricultural Engineers (ASAE). Design standards for flow control and measurement techniques are also included in the ASAE standards.

The methods of flow distribution to the fields include turnouts, siphon pipes, valved risers, gated surface pipe, and bubbling orifices. Turnouts are circular or rectangular openings which discharge flow directly from open ditches, canals, or open concrete pipe risers. Flow is controlled by slide gates, and discharge capacities are normally restricted to velocities of 3 ft/s (1 m/s) or less.

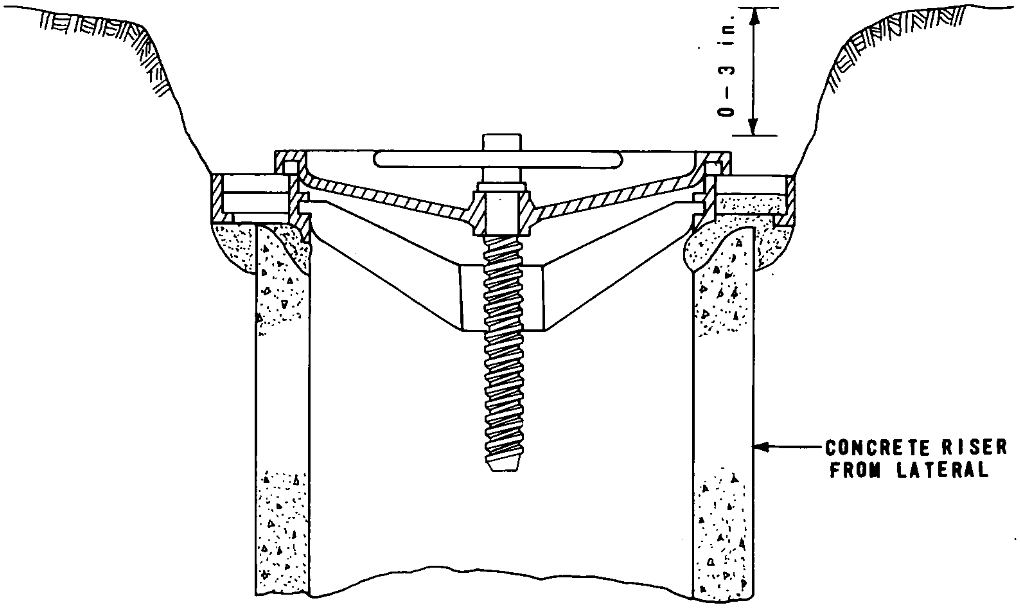
Siphon pipes are steel, aluminum, or plastic tubes (shown previously in Figure 5-9) used to siphon water from open ditches to supply furrows with irrigation water. Flow control is accomplished by combinations of pipe sizes or varying the number of pipes used. Although siphon pipes often require the least capital expenditure for distribution, operating demands are significant due to the amount of handling and the requirement for maintaining minimum water levels in the supply ditch to ensure continuity of flow.

Valved risers are vertical concrete pipe risers attached to buried concrete pipelines and are used for surface flooding irrigation or discharge to gated pipe hydrants. Flow is controlled by a simple wafer-shaped valve which is adjusted by a threaded stem. The more common valves are the alfalfa valve (mounted on top of the riser) and the orchard valve (mounted inside the riser). Typical cross-sections and capacities of these valves are shown in Figures 5-15 and 5-16.

Gated surface pipe, which is attached to aluminum hydrants, is aluminum pipe with multiple outlets. The pipe and hydrants are portable so that they may be moved for each irrigation. As described in the preceding paragraph, the hydrants are mounted on valved risers. Operating handles extend through the hydrants to control the alfalfa or orchard valves located in the risers. Control of flow is accomplished with slide gates or screw adjustable orifices at each outlet. The outlets are spaced to match furrow spacings and are usually fabricated to order. Gated outlet capacities vary with the available head at the gate, the velocity of flow passing the gate, and the gate opening. Typical gate capacities of standard gated pipe for various flow velocities are shown in Table 5-25. Hydrant spacings (and valved riser spacings) are controlled either by the losses in the gated pipe or by widths of border strips when border and furrow methods are alternated.

FIGURE 5-15

ALFALFA VALVE CHARACTERISTICS



CROSS SECTIONAL VIEW

SIZES AND RECOMMENDED MAXIMUM DESIGN CAPACITIES

Inside diameter of riser, in.	Diameter of port, in.	Maximum design capacity	
		Usual low head, ft^3/s^a	High head, ft^3/s^b
6	6	0.8	1.6
8	8	1.4	2.8
10	10	2.2	4.4
12	12	3.1	6.3
14	14	4.3	8.6
16	16	5.6	11.2
18	18	7.1	14.2
20	20	8.7	17.5

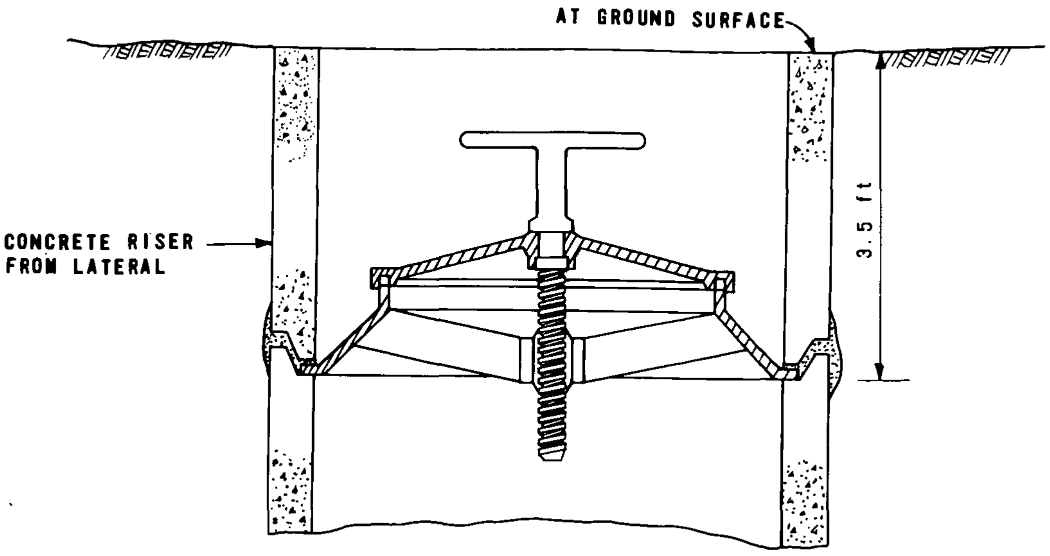
a. Recommended for minimum erosion with hydraulic gradient 1 ft above ground. Assumed 0.5 ft ponding over valve.

b. Can be used where higher pressures are available (hydraulic gradient 2.5 ft above ground) and precautions are taken to prevent erosion (ponding = 0.5 ft).

1 in. = 2.54 cm
1 ft^3/s = 0.284 m^3/s
1 ft = 0.305 m

FIGURE 5-16

ORCHARD VALVE CHARACTERISTICS [55]



SIZES AND RECOMMENDED MAXIMUM CAPACITIES

Inside diameter of riser, in.	Diameter of valve outlet, in.	Approximate design capacities, ft^3/s Low head[a]	Higher head[b]
6	1.5	0.04	0.08
6	2.5	0.12	0.23
6	3.5	0.23	0.45
6	6	0.67	1.34
8	5	0.46	0.93
8	8	1.18	2.37
10	6	0.67	1.34
10	6.5	0.78	1.57
10	10	1.85	3.71
12	8	1.18	2.37
12	12	2.67	5.35

a. Usual design with hydraulic gradient 1 ft above ground.

b. Higher head design with hydraulic gradient 2.5 ft above ground.

1 in. = 2.54 cm
1 ft^3/s = 0.0284 m^3/s
1 ft = 0.305 m

TABLE 5-25

DISCHARGE CAPACITIES OF SURFACE GATED PIPE OUTLETS [56]
Gallons per Minute

Velocity in pipe, ft/s	Head, ft	Gate opening					
		Full	3/4	1/2	1/4	1/8	1/16
0	1	48.8	35.7	22.8	10.6	5.0	2.3
	2	67.2	50.0	32.3	15.3	7.0	3.2
	3	80.1	60.8	39.1	18.3	8.5	3.8
	4	87.7	69.5	45.3	21.4	9.7	4.3
	5	94.2	77.1	50.6	23.7	10.7	4.8
1	1	45.3	32.8	21.3	10.2	4.9	2.2
	2	63.7	47.1	30.8	14.9	6.9	3.1
	3	76.6	57.9	37.6	17.9	8.4	3.7
	4	84.2	66.6	43.8	21.0	9.7	4.2
	5	90.7	74.2	49.1	23.3	10.7	4.7
3	1	40.0	26.7	18.2	8.8	4.3	2.1
	2	56.4	41.0	27.7	13.5	6.3	3.0
	3	69.3	51.8	34.5	16.5	7.8	3.5
	4	76.9	60.5	40.7	19.6	9.0	4.1
	5	83.4	68.1	46.0	21.9	10.0	4.6

1 gal/min = 0.063 L/s
1 ft/s = 0.305 m/s

Bubbling orifices are small diameter outlets from laterals used to introduce flow to overland flow systems or checks at low operating pressures. Such outlets may consist of orifices in the laterals or small diameter pipe stubs attached to the laterals. Outlets may range from 0.5 to 2 in. (1.3 to 5 cm) in diameter, the capacities of which are regulated by the available head.

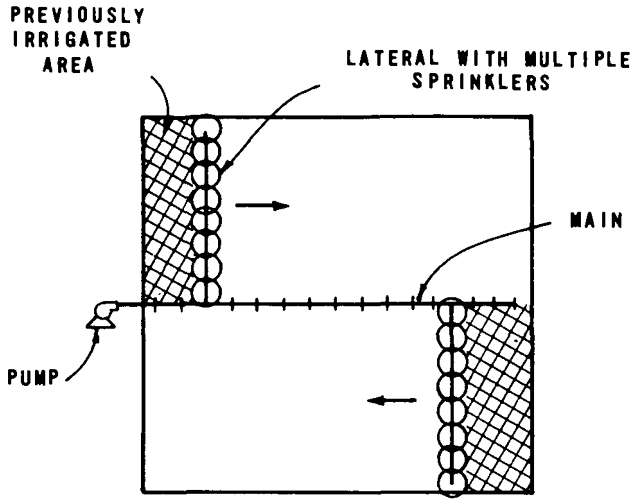
5.4.2 Sprinkler Systems

Sprinklers can be for all types of land treatment systems. The most common types of sprinklers may be categorized as hand moved, mechanically moved, and permanent set. The basic layout features of the various types of systems are depicted in Figures 5-17, 5-18, and 5-19.

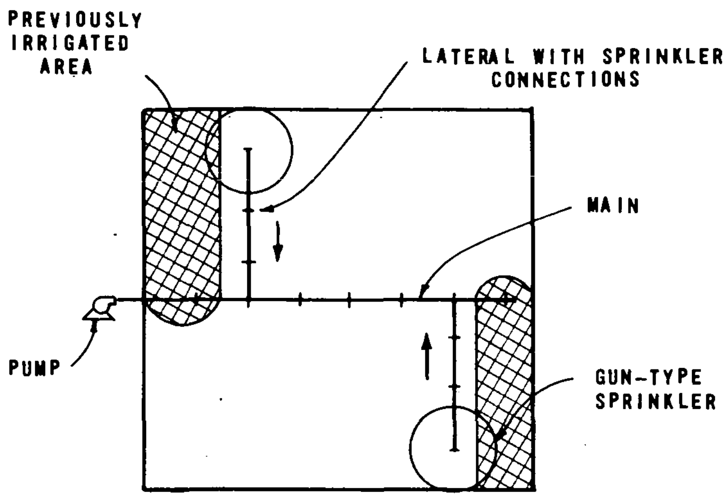
The more significant design considerations for sprinkler system selection include field conditions (shape, slope, vegetation, and soil type), climate, operating conditions (system management), and economics. These considerations are summarized in Table 5-26.

FIGURE 5-17

HAND MOVED SPRINKLER SYSTEMS



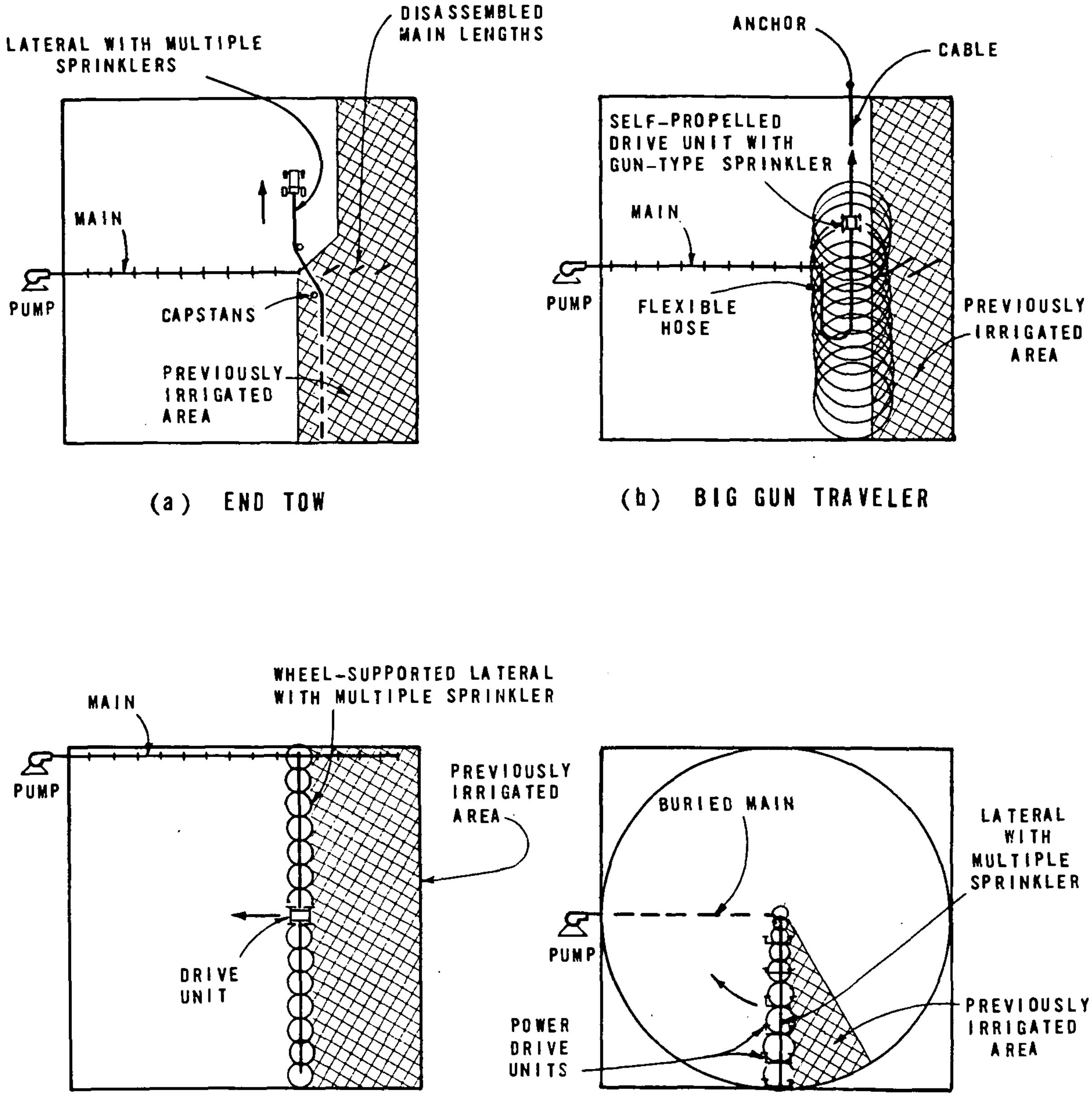
(a) PORTABLE PIPE



(b) STATIONARY BIG GUN

FIGURE 5-18

MECHANICALLY MOVED SPRINKLER SYSTEMS



(a) END TOW

(b) BIG GUN TRAVELER

(c) SIDE WHEEL ROLL

(d) CENTER PIVOT

FIGURE 5-19

PERMANENT SOLID SET SPRINKLER SYSTEM

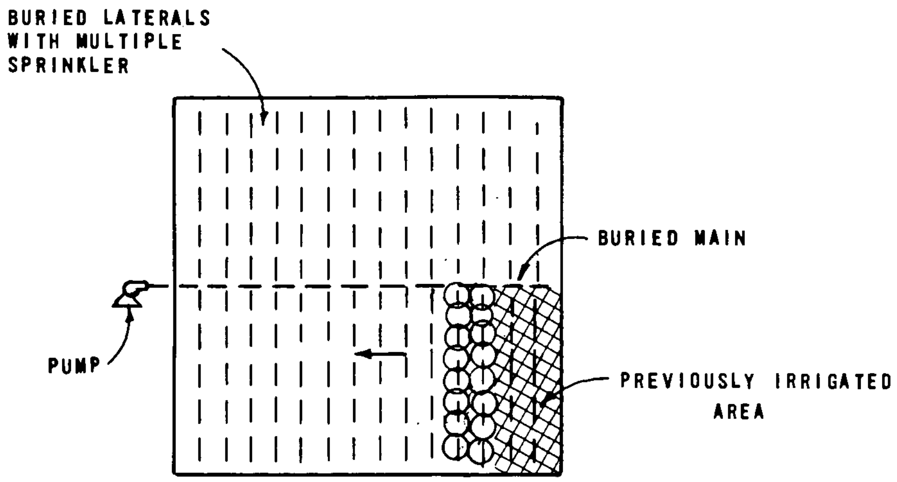


TABLE 5-26

SPRINKLER SYSTEM CHARACTERISTICS [57, 58]

	Typical application rate, in./h	Outlets per lateral	Nozzle pressure range, lb/in.^{2}	Size of single system, acres	Shape of field	Maximum slope, %	Maximum crop height, ft
Hand moved							
Portable pipe	0.1-2.0	Multiple	30-60	1-40	Any shape	20	....
Stationary gun	0.25-2.0	Single	50-100	20-40	Any shape	20	....
Mechanically moved							
End tow	0.1-2.0	Multiple	30-60	20-40	Rectangular	5-10	....
Traveling gun	0.25-1.0	Single	50-100	40-100	Rectangular	Unlimited	....
Side wheel roll	0.10-2.0	Multiple	30-60	20-80	Rectangular	5-10	3-4
Center pivot	0.20-1.0	Multiple	15-60	40-160	Circular[a]	5-15	8-10
Permanent							
Solid set	0.05-2.0	Multiple	30-60	Unlimited	Any shape	Unlimited	....

a. Travelers are available to allow irrigation of any shape field.

1 in./h = 2.54 cm/h
1 lb/in.^{2} = 0.69 N/cm^2
1 acre = 0.405 ha
1 ft = 0.305 m

5.4.2.1 Hand Moved Systems

Hand moved sprinkler systems include portable pipe and stationary gun systems. As the name implies, each is placed and removed manually for each irrigation set or period.

Portable pipe systems are surface pipe systems consisting of lateral lines which are moved between sets (piping position for one application) and a main line which may also be moved, or it may be permanent. The laterals are usually constructed of aluminum pipe in 30 or 40 ft (9 or 12 m) lengths with sprinklers mounted on risers extending from the laterals. Riser heights are determined by crop heights and angle of spray. In general, lateral spacings and sprinkler spacings are located at approximately equal intervals, usually ranging from 40 to 90 ft (12 to 27 m). Sprinklers may operate at a wide range of pressures and application. If sufficient pipe is available so that movement between sets is not required, the system is referred to as solid set.

The major advantages of portable systems include low capital costs and adaptability to most field conditions and climates. They may also be removed from the fields to avoid interferences with farm machinery. The principal disadvantage is the extensive labor requirement to operate the system.

Stationary gun systems are wheel-mounted or skid-mounted single sprinkler units (see Figure 5-20), which are moved manually between hydrants located along the laterals. Since the sprinkler operates at greater pressures and flowrates than multiple sprinkler systems, the irrigation time is usually shorter. After a set has been completed for the lateral, the entire lateral is moved to the next point along the main. In some cases a number of laterals and sprinklers may be provided to minimize movement of laterals.

The advantages of a stationary gun are similar to those of portable pipe systems with respect to capital costs and versatility. In addition, the larger nozzle of the gun-type sprinkler is relatively free from clogging. The drawbacks to this system are also similar to those for portable pipe systems in that labor requirements are high due to frequent sprinkler moves. Power requirements are relatively high due to high pressures at the nozzle, and windy conditions adversely affect distribution of the fine droplets created by the higher pressures.

5.4.2.2 Mechanically Moved Systems

The most common types of mechanically moved systems are end tow, traveling gun, side wheel roll, and center pivot. These systems may be moved after each irrigation by external drive mechanisms (tractors or winches) or integral drive units, or they may be self-propelled, continuous-moving systems.

FIGURE 5-20

STATIONARY GUN SPRINKLER MOUNTED ON A TRACTOR TRAILER

End tow systems are multiple-sprinkler laterals mounted on skids or wheel assemblies to allow a tractor to pull the lateral intact from one setting to the next. As indicated in Figure 5-18, the lateral is guided by capstans to control its alignment. The pipe and sprinkler design considerations are identical to those for portable pipe systems with the exception that pipe joints are stronger to accommodate the pulling requirements.

The primary advantages of an end tow system are relatively low labor requirements and overall system costs, and the capability to be readily removed from the field to allow farm implements to operate. Disadvantages include crop restrictions to movement of laterals and cautious operation to avoid crop and equipment damage.

Traveling gun systems are self-propelled, single sprinkler units which are connected to the supply source by a flexible hose (see Figure 5-21). The traveler is driven by a hydraulic or gas-driven winch located within the unit, or a gas-driven winch located at the end of the run. In both cases, a cable anchored at the end of the run guides the unit in a straight path. Variable speed drives are used to control application rates. Typical lengths of run are 660 or 1 320 ft (201 or 403 m), and spacings between travel lanes are commonly 330 ft (100 m). The rubber hose, which may be 2.5 to 5 in. (6.4 to 12.7 cm) is a specially-constructed item and may constitute a considerable portion of the total cost of the system.

FIGURE 5-21

TRAVELING GUN SPRINKLER

The more important advantages of a traveling gun system are low labor requirements and relatively clog-free nozzles. They may also be adapted to fields of somewhat irregular shape and topography. Disadvantages are high initial costs and power requirements, hose travel lanes required for most crops, and drifting of sprays in windy conditions.

Side wheel roll systems consist of aluminum or galvanized steel pipe laterals 4 to 5 in. (10.2 to 12.7 cm) in diameter, which act as axles for 5 to 7 ft (1.5 to 2.1 m) diameter wheels (see Figure 5-22). The end of the lateral, which is typically 1 320 ft (403 m) long, is connected to hydrants located along the main line. The system is moved between sets by an integral drive unit located at the center of the lateral. The unit is a gas-driven engine operated by the irrigator. The sprinklers, which have the same general characteristics as those for portable pipe systems, are mounted on swivel connections to ensure upright positions at all times. Sprinkler spacings are typically 30 or 40 ft (9.2 to 12.5 m) and wheel spacings may range from 30 to 100 ft (9.2 to 30.5 m). Side wheel laterals may be equipped with trail lines up to 90 ft (27 m) in length located at each sprinkler connection on the axle lateral. Each trail line has sprinklers mounted on risers spaced typically at 30 to 40 ft (9 to 12 m). Use of trail lines allows several lateral settings to be irrigated simultaneously and reduces the number of moves required to irrigate a field.

FIGURE 5-22

SIDE WHEEL ROLL SPRINKLER SYSTEM

The principal advantages of side wheel roll systems are relatively low labor requirements and overall costs, and freedom from interference with farm implements. Disadvantages include restrictions to crop height and field shape, and misalignment of the lateral caused by uneven terrain.

A <u>center pivot system</u> is a lateral with multiple sprinklers which is mounted on self-propelled, continuously moving tower units (see Figure 5-23) rotating about a fixed pivot in the center of the field. Water is supplied by a well or a buried main to the pivot, where power is also furnished. The lateral is usually constructed of 6 to 8 in. (15 to 20 cm) steel pipe 200 to 2 600 ft (61 to 793 m) in length. A typical system irrigates a 160 acre (64 ha) parcel (see Figure 5-24) with a 1 288 ft (393 m) lateral. The circular pattern reduces coverage to about 130 acres (52 ha), although systems with traveling end sprinklers are available to irrigate the corners.

FIGURE 5-23

CENTER PIVOT RIG

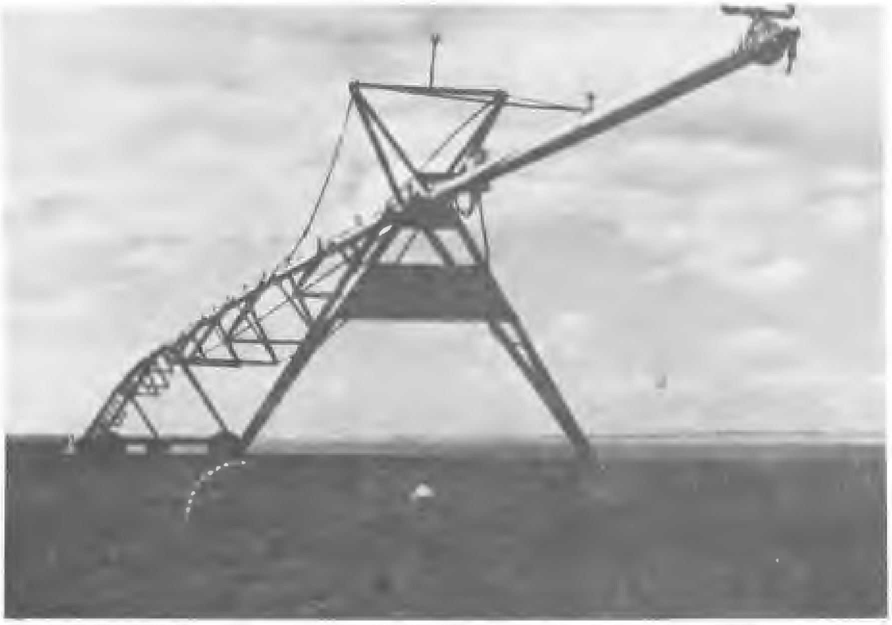


FIGURE 5-24

CENTER PIVOT IRRIGATION SYSTEM

The tower units are driven electrically or hydraulically and may be spaced from 80 to 250 ft (24 to 76 m) apart. The lateral is supported between the towers by cables or trusses. Control of the application rate is achieved by varying the running time of the tower motors. Variations in sprinkler sizes or spacings must be provided along the lateral for uniform distribution, since the area of coverage per sprinkler increases with the distance from the center. The relatively low application rates shown in Table 5-26 account for the fact that center pivot systems irrigate more frequently and at lower rates than other systems.

Another type of center pivot system is the rotating boom. This system eliminates the need for wheel-mounted power units by supporting the lateral with cables extending from a tower at the pivot. These systems have limited applications, as the area of coverage is small, up to 40 acres (16 ha), relative to conventional center pivot systems, and the corresponding per acre costs are high when multiple systems are required.

The main advantage of a center pivot system is the high degree of automation and a corresponding low requirement for labor. Limitations include restricted area of coverage (dead spaces in corners of fields), crop heights, and potential maintenance problems related to the numerous mechanical components.

5.4.2.3 Permanent Solid Set Systems

Permanent solid set systems are distinguished from portable solid set systems only in that the laterals are buried and constructed of plastic pipe instead of aluminum. Sprinkler selection and spacing criteria are identical. Risers may be fixed or removable to accommodate farm equipment. The primary advantages of solid set systems are low labor requirements and maintenance costs, and adaptability to all types of terrain, field shapes, and crops. They are also the most adaptable systems for climate control requirements. The major disadvantages are high installation costs and obstruction of fixed risers to farming equipment.

5.4.2.4 Overland Flow Systems

Sprinkler application for overland flow consists either of permanent solid set systems or rotating booms. These systems are distinguished from those for slow rate by their layout arrangements (single row of sprinklers) and application rates (designed for runoff). Sprinkler spacing is normally equal to the radius of the wetted circle. The sprinkler discharge rate, Q , may be computed as follows:

$$Q = \frac{I \times R \times L}{t \times C} \qquad (5\text{-}7)$$

where Q = nozzle discharge, gal/min (L/s)
I = total depth of water applied, in. (cm)
t = period of time to apply water, h
R = spacing between sprinklers, ft (m)
L = length of overland slope, ft (m)
C = constant = 96.3 (360)

Sprinkler heads may be arranged to avoid drifting of sprays at the expense of reducing the area of coverage. The primary objective of the distribution system is to concentrate the applied water at the upper ends of the slopes to produce runoff.

Fan nozzles may be used for overland flow distribution to minimize pumping pressure head and minimize aerosol formation. At Pauls Valley, Oklahoma (Section 7.11), fixed nozzles are being used. At Ada, Oklahoma, rotating booms are being used with fan nozzles on both ends as shown in Figure 5-25.

FIGURE 5-25

ROTATING BOOM, FAN SPRINKLER, ADA, OKLAHOMA

5.4.2.5 System Design

The procedure for sprinkler system design involves the determination of the optimum rate of application, sprinkler selection, sprinkler spacings and performance characteristics, lateral design, and miscellaneous requirements. Although the following discussions are limited to stationary systems, the general theory applies to moving systems as well. Detailed design requirements for specific systems may be obtained from equipment suppliers.

The optimum rate of application for a sprinkler system is the rate that ensures uniform distribution under prevailing climatic conditions without exceeding the basic intake rate of the soil (except for overland flow systems).

Sprinkler selection is primarily based on conditions of service, such as type of distribution system, pressure limitations, application rate, clogging potential, and effects of winds. Sprinklers used for application of wastewater are usually of the rotating head type with one or two nozzles. Special attention should be given to sprinkler design for low temperature winter operation. A general classification of sprinklers and their adaptabilities to various service conditions is presented in Table 5-27. More specific performance characteristics for the many types of sprinklers are available from the sprinkler manufacturers.

Sprinkler spacings and performance characteristics are jointly analyzed to determine the most uniform distribution pattern at the optimum rate of application. Distribution patterns of individual sprinklers are affected primarily by pressure--low pressures cause large drops which are concentrated in a ring a certain distance away from the sprinkler, whereas high pressures result in fine drops which fall near the sprinkler. These finer sprays are easily distorted by winds.

Since the amount of water applied by a sprinkler decreases with the distance from the nozzle and the distribution pattern is circular, sprinklers and laterals are spaced to provide overlapping of the wetted diameters. Spacings are normally related to the wetted diameters specified by the sprinkler manufacturers. These spacings may be determined empirically or by using published guidelines. The SCS recommends limiting sprinkler spacings along the lateral (S_L) to 50% or less of the wetted diameter, and lateral spacings along the main (S_M) to less than 65%. In windy areas, S_M should be reduced to 50% for velocities of 5 to 10 mi/h (2.2 to 4.4 m/s) and to 30% for velocities greater than 10 mi/h (4.4 m/s) [59]. For high pressure sprinklers, the SCS recommends a maximum diagonal distance between sprinklers of two-thirds the wetted diameter with similar deductions for wind as discussed for lower pressure sprinklers.

TABLE 5-27

CLASSIFICATION OF SPRINKLERS AND THEIR ADAPTABILITY [58]

Type of sprinkler	General characteristics	Range of wetted diameters, ft	Recommended minimum application rate, in./h	Moisture distribution pattern[a]	Adaptations and limitations
Low pressure, 5-15 $lb/in.^2$	Special thrust springs or reaction-type arms	20-50	0.40	Fair	Small acreages; confined to soils with intake rates exceeding 0.50 in./h and to good ground cover on medium- to coarse-textured soils
Moderate pressure, 15-30 $lb/in.^2$	Usually single-nozzle oscillating or long-arm dual-nozzle design	60-80	0.20	Fair to good at upper limits of pressure range	Primarily for undertree sprinkling in orchards; can be used for field crops and vegetables
Intermediate pressure, 30-60 $lb/in.^2$	Either single or dual-nozzle design	75-120	0.25	Very good	For all field crops and most irrigable soils; well-adapted to overtree sprinkling in orchards and groves and to tobacco shades
High pressure, 50-100 $lb/in.^2$	Either single of dual-nozzle design	110-230	0.50	Good except where wind velocities exceed 4 mi/h	Same as for intermediate pressure sprinklers except where wind is excessive
Hydraulic or giant, 80-120 $lb/in.^2$	One large nozzle with smaller supplemental nozzles to fill in pattern gaps; small nozzle rotating sprinkler	200-400	0.65	Acceptable in calm air; severely distorted by wind	Adaptable to close-growing crops that provide good ground cover; for rapid coverage and for odd shaped areas; limited to soils with high intake rates
Undertree low-angle, 10-50 $lb/in.^2$	Designed to keep stream trajectories below fruit and foliage by lowering the nozzle angle	40-90	0.33	Fairly good; diamond pattern recommended where laterals spaced more than one tree interspace	For all orchards or citrus groves; in orchards where wind will distort overtree sprinkler patterns; in orchards were available pressure is not sufficient for operation of overtree sprinklers
Perforated pipe, 4-20 $lb/in.^2$	Portable irrigation pipe with lines of small perforations in upper third of pipe perimeter	10-50[b]	0.50	Good pattern is rectangular	For low growing crops only; unsuitable for tall crops; limited to soils with relatively high intake rates; best adapted to small acreages of high value crops; low oeprating pressure permits use of gravity or municipal supply

a. Assuming proper spacing and pressure nozzle size relationships.

b. Rectangular strips.

1 ft = 0.305 m
1 in. = 2.54 cm
1 $lb/in.^2$ = 0.69 N/cm^2
1 mi/h = 0.44 m/s

Once the preliminary spacing has been determined, the nozzle discharge capacity to supply the optimum application rate is found by the equation

$$Q = \frac{S_L \times S_M \times I}{C} \qquad (5\text{-}8)$$

in which Q = flow rate from nozzle, gal/min (L/s)
S_L = sprinkler spacing along lateral, ft (m)
S_M = sprinkler spacing along main, ft (m)
I = optimum application rate, in./h (cm/h)
C = constant = 96.3 (360)

This establishes the basis for final sprinkler selection, which is a trial and adjustment procedure to match given conditions with performance characteristics of available sprinklers. The normal selection procedure is to assume a spacing and determine the nozzle discharge capacity. Manufacturers' data are then reviewed to determine the nozzle sizes, operating pressures, and wetted diameters of sprinklers operating at the desired discharge rate. The wetted diameters are then checked with the assumed spacings for conformance with the established spacing criteria.

Lateral design consists of selecting lateral sizes to deliver the total flow requirement of the lateral with friction losses limited to a predetermined amount. A general practice is to limit all hydraulic losses (static and dynamic) in a lateral to 20% of the operating pressure of the sprinklers. This will result in sprinkler discharge variations of about 10% along the lateral [58]. Since flow is being discharged from a number of sprinklers, the effect of multiple outlets on friction loss in the lateral must be considered. A simplified approach developed by Christiansen is to multiply the friction loss in the entire lateral at full flow (discharge at the distal end) by a factor based on the number of outlets. The factors for selected numbers of outlets are presented in Table 5-28. For long lateral lines, capital costs may be reduced by using two or more lateral sizes which will satisfy the head loss requirements.

System automation is receiving greater attention as labor costs increase. All of the systems described herein may be automatically controlled to some degree. The most common control devices are remote control valves energized electrically or pneumatically to start or stop flow in a lateral or main. The energy source for operating these valves may be activated manually at a push-button station or automatically by a time-controlled switch. In order to determine the economics of a control system, the designer must compare the costs of labor with the costs of controls at the desired level of operating flexibility.

TABLE 5-28

FACTOR (F) BY WHICH PIPE FRICTION LOSS IS MULTIPLIED TO OBTAIN ACTUAL LOSS IN A LINE WITH MULTIPLE OUTLETS [49]

No. of outlets	Value of F
1	1.000
2	0.634
3	0.528
4	0.480
5	0.451
6	0.433
7	0.419
8	0.410
9	0.402
10	0.396
15	0.379
20	0.370
25	0.365
30	0.362
40	0.357
50	0.355
100	0.350

5.5 Management of Renovated Water

5.5.1 General Considerations

5.5.1.1 Flow to Groundwaters

For rapid infiltration, an unsaturated soil zone is necessary to maintain desired infiltration rates since oxygen is usually depleted when inundation periods exceed 48 hours. However, good internal drainage must be present to reinstate an aerobic zone during the dry-up period. Bouwer reports that only 5 ft (1.5 m) of unsaturated soil need be maintained [52]. A deeper water table does not materially increase the depth of the aerobic zone since oxygen diffusion is slowed considerably below about 3 ft (1 m).

5.5.1.2 Stormwater Runoff Considerations

The quality of stormwater runoff is essentially unknown, but the nitrogen and phosphorus values given in Table 3-11, measured in rural stormwater runoff studies, should give perspective to the magnitude of the problem. The principal considerations are to minimize the quantity of runoff and to minimize the sediment load in the runoff. This can be accomplished for the most part by sound farm management practices.

Overland flow systems are designed to shed water and must be capable of handling storm runoff flows. It has been shown at Paris, Texas [54], that the effect of precipitation is to improve the quality of overland flow runoff as measured by electrical conductivity.

5.5.2 Underdrainage Systems

Underdrains are mainly associated with slow rate treatment but can also be used with rapid infiltration treatment. The underdrainage system must control the water table to provide sufficient soil detention time and underground travel distance if the desired quality of renovated water is to be achieved. In the case of slow rate treatment, the ability to plant, grow, and harvest a crop properly also depends on the drainage conditions. Skaggs has developed a model to manage water in soils with high groundwater [60, 61].

In arid regions, drains are usually placed at much greater depths and farther apart than in humid regions to ensure that salt-laden water cannot move upward to the root zone by capillary action. Since there is no real agreement on proper depth and spacing, the designer is forced to rely on local experience. Examples of drain depth and spacing in humid and arid climates, for slow rate systems, are shown in Table 5-29.

Control of the groundwater table is discussed in Appendix C, Section C.4. An equation for spacing and depth underdrains is presented. Additional discussion of the theoretical aspects of drain spacing is contained in references [60, 61, 62]. Procedures for planning and design of underdrainage systems are also described in <u>Drainage of Agricultural Land</u> by the U.S. Department of Agriculture, Soil Conservation Service [63].

TABLE 5-29

DEPTH AND SPACING OF UNDERDRAINS FOR SLOW RATE SYSTEMS
Feet

	Avg depth	Spacing
Arid climate		
Imperial Valley, California [62]	6-9	200-400
Delta, Utah [62]	...[a]	1 000-1 320
Humid climate		
Malheur Valley, Oregon [62]	8-9	660
Muskegon, Michigan, loamy to sandy soils [31]	5-8	500-1 000
Skaggs Water Management Model		
Sandy loam [60]	3.2	265
Sandy loam [61]	3.3	140
Clay loam [61]	3.3	40-65[b]

a. Referred to as deep drains.

b. Good surface drainage increases spacing.

1 ft = 0.305 m

Proper placement of underdrains to recover renovated water from rapid infiltration treatment is more critical than for slow rate treatment. Bouwer [52] has developed an equation to determine the distance underdrains should be placed away from the infiltration area. The height, H_c, of the water table below the outer edge of the infiltration area (see Figure 5-26) can be calculated:

$$H_c^2 = H_d^2 + IW\ (W + 2L)/K \qquad (5\text{-}9)$$

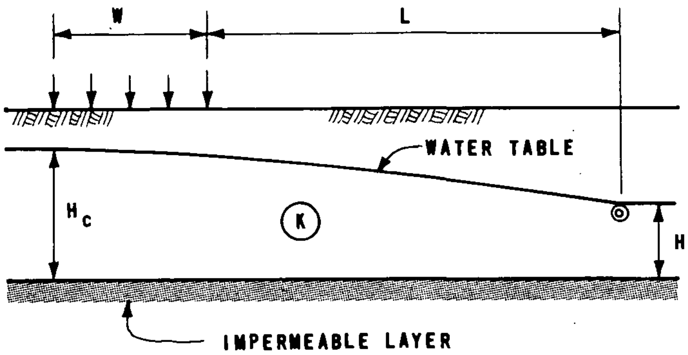
where H_d = drain height above impermeable layer, ft (m)
I = infiltration rate, in./h (cm/h)
W = width of infiltration basin, ft (m)
L = distance to underdrain, ft (m)
K = permeability of the soil, in./h (cm/h)

The location of the drain is selected and H_c is calculated with Equation 5-9. By adjusting variables (L, W, and I), a satisfactory value of H_c is obtained. An L-value less than the most desirable distance of underground travel may have to be accepted to obtain a workable system.

Plastic, concrete, and clay tile lines are used for underdrains. The choice usually depends on price and availability of materials. Where sulfates are present in the groundwater, it is necessary to use a sulfate-resistant cement pipe, if concrete is chosen, to prevent excess internal stress from crystal formation. Most tile drains are mechanically laid in a machine dug trench (see Figure 5-27) or by direct plowing. In organic soils and loam and clay-loam soils, a filter is not needed. The value of a filter is also dependent on the cost of cleaning a plugged tile line versus the cost of the filter material.

FIGURE 5-26

COLLECTION OF RENOVATED WATER BY DRAIN [52]



5.5.3 Pumped Withdrawal

Pumped withdrawal of percolated water is generally only considered for rapid infiltration systems. It can be the economical recovery method when the aquifer is deep enough (more than 15 ft or 4.5 m usually) and permeable enough to allow pumping. Evaluation of the permeability of an aquifer to properly locate recovery wells is based on the principles of groundwater flow presented in Appendix C.

Procedures for obtaining the necessary information on the permeability for rapid infiltration systems have been developed by Bouwer [64]. Two procedures, (1) an analog technique and (2) field permeability measurements, predict water table positions for a system of parallel, rectangular infiltration basins, with wells located midway between the basins as shown in Figure 5-28. The shape of the water table system can be calculated with dimensionless graphs developed with Bouwer's electrical analog technique [64] and summarized in reference [52]. The evaluation of the permeability components by the analog technique requires a knowledge of the infiltration rates and the response of water levels in the recovery (or observation) wells at different depths located between the basins.

FIGURE 5-27

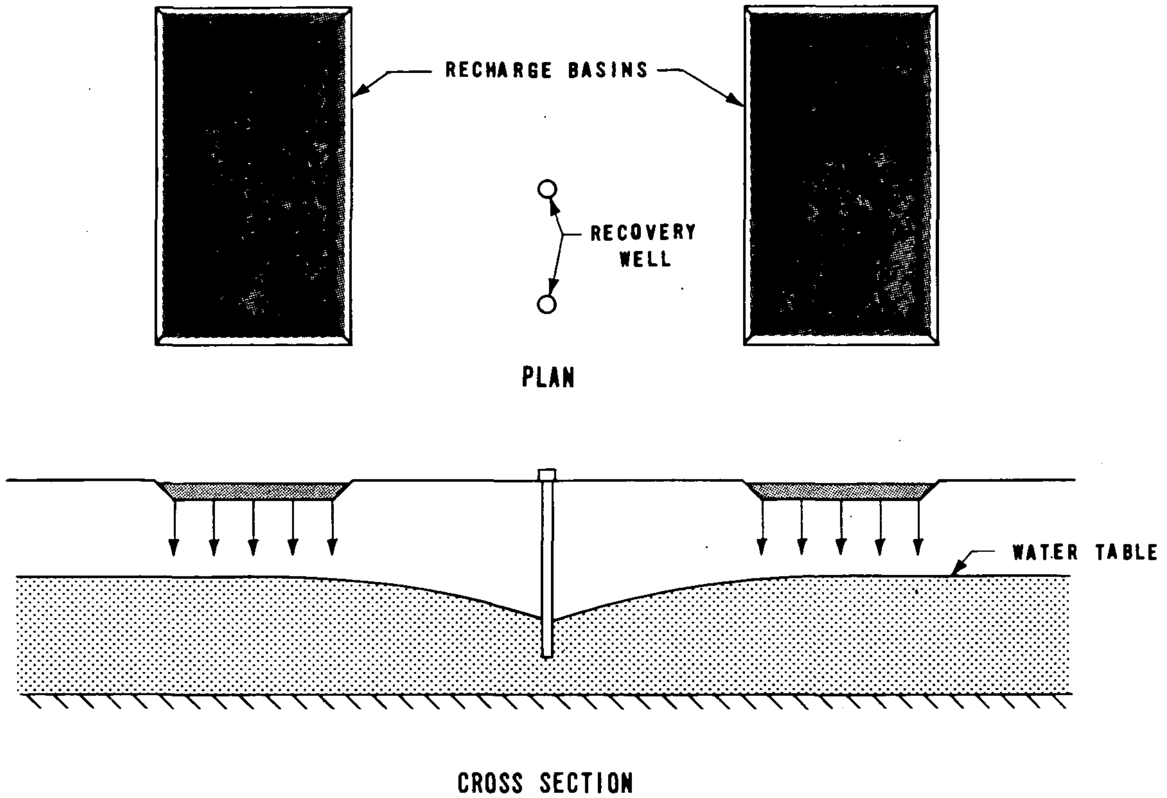
TRENCHING MACHINE FOR INSTALLATION
OF DRAIN TILE

5.5.4 Tailwater Return

It is standard design practice to include a tailwater return system for wastewater runoff from excess surface application in slow rate systems. Typically, tailwater systems consist of a small pond, a pump, and return pipeline. The system is usually sized for 25 to 50% of the applied surficial flow. Suggested guidelines, recommended by the ASAE, for determining runoff as a percentage of the application rate have been summarized by Hart [56] and are included here.

FIGURE 5-28

PLAN AND CROSS SECTION OF TWO PARALLEL RECHARGE BASINS WITH WELLS MIDWAY BETWEEN BASINS [64]



Total application time should be long enough to properly wet the lower end of a field. The time that applied wastewater is allowed to enter the tailwater runoff system before the supply source is cut off and the runoff volume depend on the intake rate of the soil. For slow rate land treatment, the practical guidelines shown in Table 5-30 provide the simplest procedure for estimating runoff factors.

TABLE 5-30

RECOMMENDED ASAE RUNOFF DESIGN FACTORS FOR SURFACE FLOOD DISTRIBUTION [56]

Permeability				
Class	Rate, in./h	Texture range	Maximum runoff duration, % of application time	Estimated runoff volume, % of application volume
Slow to moderate	0.06 to 0.6	Clay to silt	33	15
Moderate to moderately rapid	0.6 to 6.0	Clay loams to sandy loams	75	35

1 in./h = 2.54 cm/h

The rate of runoff increases with time and tends to reach a constant value as cutoff time is approached. A runoff duration of one-half the application time and a maximum runoff rate of two-thirds application rate results in a runoff volume of about 25% of application for slowly permeable soils [56]. Permeable soils, with intake rates greater than 0.8 in./h (2 cm/h), require rapid advance rates and shorter irrigation times if deep percolation is to be minimized. If deep percolation is not a problem, longer application periods can be used.

Design factors on sumps, pumps, and storage reservoirs for continuous pumping systems and cycling sump systems are beyond the scope of this manual but can be obtained from references [56, 65].

5.5.5 Overland Flow Runoff

Runoff will range from 40 to 80% of the applied liquid depending on: (1) soil infiltration capacity, (2) prior moisture condition of the soil, (3) slope, and (4) type of vegetation. Percent runoff will vary over the year depending on the rainfall and evaporation. A water balance should be performed to estimate the runoff volume.

At the Campbell Soup overland flow system in Paris, Texas, Thomas et al. determined that direct evaporation from sprinklers ranged from 2 to 8%; evapotranspiration ranged from 7 to 27% of the applied liquid (wastewater and rainfall); while runoff ranged from a midsummer low of 42% to a high of 71% in midwinter [66]. Similar studies at Ada, Oklahoma, indicated that overall recovery was about 50% of the applied wastewater, and ranged from 25% in summer to 80% in winter [17].

Runoff collection systems are commonly open, grass-lined channels at the toe of the overland flow slopes. They must be graded to prevent erosion (typically 0.3 to 1%) and have sufficient slope to prevent ponding in low spots. Channel slopes greater than 1% will begin to influence the distribution of the sheet flow on the overland flow slopes. Gravity pipe systems may be required when unstable soil conditions are encountered, or when flow velocities are prohibitively erosive. The collection system must be designed to accept a realistic amount of storm runoff--design storms of 2 to 10 years may not be unreasonable.

5.5.6 Stormwater Runoff Provisions

For slow rate systems, control of stormwater runoff to prevent erosion is necessary. Terracing of steep slopes is a well known agricultural practice to prevent excessive erosion. In general, the management techniques recommended in 208 planning for nonpoint discharges are applicable. Sediment control basins and other nonstructural control measures, such as contour plowing, no-till farming, grass border strips, and stream buffer zones can be used. As wastewater application will usually be stopped during storm runoff conditions, recirculation of storm runoff for further treatment is usually unnecessary.

For overland flow systems, even the "first flush" of a high intensity storm should meet water quality standards. Where the treated runoff is to be disinfected or collected for other uses, the quantity of stormwater will require that provisions for maximum treatment capacity be made. Stormwater in excess of this capacity should be allowed to overflow to a planned stormwater runoff system or to natural drainage. When more than 2 or 3 terraces discharge to the same collection main, provisions should be made to dampen the peak runoff from storms to minimize erosion and channel maintenance problems.

5.6 Vegetation

Vegetation in land treatment serves three major functions:

1. As a nutrient extractor, vegetation concentrates nitrogen and phosphorus above the ground and thus makes these nutrients available for removal through harvest.

2. Plants effectively reduce erosion by reducing surface runoff velocity. The extension of root growth maintains and increases soil permeability, and the leaf shelter protects the soil against the compacting effect of falling water. The overall effect of various ground covers on soil infiltration rates for one soil is shown in Figures 5-29 and 5-30.

3. For overland flow and wetlands, the vegetation, in addition to taking up nutrients, provides a matrix for the growth of microorganisms that decompose the organic matter in the wastewater.

5.6.1 Selection of Vegetation

For slow rate systems, the important considerations for agricultural crops are:

1. Rate of water uptake

2. Rate of nitrogen and phosphorus uptake

3. Tolerance to potentially harmful wastewater constituents

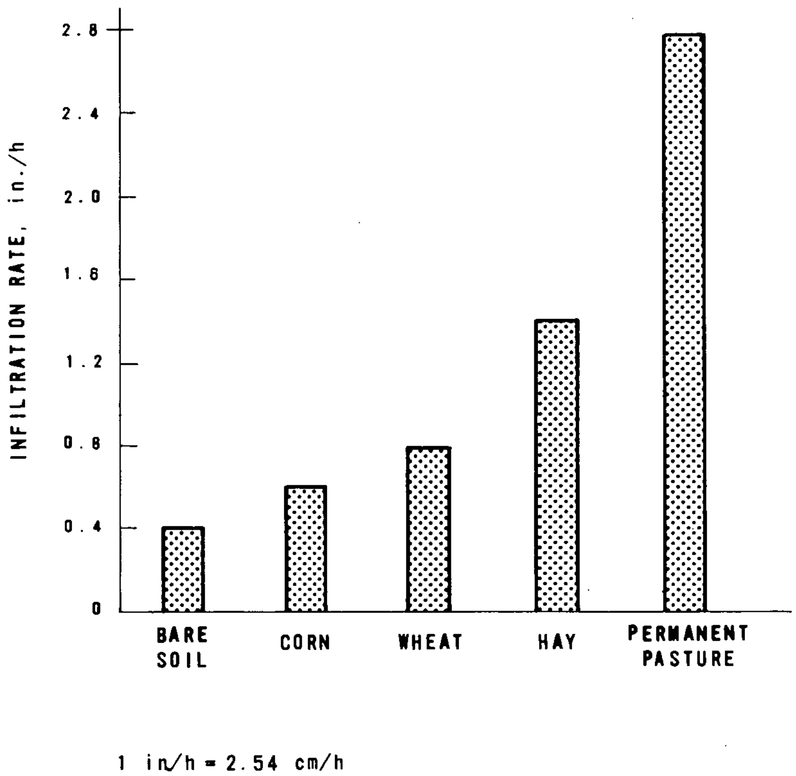
4. Ease of cultivation

5. Production of a marketable crop

6. Minimum net cost of production, after deducting the current market value of the crop

FIGURE 5-29

EFFECT OF SELECTED VEGETATION ON SOIL INFILTRATION RATES [67]

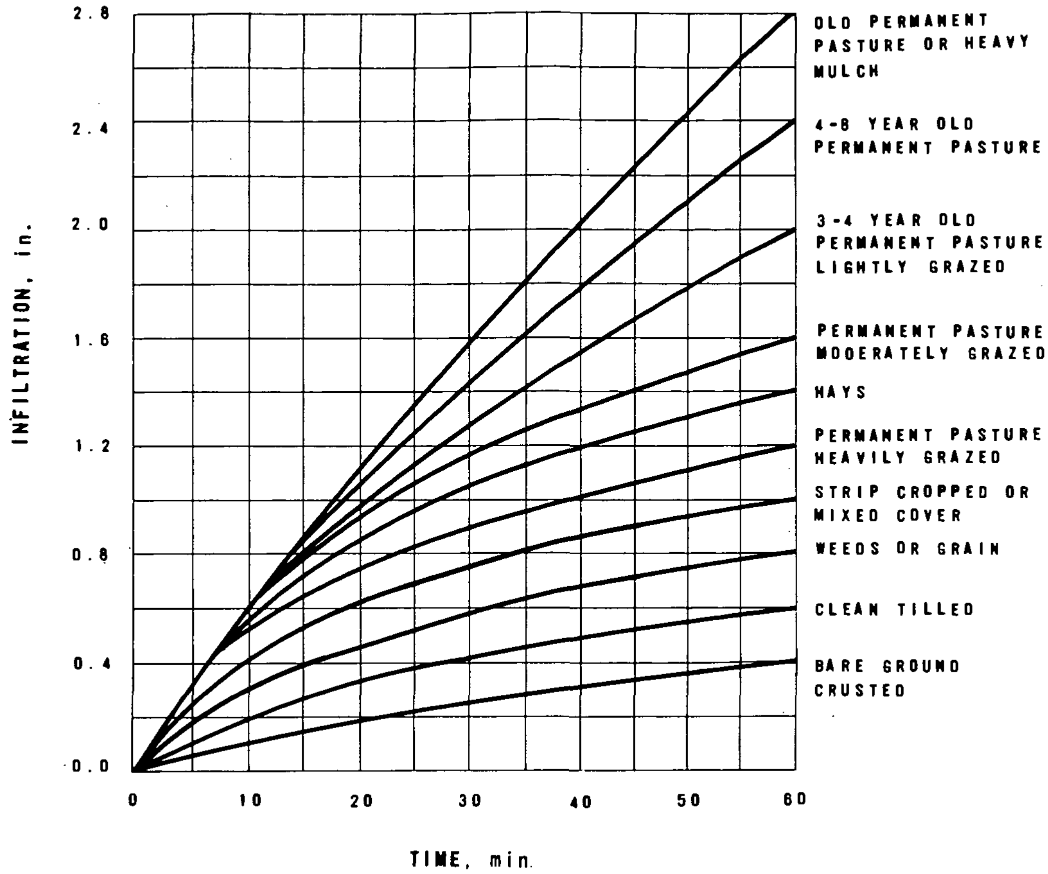


For rapid infiltration systems, the primary requirement is for a water-tolerant species that will help to maintain high infiltration rates. For overland flow systems, the need is for a vegetative cover that is well rooted in impermeable soils, is water tolerant (withstands flooding), and has a high rate of nitrogen uptake.

In general, the forage and fodder crops are preferred because they: (1) treat large amounts of wastewater, (2) are tolerant of variations in wastewater quality, and (3) require less maintenance and skill to grow. However, they have a lower market value. Successful forage crops used to date include: Reed canary grass, fescue, perennial rye, orchard grass, and Bermuda grass.

FIGURE 5-30

INFILTRATION RATES FOR VARIOUS CROPS [68]



1 in. = 2.54 cm

5.6.1.1 Hydraulic Considerations

Peak consumptive water use and rooting depth for various crops and regional areas are presented in Table 5-31 as an aid in system design. The tolerance of individual species to flooding is based on the rooting depth and the duration of flooding. Rooting depths for various crops are also listed in Table 5-31. The soil should drain and become unsaturated to these depths during the irrigation resting cycle to obtain optimal growth. Some saturation of the root zone by groundwater may be tolerated, but the usual result is decreased plant performance.

In general, grain crops such as wheat, oats, and barley will suffer high yield losses if subjected to soil saturation. Vegetable and row crops are slightly more tolerant, but they are still susceptible to damage.

TABLE 5-31

PEAK CONSUMPTIVE WATER USE AND ROOTING DEPTH [69]

Crop	Washington, Columbia Basin		California, San Joaquin Valley		Texas, southern high plains		Arkansas, Mississippi bottoms		Nebraska, eastern part		Colorado, western part	
	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d	Depth, in.	rate, in./d	Depth, in.	rate, in./d	Depth, in.	rate, in./d
Corn	42	0.27	60	0.26	72	0.30	30	0.23	72	0.28	48	0.23
Alfalfa	60	0.25	72	0.25	72	0.30	42	0.24	96	0.27	72	0.23
Pasture	24	0.29	24	0.32	42[b] 72[c]	0.25 0.30	36[b] 36[c]	0.13 0.22	48	0.29	36	0.23
Grain	48	0.21	48	0.17	72	0.15	24	0.15	48	0.26	36	0.22
Sugar beets	36	0.26	72	0.22	..	....	..	....	48	0.26	48	0.20
Cotton	..	....	72	0.22	72	0.25	36	0.18	..	....	..	....
Potatoes	24	0.29	48	0.24	..	....	..	....	36	0.26	36	0.22
Deciduous orchards	..	....	96	0.21	..	....	..	....	..	....	72	0.18
Citrus orchards	..	....	72	0.19	..	....	..	....	..	....	..	....
Grapes	..	....	72	0.18	..	....	..	....	..	....	..	....
Annual legumes	..	....	..	....	48	0.18	18	0.28	..	....	..	....
Soybeans	..	....	..	....	..	....	36	0.19	60	0.27	..	....
Shallow-rooted truck crops	..	....	..	....	..	....	..	....	..	....	..	....
Medium-rooted truck crops	..	....	..	....	..	....	18[d] 18[e]	0.20 0.12	..	....	..	....
Deep-rooted truck crops	..	....	..	....	..	....	..	....	..	....	..	....
Tomatoes	..	....	..	....	..	....	..	....	..	....	36	0.22
Tobacco	..	....	..	....	..	....	..	....	..	....	..	....
Rice	..	....	..	....	..	....	24	0.17	..	....	..	....

TABLE 5-31
(Concluded)

Crop	State of Wisconsin		State of Indiana		Piedmont Plateau[f]		Virginia, coastal plain		State of New York	
	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d	Depth, in.	Use rate, in./d
Corn	24	0.30	24	0.30	24	0.22	24	0.18	24	0.20
Alfalfa	36	0.30	36	0.30	36	0.25	36	0.22	30	0.20
Pasture	24	0.20	30	0.30	24	0.25	20	0.22	..	....
Grain	18	0.25	..	....	24	0.16	..	....	..	....
Sugar beets	18	0.25	..	....	..	....	..	....	..	....
Cotton	..	....	..	....	24	0.21	..	....	..	....
Potatoes	18	0.20	12	0.25	24	0.18	18	0.18	18	0.18
Deciduous orchards	36	0.30	..	....	36	0.25	36	0.22	36	0.20
Citrus orchards	..	....	..	....	..	....	..	....	..	....
Grapes	..	....	24	0.25	30	0.20	..	....	..	....
Annual legumes	..	....	..	....	..	....	..	....	..	....
Soybeans	18	0.25	24	0.30	24	0.18	..	....	..	....
Shallow-rooted truck crops	12	0.20	9	0.20	12	0.14	..	....	12	0.18
Medium-rooted truck crops	18	0.20	12	0.20	18	0.14	18	0.16	18	0.18
Deep-rooted truck crops	24	0.20	18	0.20	24	0.18	..	....	24	0.18
Tomatoes	18	0.20	18	0.20	24	0.21	24	0.18	24	0.18
Tobacco	..	....	24	0.25	18	0.18	18	0.17	..	....
Rice	..	....	..	....	..	....	..	....	..	....

a. Average daily water use rate during the 6 to 10 days of the highest consumptive use of the season.

b. Cool season pasture.

c. Warm season pasture.

d. Summer.

e. Fall.

f. Parts of Georgia, Alabama, North Carolina, and South Carolina.

1 in. = 2.54 cm

Corn and potatoes will tolerate some flooding, possibly up to a few days, without suffering damage; clover, timothy, and rye are also somewhat resistant. Grasses (such as coastal Bermuda, meadow, fescue, brome, orchard, or Reed canary) are the most tolerant species and can sustain several weeks of flooding without injury. Reed canary grass, a tall cool-season perennial with a rhizomatous root system, will grow in a very wet, marshy area, and reportedly has withstood flooding for as long as 49 days without permanent injury [70].

5.6.1.2 Nutrient Uptake

The major nutrients essential to plant growth are nitrogen, phosphorus, potassium, calcium, magnesium, and sulfur. Of these, the prominent constituents in wastewater are nitrogen, phosphorus, and potassium. Typical uptake rates of these elements for various crops are listed in Table 5-32. Variations noted in the amount of nutrient uptake from the soil can arise from changes in either (1) the amount and form of the nutrient present, or (2) the net yield of the crop.

TABLE 5-32

NUTRIENT UPTAKE RATES FOR SELECTED CROPS
[3, 4, 5, 6, 70, 71]

	Uptake, lb/acre·yr		
	Nitrogen	Phosphorus	Potassium
Forage crops			
Alfalfa[a]	200-480	20-30	155-200
Bromegrass	116-200	35-50	220
Coastal Bermuda grass	350-600	30-40	200
Kentucky bluegrass	180-240	40	180
Quackgrass	210-250	27-41	245
Reed canary grass	300-400	36-40	280
Ryegrass	180-250	55-75	240-290
Sweet clover[a]	158	16	90
Tall fescue	135-290	26	267
Field crops			
Barley	63	15	20
Corn	155-172	17-25	96
Cotton	66-100	12	34
Milomaize	81	14	64
Potatoes	205	20	220-288
Soybeans[a]	94-128	11-18	29-48
Wheat	50-81	15	18-42

a. Legumes will also take nitrogen from the atmosphere and will not withstand wet conditions.

1 lb/acre·yr = 1.12 kg/ha·yr

Nutrient content of a plant depends, in part, on the amounts of nutrients available to the plant. The minimum cellular amounts required are about 2% nitrogen, 0.2% phosphorus, and 1+% potassium, but when sufficient quantities are available, these amounts can easily double [71, 72]. For forage crops in general, the percent composition for nitrogen can range from 1.2 to 2.8% and averages around 1.8% (dry weight of the plant); but with wastewater irrigation it can range from 3.0 to 4.5% [72].

The total uptake of nutrients from applied wastewater increases as crop yield increases (see Figure A-3, Appendix A). Crop yield increases ranging up to twofold to fourfold have been achieved when wastewater effluent irrigation is used instead of ordinary irrigation water [73]. Although nutrient uptake continues to increase with yield, the relationship is not linear.

A factor that affects both percent nitrogen composition and yield of forage crops is stage of growth. In general, grasses contain the highest percentage of nitrogen during the green, fast growth stage. The nitrogen uptake decreases with maturity. These effects are demonstrated in Figure 5-31. For corn and grasses, nitrogen uptake is very low during early growth (the first 30 to 40 days) and thereafter climbs sharply. For corn, this rise is maintained until harvest. For grasses, nitrogen uptake reaches a peak around the 50th day and thereafter declines. This suggests that harvesting these grasses every 8 to 9 weeks (for a total of two to three harvests per season) will result in maximum nitrogen uptake.

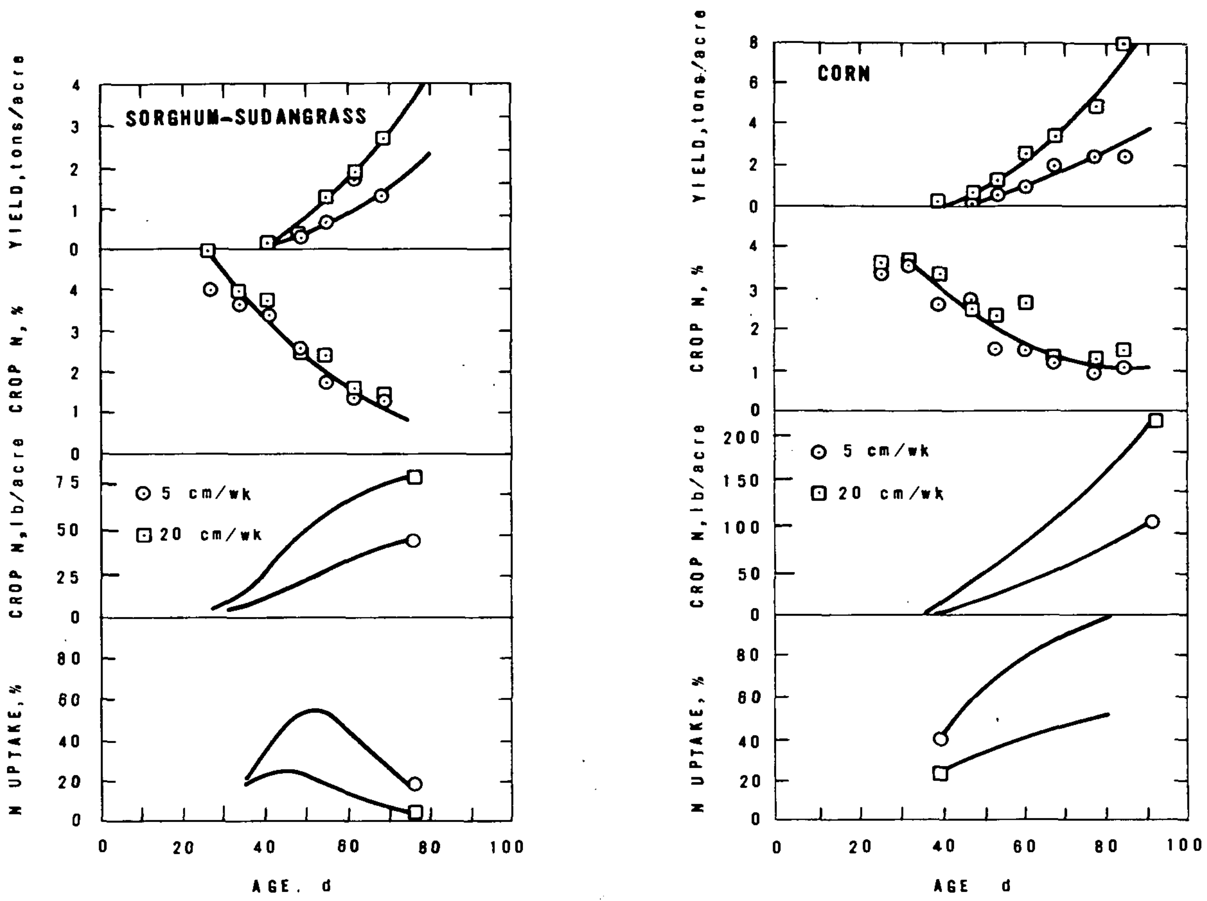
The amounts of phosphorus in applied wastewaters are usually much higher than plant requirements. Fortunately, many soils have a high sorption capacity for phosphorus and very little of the excess is passed on to the groundwater. Instead, it is held in the soil and serves to enrich the soil [74].

Potassium is used in large amounts by many crops, but typical wastewater is relatively deficient in this element. In some cases fertilizer potassium may be needed to provide for optimal plant growth, depending on the soil and crop grown.

The micronutrients important to plant growth (in descending order) are: iron, manganese, zinc, boron, copper, molybdenum, and occasionally, sodium, silicon, chloride, and cobalt. Most wastewaters contain an ample supply of these elements, and in some cases, phytotoxicity may be a consideration.

FIGURE 5-31

CROP GROWTH AND NITROGEN UPTAKE VERSUS DAYS FROM PLANTING FOR FORAGE CROPS UNDER EFFLUENT IRRIGATION [75]



1 in./wk = 2.54 cm/wk
1 lb/acre = 1.12 kg/ha
1 ton/acre = 2.24 Mg/ha

5.6.1.3 Sensitivity to Wastewater Constituents

Plant growth can be adversely affected by excess salts (generally chloride and sodium), excess acidity, or excess concentrations of any of a large number of microelements, including the micronutrients.

Tolerances of selected crops to salinity, boron, and acidity are presented in Tables 5-33, 5-34, and 5-35, respectively. In general, forage crops are the most tolerant, field crops are less tolerant, and vegetable and row crops are least tolerant. There are many exceptions to this rule, however, and wide differences can be found even between two varieties of the same crop. Data on crops not listed in these tables are available in references [76-78], and the local Agricultural Extension Service can give details on crops suitable for a proposed site.

TABLE 5-33

ELECTRICAL CONDUCTIVITY VALUES RESULTING IN REDUCTIONS IN CROP YIELD [77]
mmhos/cm

	EC_e values (saturated paste extract) for a reduction in crop yield of		
	0%	25%	100%
Forage crops			
Alfalfa	2.0	5.4	15.5
Bermuda grass	6.9	10.8	22.5
Clover	1.5	3.6	10
Corn (forage)	1.8	5.2	15.5
Orchard grass	1.5	5.5	17.5
Perennial rye grass	5.6	8.9	19
Tall fescue	3.9	8.6	23
Vetch	3.0	5.3	12
Tall wheat grass	7.5	13.3	31.5
Field crops			
Barley	8.0[a]	13	28
Corn	1.7	3.8	10
Cotton	7.7	13	27
Potato	1.7	3.8	10
Soybeans	5.0	6.2	10
Sugarbeets	7.0	11.0	24
Wheat	6.0	9.5	20

a. Barley and wheat are less tolerant during germination and seedling stage. EC_e should not exceed 4 or 5 mmhos/cm.

When evaporation is high, problems can arise from the use of sprinklers. When water is applied to vegetative surfaces, excess quantities of sodium and chloride can be absorbed through the wet leaves and cause leaf burn. Nighttime applications can alleviate foliar absorption and leaf burn due to chlorides or bicarbonates.

TABLE 5-34

CROP BORON TOLERANCE [77]
mg/L

Tolerant, 1-3 mg/L boron	Semitolerant 0.67-2 mg/L boron	Sensitive, <1 mg/L boron
Alfalfa	Barley	Citrus
Cotton	Corn	American elm
Sugarbeet	Kentucky bluegrass	Berries
Sweetclover	Potato	
	Tomato	
	Wheat	

TABLE 5-35

CROP ACIDITY TOLERANCE [78]

Will tolerate mild acidity, pH 5.8 to 6.5	Will tolerate slight acidity, pH 6.2 to 7.0	Very sensitive to acidity, pH 6.8 to 7.5
Cotton	Corn	Alfalfa
Buckwheat	Beans	Barley
Bentgrass	Kentucky bluegrass	Carrot
Millet	Clovers: alsike crimson, red, white	Sweet clover
Potato	Kale	Sugarbeet
Poverty grass	Tomato	
Oats	Soybean	
Rye	Wheat	
Sudan grass		
Vetch		

There are two considerations in trace element accumulation in the soil: (1) phytotoxicity, and (2) translocation into the food chain. Copper, zinc, and nickel are the prime examples of elements that can be toxic to some plants at relatively high levels. At present there is little definitive evidence that these elements have accumulated to phytotoxic levels in any land treatment system [79]. The principal element of concern for potential translocation into the food chain is cadmium. This is discussed in detail in Appendix A.

When selecting resistant species to prevent toxicity, a distinction should be made between accumulators and excluders. Accumulators will tolerate high levels of an element while transferring large quantities of it into the harvestable portions of the plant, making it available for removal. Excluders will also tolerate high levels, but prohibit passage of the toxifying element into the fruit, root, or leaf tissue that is to be consumed. For example, corn may take up cadmium but it is mostly excluded from the grain. In general, grain crops are superior to vegetables in excluding heavy metals [79].

5.6.1.4 Selection of Overland Flow Vegetation

Perennial grasses with long growing seasons, high moisture tolerance (hydrophytic), and extensive root formation are best suited to the process. The grass should form a sod and not grow in bunches. While common Bermuda, red top, fescue, and rye grass all form sod, none of these is always suitable for all weather conditions. Bermuda goes dormant in winter while red top, fescue, and rye grass are cool season grasses. Reed canary grass is the most versatile but it is a bunch grass. It should therefore be planted with a mixture of other grasses such as red top, fescue, and rye grass.

Comparative field studies at Paris, Texas, indicated that Reed canary grass was the superior grass at that location. It demonstrated a very high nutrient uptake capacity and yielded a high quality hay upon harvest [54]. Hauling the crop away during harvest provides permanent removal of the nutrients taken up during plant growth. The harvested grass is suitable for feeding to cattle.

5.6.1.5 Other Vegetation

Sod, landscape vegetation, trees, and wetlands vegetation are discussed separately because of their unique features. Much of the previous discussion will apply.

5.6.1.5.1 Sod

Sod farming is the controlled growth of turf grasses for transplanting to lawns, golf courses, and parks. Usually, public access to the growing site is restricted so that bacteriological quality of the wastewater is not a major concern. Because the sod is removed periodically, the nitrogen loadings can exceed crop uptake as well as soil nitrogen accumulation.

5.6.1.5.2 Landscape Irrigation

Application of wastewater on landscape areas such as highway median and border strips, airport strips, golf courses, parks and recreational areas, and nature-wildlife areas has several advantages. The areas irrigated are already publicly owned, saving acquisition cost, and problems associated with crops for consumption are avoided. Additionally, the maintenance of landscape projects generally requires less water than other vegetation (since watering in these cases is based on vegetative maintenance rather than production); hence, the wastewater can be spread over a greater area.

Although sufficient areas to accept available effluent are usually available, wastewater distribution, especially for roadside rights-of-way, can be a problem. For roadside application, sprinkler trucks are commonly used; for application to golf courses, playgrounds, and nature areas, fixed sprinklers are most commonly used.

5.6.1.5.3 Woodlands Irrigation

Approximate average water consumption rates for native stands of different tree species are given in Table 5-36.

TABLE 5-36

EVAPOTRANSPIRATION OF WOODLAND AND FOREST CROPS [80]

	No. of studies	Evapotranspiration, in./yr	
		Average	Range
Pines	32	15	5-34
Mixed coniferous and deciduous	6	25	18-34
Deciduous	58	17	8.5-34
Mixed hardwoods	2	31	27-35

1 in./yr = 2.54 cm/yr

Recommended irrigation rates for maintaining desired forest crops, determined from studies using wastewater irrigation, are shown in Table 5-37. These rates, which generally agree with those in Table 5-36, suggest that where water consumption is a primary consideration, pines are at a disadvantage.

TABLE 5-37

RECOMMENDED IRRIGATION RATES OF FOREST CROPS

Species	Maximum recommended irrigation rate, in./wk	Reason for limit
Pines [75, 76, 77]	1	Satisfactory tree growth rate and nitrogen removal
Hardwoods		
[81]	1	Satisfactory nitrogen removal
[82, 83]	2-4	Satisfactory tree growth rate
Douglas firs cottonwoods [84]	2	Trees grow well and consume all available water at this rate
Conifers [85]	1 (winter) 4 (summer) (104 in./yr)	Satisfactory tree growth rate

1 in./wk = 2.54 cm/wk

Pines and other conifers, however, have an advantage in that they maintain their water uptake rates year-round, if freezing temperatures do not make the water unavailable. Deciduous species exhibit cyclical water needs with a very active growing season during the summer, followed by a dormant phase in the winter. Water consumption then drops to a level of one-half to one-fourth the summer rates, generally less than 1 in./wk (2.54 cm/wk). A major objective in silviculture is to maintain an adequate unsaturated soil zone for the proper development of the tree root system.

Wood quality associated with effluent-irrigated stands, as studied by Murphey et al. [86], indicates that the pulpwood characteristics of pine and oak are improved via an increase in fibre length and cell wall thickness. Structural strength, however, appears to suffer a decrease, rendering the wood less suitable for construction purposes.

For harvesting purposes, cottonwood seems to show the greatest growth response to effluent irrigation [82, 83], and tree harvests every 6 to 10 years may be possible. Eucalyptus is also a fast grower, but is limited to areas without hard frosts. Studies at Stanford Research Institute have suggested the creation of eucalyptus biomass plantations to be harvested and burned for the production of electricity [87].

A major limitation to the use of woodlands and forests is the relatively low rates of nutrient uptake. Typical rates of nitrogen uptake for different forest crops were listed in Table 5-2. These rates will usually be maintained through the growing phase (20 to 40 years) and will taper off as maturity is reached. Conifers as Christmas trees should be abandoned. The extra water and nutrients cause the trees to grow upward, rather than outward, resulting in spindly, unattractive trees.

5.6.1.5.4 Wetlands

Experience has shown that duckweed (Lemna minor) and various species of bulrush (Scirpus acustus, Scirpus lacustris, Scirpus validus) are the most desirable species, based on treatment capabilities, growth rates, and harvest response for marshes [22, 23, 88]. Cattails seem to have trouble competing with the bulrushes and duckweed under harvest conditions [22].

Marsh studies by the National Aeronautics and Space Administration concluded that water hyacinths (Eichornia crassipes), and to a lesser extent alligator weed (Alternanthera philovernides) are effective in removal of both organics and some metals [89, 90].

Experiments have been conducted in Florida with cypress domes as nutrient sinks, and they appear to be quite efficient [27]. Artificial peat beds also appear to be effective, removing 85% of the nitrogen, 99.3% of the phosphorus, and 99.99% of the coliform bacteria when grown with a quackgrass or bluegrass cover [27, 91].

5.6.1.6 Regulatory Constraints

Many states regulate the type of wastewater that can be used to irrigate some crops. In addition, several states require that a suitable crop be planted before land application begins [92]. In some cases the type of crop proposed affects the slope of the site that is acceptable.

5.6.1.7 Crop Utilization

Of crops historically grown with wastewater, under present cost conditions, corn appears to provide the greatest (net) profit [93, 94]. At the Muskegon Project, the 1976 revenue from their corn harvest was approximately $1 000 000 (see Section 7.6). There are no restrictions placed on the sale of this corn.

Among the trees, maples (and certain other hardwoods), cotton woods, and pines grown under wastewater irrigation are suitable for sale as pulp, but not for structural wood [86]. Cotton wood and eucalyptus are suitable for sale as fuel [87].

5.6.2 Site Preparation and Management

It is critical to maintain the soil-vegetation system in a healthy, productive, and renovative state. A successful agricultural system requires knowledge of farming operations, which are described briefly in this section. Assistance in design and planning can be provided by local farm advisers and land grant college extension specialists.

5.6.2.1 Field Preparation

Procedures for preparing fields for slow rate systems may include clearing the fields of vegetative growth (bulldozing of heavy vegetation into piles followed by burning, or heavy stubble disking on lighter vegetation); planing and grading, if required, and ripping, disking, and tilling of the soil to loosen and aerate it. Undeveloped soils may require chemical soil amendments, including gypsum to reclaim sodic soils and increase permeability, and lime to reduce acidity and metals toxicity. Determination of amendment needs is discussed in Section 5.7.3. The effects of lime on element availability are indicated in Table 5-38. Fertilizers may also be added for nutrient-deficient soils, although nutrient-rich wastewaters often make this unnecessary.

TABLE 5-38

EFFECT OF LIME ON ELEMENT AVAILABILITY IN SOIL [76, 78]

Elements for which liming	
Reduces availability	Increases availability
Aluminum	Calcium
Barium	Magnesium
Beryllium	Molybdenum
Boron[a]	Nitrogen[a]
Cadmium	Phosphorus
Cobalt	Potassium
Copper[a]	Sulfur[a]
Fluoride	
Iron	
Lithium	
Manganese	
Nickel	
Zinc	

a. Minor effect on availability.

5.6.2.2 Maintenance of Infiltration for Slow Rate Systems

Soil-water infiltration rates can be reduced by surface sealing and clogging. The sealing is the result of: (1) compaction of the surface from machine working, (2) compaction from raindrops and sprinkler drops, (3) a clay crust caused by water flowing over the surface (fine particles are fitted around larger particles to form a relatively impervious seal), or (4) clogging due to suspended particles, buildup of organic matter, or trapped gases. This surface layer can be broken up by plowing, cultivation, or any other stirring of the soil that will result in increased water intake. Tillage beyond the point of breaking up an impermeable layer is generally harmful in that it results in further soil compaction. The effect of surface sealing on intake can be greatly reduced, and possibly eliminated, by cultivating grass or other close-growing vegetation. Maintenance of soil organic matter through the use of high residual crops, such as barley, and plowing under of stubble is another step that helps maintain soil permeability.

5.6.2.3 Salinity Control

If the soil is saline (EC >4 mmhos/cm) for most crops, control measures must be taken. The average salt concentration of the soil solution of the rooting depth is usually three times the concentration of the salts in the applied water (in arid climates) and is believed to be representative of the salinity to which the crop responds [77]. If excessive salts build up, the method of control is leaching by adding enough irrigation water so that water in excess of crop needs percolates below the root zone, lowering the overall salinity. The most important zone for leaching is the upper quarter of the root zone where the primary (40%) water use by the plant occurs. As a rule-of-thumb, about a 12 in. (30 cm) depth of water leached through a 12 in. (30 cm) depth of soil should remove about 80% of the soluble salts.

5.6.2.4 Crop Management

5.6.2.4.1 Planting

Local extension services or similar experts should be consulted regarding planting technique and schedules.

5.6.2.4.2 Harvesting

Harvesting for grass crops and alfalfa involves regular cuttings, and a decision regarding the trade-off between yield and quality must be made. Crop yield will usually increase up to and beyond the flowering stage, but quality (amount of stems versus leaves and the amount of digestible material) is highest in the younger growth stages and falls off very rapidly once the flowering stage is reached. Advice can be obtained from local extension services.

5.6.2.4.3 Double Cropping

Double cropping can extend the operating period for slow rate systems, increase the economic return for the system, and increase the nitrogen uptake capacity.

A growing practice in the East and Midwest is to provide a continuous vegetative cover with grass and corn. This "no-till" corn management consists of planting grass in the fall and then applying a herbicide in the spring before planting the corn. When the corn completes its growth cycle, grass is reseeded. Thus, cultivation is avoided, water rates are maximized, and nutrient uptake is enhanced.

5.6.2.4.4 Grazing

Grazing of pasture by beef cattle or sheep can provide an economic return for slow rate systems (see Pleasanton, California, and San Angelo, Texas, in Chapter 7). This approach has also been successsfuly pursued at the land treatment farm in Melbourne, Australia, for the past 65 years [95]. Grazing cattle and sheep keep the vegetative cover short for maximum wastewater renovation efficiency. No health hazard has been associated with the sale of the animals for human consumption.

Grazing animals do return nutrients to the ground in their waste products. The chemical state (organic and ammonia nitrogen) and rate of release of the nitrogen reduces the threat of nitrate pollution of the groundwater. Much of the ammonia-nitrogen volatilizes. The organic nitrogen is held in the soil and is slowly mineralized. As a result, only a portion of the nitrogen is slowly recycled.

One precaution that must be taken is not allowing the cattle and sheep to graze on wet fields. This would compress the ground and reduce the permeability of the soil. As described in Chapter 7, Pleasanton, California, and San Angelo, Texas, solve the problem by using a series of fields in rotation. Wastewater irrigation proceeds on a field as soon as the cattle are moved off. In this manner, by the time the cattle are moved back onto a field to graze, it has had several weeks to dry out and firm up.

Another concern is the physical contact between the udders of milking animals (cows, goats) and pastures irrigated with wastewater. This could represent a direct vector to human food supplies and should be avoided.

5.7 System Monitoring

Monitoring of land treatment systems involves the observation of significant changes resulting from the application of wastewater. The monitoring data are used to confirm environmental predictions and to determine if any corrective action is necessary to protect the environment or maintain the renovative capacity of the system. The components of the environment that need to be observed include wastewater, groundwater, and soils upon which wastewater is applied and, in some cases, vegetation growing in soils that are receiving wastewater.

5.7.1 Water Quality

Monitoring of water quality for land application systems is generally more involved than for conventional treatment systems because nonpoint discharges of system effluent into the environment are involved. Monitoring of water quality at several stages of a land treatment process may be needed for process control. These stages may be: (1) applied wastewater, (2) renovated water, and (3) receiving waters--surface water or groundwater.

5.7.1.1 Applied Wastewater

The water quality parameters and the frequency of analyses will vary from site to site depending on the regulatory agencies involved and the nature of the applied wastewater. The measured parameters may include (1) those that may adversely affect receiving water quality either as a drinking water supply or an irrigation water supply, (2) those required by regulatory agencies, and (3) those necessary for system control. An example of a suggested water quality monitoring program for a large scale slow rate system is presented in Table 5-39.

5.7.1.2 Renovated Water

Renovated water may be recovered as runoff in an overland flow system, or as drainage from underdrains or groundwater from recovery wells in slow rate and rapid infiltration systems. Point discharge to surface waters must satisfy the NPDES permit.

TABLE 5-39

EXAMPLE MONITORING PROGRAM FOR A LARGE SLOW RATE SYSTEM

	Frequency of analysis					
				Groundwater		
Parameter	Applied wastewater	Soil	Plants	Onsite wells	Perimeter wells	Background wells
Flow	C	..	..	..	..	..
BOD or TOC	W	..	..	Q	Q	Q
COD	W	..	..	Q	Q	Q
Suspended solids	W	..	..	..	..	..
Nitrogen, total	W	2A	A	Q	Q	Q
Nitrogen, nitrate	..	..	..	Q	Q	Q
Phosphorus, total	M	2A	A	Q	Q	Q
Coliforms, total	W	..	..	Q	Q	Q
pH	D	Q	..	Q	Q	Q
Total dissolved solids	M	..	..	Q	Q	Q
Alkalinity	M	..	..	0	Q	Q
SAR	M	Q	..	Q	Q	Q
Static water level	..	..	..	M	M	M

Note: C = Continuously
D = Daily
Q = Quarterly
W = Weekly
2A = Two samples per year
A = Annually
M = Monthly

a. Wastewater applied and groundwater should be tested initially and periodically thereafter, as appropriate, for heavy metals, trace organics, or other constituents of environmental concern.

5.7.1.3 Groundwaters

In groundwaters, travel time of constituents is slow and mixing is not significant compared with surface waters. Surface inputs near a sampling well will move vertically and arrive at the well much sooner than inputs several hundred feet away from the well. Thus, the groundwater sample represents contributions from all parts of the surface area with each contribution arriving at the well at a different time. A sample may reflect surface inputs from several years before sampling and have no association with the land application system. Consequently, it is imperative to obtain adequate background quality data and to locate sampling wells so that response times are minimized.

If possible, existing background data should be obtained from wells in the same aquifer both beyond and within the anticipated area of influence of the land application system. Wells with the longest history of data are preferable. Monitoring of background wells should continue after the system is in operation to provide a base for comparison.

In addition to background sampling, samples should be taken from groundwater at perimeter points in each direction of groundwater movement from the site. In locating the sampling wells, consideration must be given to the position of the groundwater flow lines resulting from the application [96, 97]. Perimeter wells should be located sufficiently deep to intersect flow lines emanated from below the application area but not so deep as to prolong response times.

A schematic showing correct and incorrect groundwater sampling locations is given in Figure 5-32; monitoring points for a hypothetical application site are also shown. If samples are taken at A and B, the groundwater flow lines from the application area indicate that treated effluent would reach these points. It may require several years for treated effluent to reach point C because the flow lines are a long distance from the application surface. If samples were taken from point D, mixing with surface water could make results invalid for groundwater characterization.

A groundwater flow model that predicts groundwater movement in the area of influence of the site will be helpful in locating sampling wells. Guidelines for sampling well construction and sampling procedures are given by Blakeslee [98].

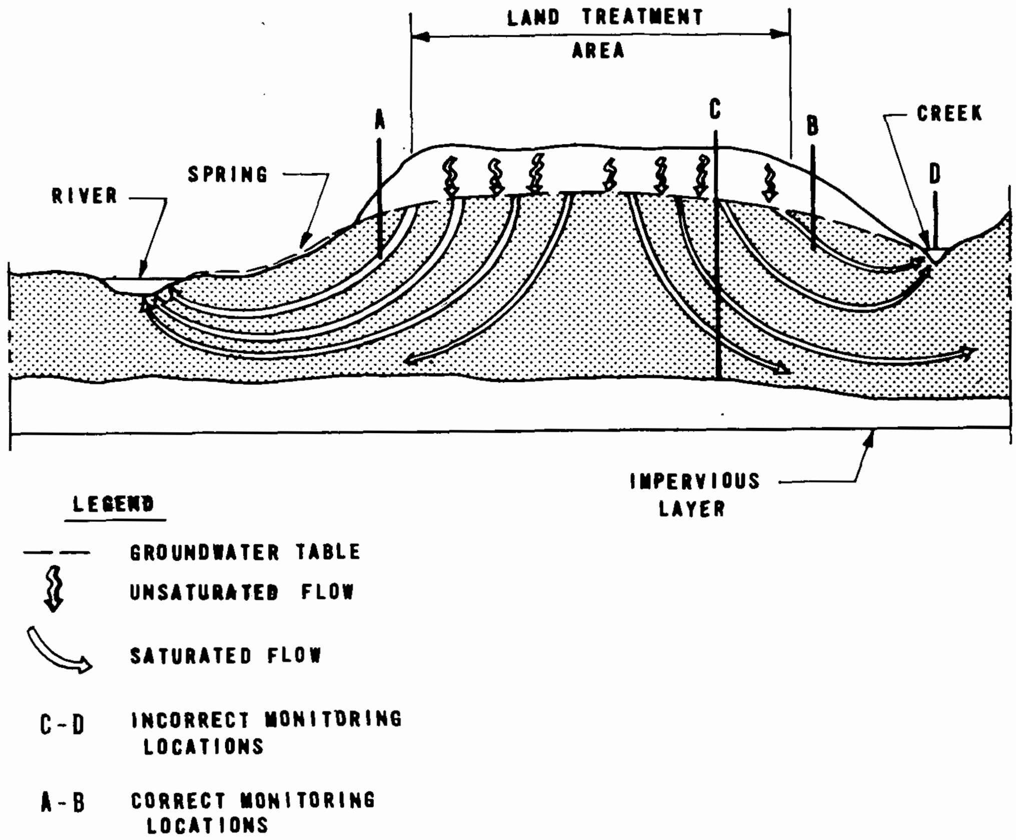
In addition to quality, the depth to groundwater should be measured at the sampling wells to determine if the hydraulic response of the aquifer is consistent with what was anticipated. For slow rate systems, a rise in water table levels to the root zone would necessitate corrective action such as reduced hydraulic loading or adding underdrainage. The appearance of seeps or perched groundwater tables might also indicate the need for corrective action.

5.7.2 Soils Management

In some cases, application of wastewater to the land will result in changes in soil properties. Results of soil sampling and testing will serve as the basis for deciding whether or not soil properties should be adjusted by the application of chemical amendments. Soil properties that are important to management include: (1) ph, (2) exchangeable sodium percentage, (3) salinity, (4) nutrient status, and (5) metals.

FIGURE 5-32

SCHEMATIC OF GROUNDWATER FLOW LINES AND
ALTERNATIVE MONITORING WELL LOCATIONS [95]



5.7.2.1 pH

Soil pH below 5.5 or above 8.5 generally is harmful to most plants (see Table 5-35). Below pH 6.5 the capacity of soils to retain metals is reduced significantly, the soil above pH 8.5 generally indicates a high sodium content and possible permeability problems. If wastewaters contain high concentrations of sodium, the soil pH may rise in the long term. A pH adjustment program should be based on the recommendations of a professional agricultural consultant or county or state farm advisor.

5.7.2.2 Exchangeable Sodium Percentage

When the percentage of sodium on the soil exchange complex (ESP) exceeds 10 to 15%, problems with reduced soil permeability can occur. Sodic soil conditions may be corrected by adding soluble calcium to the soil to displace the sodium on the exchange and removing the displaced sodium by leaching. Calcium may be applied in the form of gypsum ($CaSO_4$) either as a dry powder or dissolved in the applied wastewater. The amount of gypsum to apply may be determined by a laboratory gypsum requirement test as described in the standard references. If a soil is calcareous, that is, containing calcium in the form of insoluble salts such as carbonates, sulfates, or phosphates, the calcium may be solubilized and made available for sodium displacement by the addition of acidulating chemicals--sulfur, sulfuric acid, or iron and aluminum sulfate. A comparison of these chemicals to gypsum is presented in Table 5-40.

TABLE 5-40

A COMPARISON OF CHEMICALS TO GYPSUM [99]

Amendment	Tons equivalent to 1 ton of gypsum
Sulfur, S	0.19
Nitrosol, 20% N, 40% S	0.47
Sulfuric acid, H_2SO_4	0.57
Limestone, $CaCO_3$	0.58
Lime-sulfur, 24% S	0.79
Gypsum, $CaSO_4 \cdot 2H_2O$	1.00
Alum, $Al_2(SO_4)_3 \cdot 17\ H_2O$	1.29
Ferrous sulfate, $FeSO_4 \cdot 7\ H_2O$	1.61

1 ton = 0.907 Mg

5.7.2.3 Salinity

The levels at which salinity becomes harmful to plant growth depend on the type of crop. Salinity in the root zone is controlled by leaching soluble salts to the subsoil or drainage system (see Section 5.6.2.3).

5.7.2.4 Nutrient and Trace Element Status

The nutrient status of the soil and the need for supplemental fertilizers should be periodically assessed. The levels of metals in the soil may be the factor determining the ultimate useful life of the system. University agricultural extension services may provide the service or recommend competent laboratories.

5.7.3 Vegetation

Plant tissue analysis is probably more revealing than soil analysis with regard to deficient or toxic levels of elements. All of the environmental factors that affect the uptake of an element are integrated by the plant, thus eliminating much of the complexity associated with interpretation of soil test results. If a regular plant tissue monitoring program is established, deficiencies and toxicities can be determined and corrective action can be taken. Detailed information on plant sampling and testing may be found in Walsh and Beaton [100] and Melsted [101].

5.8 Facilities Design Guidance

The purpose of this section is to provide guidance on aspects of facilities design that may be unfamiliar to some environmental engineers.

- Standard surface irrigation practice is to produce longitudinal slopes of 0.1 to 0.2% with transverse slopes not exceeding 0.3%.

  Step 1. Rough grade to ± 0.15 ft (5 cm) at 100 ft (30 m) grid stations.

  Step 2. Finish grade to ± 0.10 ft (3 cm) at 100 ft (30 m) grid stations with no reversals in slope between stations.

  Step 3. Land plane with a 60 ft (18 m) minimum wheel base, land plane to a "near perfect" finished grade.

- Specifications are available from the SCS for agricultural land leveling [102].

- Overland flow slopes should be graded to specification twice and checked for bulk density and degree of compaction to ensure relatively uniform conditions and prevent settlement during initial operation.

- If the site is large and intense rainfall is likely to occur, a minimum amount of finished slope should be prepared at any one time.

- Access to sprinklers or distribution piping should be provided every 1 300 ft (390 m) for convenient maintenance.

- Both asbestos-cement and PVC irrigation pipe are rather fragile and require care in handling and installation.

- Topsoil should be stripped and preserved during initial overland flow site grading, then replaced on the slope.

- Reed canary grass requires about 1 year to become established. A companion crop of orchard or rye grass is recommended for the first year.

- Tailwater return systems should be designed to distribute collected water to all parts of the field, not consistently to the same area.

- Screening should be provided for distribution pumping on the suction side to help prevent nozzle plugging.

- Diaphragm-operated globe valves should be used for controlling flow to laterals.

- All electric equipment should be grounded, especially when associated with center pivot systems.

- Automatic controls can be electrically, hydraulically, or pneumatically operated. Solenoid actuated, hydraulically operated (by the wastewater) valves with small orifices will clog from the solids.

- Use 36 in. (1 m) or larger valve boxes made of corrugated metal, concrete, fiberglass, or pipe material. Valve boxes should extend 6 in. (15 cm) above grade to exclude stormwater.

- Low pressure shutoff valves should be used to avoid continuous draining of the lowest sprinkler on the lateral.

- Automatic operation can be controlled by timer clocks. It is important that when the timer shuts the system down for any reason that the field valves close automatically and that the sprinkling cycles resume as scheduled when sprinkling commences. The clock should not reset to time zero when an interruption occurs.

- High flotation tires are recommended for land treatment systems. Allowable soil contact pressures for center pivot machines are presented in Table 5-41.

- Underdrains are only effective in saturated soil. If they are placed in a well to moderately-well drained soil above the water table, they will not recover any water.

TABLE 5-41

ALLOWABLE SOIL CONTACT PRESSURE

% fines	Contact pressure, lb/in.^{2}
20	25
40	16
50	12

Note: To illustrate the use of this table, if 20% of the soil fines pass through a 200-mesh screen, the contact pressure of the supporting structure to the ground should be no more than 25 lb/in.^{2}. If this is exceeded, one can expect wheel tracking problems to occur.

- Perforated continuous plastic pipe is generally more economical than clay tile or bell spigot pipe for underdrains.

- A filter sock placed over plastic drain pipe will help prevent clogging--a gravel envelope is unnecessary. Encrusting by iron, etc., can prove to be a problem over time.

- Plastic drainage pipe with cut or preformed openings is less likely to plug than pipe with punched openings.

- Maximum depth of placement for standard agricultural continuous drain-tile trenches is 5.5 ft (1.6 m). Bucket-type trenches are needed to place tile deeper.

- Intensive shallow drainage may be more economical than deep widely-spaced drains.

- Disking or harrowing soil surface about once per year can help maintain infiltration capacity.

- Plowing in "heavy" soils will develop a plowpan layer at the tip depth of the plow. Ripping or deep plowing at 2 to 4 year intervals may be necessary.

5.9 References

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CHAPTER 6

SMALL SYSTEMS

6.1 General Considerations

According to a 1973 survey, 54% of the land treatment systems were less than 1.0 Mgal/d (43.8 L/s) [1]. While the design principles of land treatment are the same for all sizes of systems, the approach should consider a system that is compatible with community resources, design and operational complexity, and possible environmental impact. The criteria presented in this chapter are principally intended for systems with a daily wastewater flow of 0.2 Mgal/d (8.8 L/s) or less but in some cases may be used for intermediate systems with flows from 0.2 to 1.0 Mgal/d (8.8 to 43.8 L/s). For treatment systems with flows greater than 1.0 Mgal/d (43.8 L/s), the additional design details presented in Chapter 5 should be considered. Sources for cost data are described in Chapter 3.

Small systems generally do not have full-time operators, so a design that requires a few days of field operator time per week or a few hours each day is desirable. In recognition of this, a small system may be designed somewhat conservatively, and hence should be less affected by climatic and wastewater variations than larger systems. Further, a conservative approach to design is often necessary because actual field data can be quite limited. If, for example, there is a range of soil permeabilities, the lower values should be selected for design. The capabilities of the system or the operators will generally not be sufficient to take advantage of varying site conditions to minimize costs. The type of information typically required for the design of a small system and sources of information are presented in Table 6-1.

6.2 Design Procedures

The design procedure for small systems follows a sequence of events as presented in Figure 6-1. The necessary information to complete each step is presented in the following sections.

6.2.1 Wastewater Characteristics and Flows

The determination of wastewater characteristics and flows is the initial design step. For existing treatment systems, the preferred method is to measure actual flows and wastewater characteristics. For systems under planning or construction, an estimate of important wastewater characteristics can be made with the aid of Table 6-2, using medium strength values for average domestic/commercial conditions. The strong values would apply for new systems with low water use and some minor industrial wastewater contributions. Weak values would be more applicable to systems where an older collection system with little or no industrial wastewaters and where infiltrating water results in dilution of the wastewater strength.

TABLE 6-1

TYPES AND SOURCES OF DATA REQUIRED FOR LAND TREATMENT DESIGNS

Type of data	Principal source
Wastewater data	Local wastewater authorities
Soil type and permeability	SCS soil survey
Temperature (mean monthly and growing season)	SCS soil survey, NOAA, local airports, newspapers
Precipitation (mean monthly, maximum monthly)	SCS soil survey, NOAA, local airports, newspapers
Evapotranspiration and evaporation (mean monthly)	SCS soil survey, NOAA, local airports, newspapers, agricultural extension service
Land use	SCS soil survey, aerial photos from the Agricultural Stabilization and Conservation Service, and county assessors' plats
Zoning	Community planning agency, city or county zoning maps
Agricultural practices	SCS soil survey, agricultural extension service, county agents
Surface and groundwater discharge requirements	State or EPA
Groundwater (depth and quality)	State water agency, USGS, driller's logs of nearby wells

Another source of dilution may be cooling waters and other low strength discharges from local industries. Special attention should be given to wastewater from nonhousehold sources that may contain constituents significantly different than those in Table 6-2. Characterization of nonhousehold wastewater should be made from field sampling, measurements at existing facilities, at some other similar facility, or from published values [2]. Significant amounts of nonhousehold wastewater may require additional design consideration from that given in this chapter.

FIGURE 6-1

SMALL SYSTEM DESIGN PROCEDURE

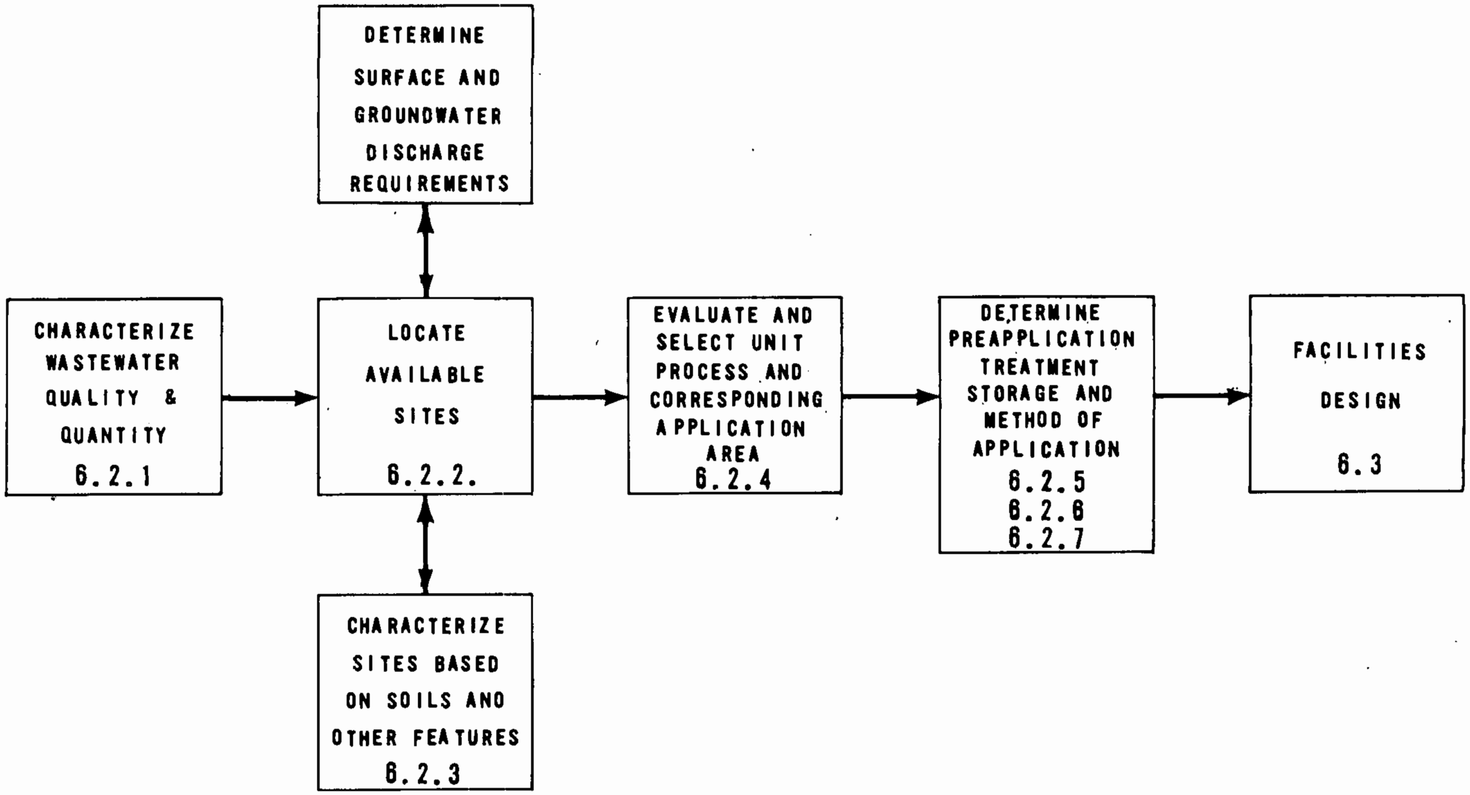


TABLE 6-2

IMPORTANT COMPONENTS OF DOMESTIC WASTEWATER [3]
mg/L

	Concentration		
	Strong	Medium	Weak
BOD_5, 20°C	300	200	100
Suspended solids	350	200	100
Nitrogen as N			
Organic	35	15	8
Free ammonia	50	25	12
Nitrates	0	0	0
Total	85	40	20
Phosphorus as P			
Organic	5	3	2
Inorganic	15	7	4
Total	20	10	6

The annual volume of wastewater will be used to estimate the application area. Due to the extremely variable nature, wastewater flows are best determined from field measurement. In cases where this is not possible, an estimate can be made from available data using a per capita or fixture basis [4]. The per capita basis is generally preferred. Typical flows from recreational facilities and institutional facilities are presented in Tables 6-3 and 6-4. A common value used to estimate daily wastewater flows is 75 gal/capita (284 L/capita) with a peaking factor of 4.0 for the peak flow [5]. Seasonal variations in flow should be considered in the estimate of annual wastewater volume and in discharge requirements.

6.2.2 Locate Available Sites

Identification of sites for small systems is usually much less complicated than that for larger systems. The search begins at the point of wastewater collection and radiates outward until one or more potentially suitable sites have been located. These sites may be identified by the following desirable features:

1. Fairly large tracts of undeveloped land or farms under a single ownership.

2. Land that is now or has been farmed, or is forested.

TABLE 6-3

DESIGN UNIT WASTEWATER FLOWS FOR RECREATIONAL FACILITIES, YELLOWSTONE NATIONAL PARK [3]

Establishment	Unit	Unit flow, gal/d
Campground (developed)	Person	25
Lodge or cabins	Person	50
Hotel	Person	75
Trailer village	Person	35
Dormitory, bunkhouse	Person	50
Residence homes, apartments	Person	75
Mess hall	Person	15
Offices and stores	Employee	25
Visitor centers	Visitor	5
Cafeteria	Table seat	150
Dining room	Table seat	150
Coffee shop	Counter seat	250
Cocktail lounge	Seat	20
Laundromat	Washing machine	500
Gas station	Station	2 000-5 000
Fish-cleaning station	Station	7 500

TABLE 6-4

AVERAGE WASTEWATER FLOWS FROM INSTITUTIONAL FACILITIES [3]

Institution	Avg flow, gal/capita
Medical hospital	175
Mental hospital	125
Prisons	175
High schools	20
Elementary schools	10

1 gal/capita = 3.78 L/capita

3. Location is relatively near point of wastewater collection.

4. Groundwater is more than 10 ft (3 m) deep or there is a nearby water body that could be used to receive the underdrainage needed to lower the water table and to receive the percolated effluent.

5. Land that is already for sale or that can be bought with reasonable negotiations.

6. Zoning that is compatible with land treatment facilities requirements, such as areas zoned for greenbelts.

7. Existing irrigated lands (e.g., golf courses, parks, highway landscaping).

8. Access from developed roads and power supply.

At this point in the site investigations neither the land treatment process nor the total land area is known. In order to make some initial assessment of the sites, some preliminary estimate of area is needed. Guidelines to land area needs for preliminary site identification are provided in Table 6-5. These values are for screening purposes only and must be refined as the study progresses.

TABLE 6-5

TOTAL LAND AREA GUIDELINES FOR PRELIMINARY SITE IDENTIFICATION

Avg design flow, gal/d	Land area, acres			
	Slow rate		Rapid infiltration	Overland flow
	6 mo/yr	12 mo/yr	12 mo/yr	10.5 mo/yr
100 000	15-30	7.5-20	0.5-6	3-10
200 000	30-50	15-40	1-12	5-20
300 000	40-80	20-60	1.5-20	10-30
500 000	60-150	30-100	2.5-30	15-50
750 000	100-200	50-150	4-45	25-75
1 000 000	150-300	75-200	5-60	35-100

100 000 gal/d = 4.38 L/s
1 acre = 0.405 ha

6.2.3 Site Characterization

Having identified the potential sites, the next step is to systematically describe the site characteristics. These characteristics and the required effluent quality requirements will combine to suggest the type of land treatment process that should be used.

Site characteristics that should be noted include the following:

1 Soils--type, distribution, permeability of most restrictive layers, physical and chemical characteristics, and depth to groundwater

2. Available land area, both gross and net areas (i.e., excluding roads, rights-of-way encroachments, stream channels, and unusable soils)

3. Distance from source of wastewater to site, including elevation differential

4. Topography, including relief and slopes

5. Proximity of site to industrial, commercial, residential developments; surface water streams; potable water wells; public use areas such as parks, cemeteries, or wildlife sanctuaries

6. Present and future land uses

7. Present vegetative cover

6.2.4 Select Land Treatment Process

The selection of the appropriate unit process depends primarily on the following two conditions:

1. Soil characteristics at the prospective site

2. The requirements of the discharge permit or groundwater quality

Obviously, other conditions such as other site features, total land area, operating personnel, and related economic and environmental factors, combine to help form the final conclusion. A decision matrix for forming preliminary conclusions on the land treatment process based on technical considerations only is presented in Table 6-6. Other related conditions can then be used to finalize the decision.

The preferred land treatment options for small systems are, in order: slow rate, rapid infiltration, and overland flow. Other treatment processes have been used to treat wastewater in research and demonstration projects but applicable design criteria are not generally available. Slow rate systems are the first design choice because of the similarity to normal agricultural practices, and their performance is the least sensitive to operational changes so that treatment reliability under variable conditions is greatest. Rapid infiltration systems are

TABLE 6-6

PRELIMINARY SELECTION OF LAND TREATMENT SYSTEMS

Levels of effluent quality (NPDES permit), mg/L	Range of soil permeability, in./h						
	<0.06	0.06-0.2	0.2-0.6	0.6-2.0	2.0-6.0	6.0-20.0	>20.0
≤BOD = 4 ≤SS = 2 ≤N = 4 ≤P = 0.1	....	Slow rate	Slow rate	Slow rate	Slow rate	....	....
≤BOD = 5 ≤SS = 5 ≤N = 15 ≤P = 1	....	....	....	....	Rapid infiltration	Rapid infiltration	Rapid infiltration
≤BOD = 10 ≤SS = 10 ≤N = 3 ≤P = 5	Overland flow	Overland flow	....	....	....	....	....
No surface discharge[a]	....	Slow rate	Slow rate	Slow rate	Slow rate Rapid infiltration	Slow rate Rapid infiltration	Slow rate Rapid infiltration

a. Discharge to groundwater or indirect discharge to surface water.

1 in./h = 2.54 cm/h

the second choice in small scale systems because removals of most wastewater components are excellent with low operation and maintenance requirements. A consistent level of nitrogen removal, however, is more difficult to obtain than with other systems. In some groundwater aquifers nitrogen content is of little concern, greatly enhancing the use of rapid infiltration systems. Overland flow systems require the greatest level of on-site management to maintain high levels of treatment so extra operator training is required, particularly for proper maintenance of the terraces.

After selecting the unit process, the required "wetted" or application land area can be computed. In general, this calculation requires development of the hydraulic application rate and the duration of application during the year. It also requires consideration of additional applied water in the form of precipitation and the lost water due to percolation and evapotranspiration. This computation is usually combined with a water balance computation for determining storage requirements. For each treatment system this procedure is somewhat different. Therefore, computations of wetted land area are discussed separately for each process and summarized in Table 6-7.

TABLE 6-7

SUMMARY OF APPLICATION PERIODS FOR LAND TREATMENT SYSTEMS

Unit process	Crop management	Application: Description	Application: Estimated period
Slow rate	Annual crop	Growing season only	3-5 months
	Double crop or perennials	All year unless restricted by weather or planting and harvesting	6-12 months[a] (also see Figure 6-2)
Rapid infiltration	NA	All year-round, if in free draining materials	12 months
Overland flow	Perennial grasses	All year unless restricted by weather	See Figure 6-2

NA = not applicable.

a. This period is maximum in semiarid areas. The lower values should be used where winters are severe.

6.2.4.1 Application Area For Slow Rate Systems

The application area for slow rate systems is based on a weekly application rate and the length of the application season. The permeability of the predominant soil types combined with crop water use determine a weekly application rate, as shown in Table 6-8. Water use requirements of most crops will be met using the rates presented in Table 6-8.

TABLE 6-8

DESIGN APPLICATION RATES FOR SMALL SYSTEMS

SCS permeability class	SCS permeability range, in./h	Application rate, in./wk: Slow rate[a]	Application rate, in./wk: Rapid infiltration[b]	Application rate, in./wk: Overland flow[c]
Very slow	<0.06	....	..	4-8
Slow	0.06-0.2	0.5-1.0	..	4-8
Moderately slow	0.2-0.6	1.0-1.5	..	.....
Moderate	0.6-2.0	1.5-3.0	..	.....
Moderately rapid	2.0-6.0	3.0-4.0	4-20	.....
Rapid	6.0-20	4.0	8-30	.....
Very rapid	>20	....	12-40	.....

a. Application during growing season.

b. Year-round application

c. Volume applied equally during 5 to 7 days per week; low value for screened effluent and higher rates for primary and biological treatment effluent.

1 in./wk = 2.54 cm/wk

The length of the application season should be computed on the basis of intended management. Two management techniques are commonly practiced:

1. Grow a single, annual crop

2. Grow perennial forage grasses, practice double-cropping, or use the no-till management system

For a single annual crop, the application period will be the growing season plus any preplanting or after harvest irrigation and could result in an application period as short as 3 months. For this reason, the second management technique is generally used.

For the second case, the application season is determined from climatic data given in a county soil survey or other local source, for the proposed vegetation. The mean growing season, i.e., the number of weeks between the last 32°F (0°C) occurrence in the spring and the first 32°F (0°C) occurrence in the fall, is used for all annual crops. Typical annual crops used in the United States with land treatment systems are corn, wheat, barley, cotton, and soybeans. To extend the application period for annual crops, they may be double-cropped, or winter or spring cover crops may be planted after harvesting. Perennial crops are typically forage grasses such as Bermuda grass, orchard grass, tall fescue, Reed canary grass, and alfalfa. Wastewater can be applied between occurrences of 26°F (-3.3°C) temperatures in the spring and fall. The application period should be reduced by 30 to 45 days to allow for planting (annual crops only) and harvesting periods. The annual application volume is determined by multiplying the weekly rates from Table 6-8 by the length of the application season in weeks. The annual application rate determines the required application area according to the following equation:

$$F = \frac{36.8\ Q}{LR} \qquad (6\text{-}1)$$

where F = field area, acres (ha)
Q = annual flow, Mgal/yr (m^3/yr)
L = period of application wk/yr
R = rate of application, in./wk (cm/wk)

$$36.8\ (0.01) = \text{conversion factor} = 3.06\ \frac{\text{acre-ft}}{\text{Mgal}} \times \frac{12\ \text{in.}}{\text{ft}}$$

6.2.4.2 Application Area For Rapid Infiltration Systems

Where application of wastewater to an infiltration basin is by flooding, the period of application is the entire year. An exception may occur under one of the following conditions:

1. The soil is fine textured or not free draining so freezing of water within the soil pores renders it impermeable.

2. The water is applied by sprinkler methods, and the droplets freeze and coat the surface with ice.

3. There is a severe low temperature resulting in freezing of water in the distribution piping or as it exits.

Although some provision is recommended for storage to account for one of the above events, the application period can be assumed as 12 months. The application rate can be selected from Table 6-8 based on soil permeability. Then, using Equation 6-1 and an application period of 52 weeks, the application area can be computed.

6.2.4.3 Application Area For Overland Flow Systems

This process requires an effluent discharge to either a surface water body or another unit process. Consequently, application rates are not dependent on soil permeability but rather on biological activity. Experience has indicated that an application rate of 4 in./wk (10 cm/wk) will easily match biological activity on the prepared slopes.

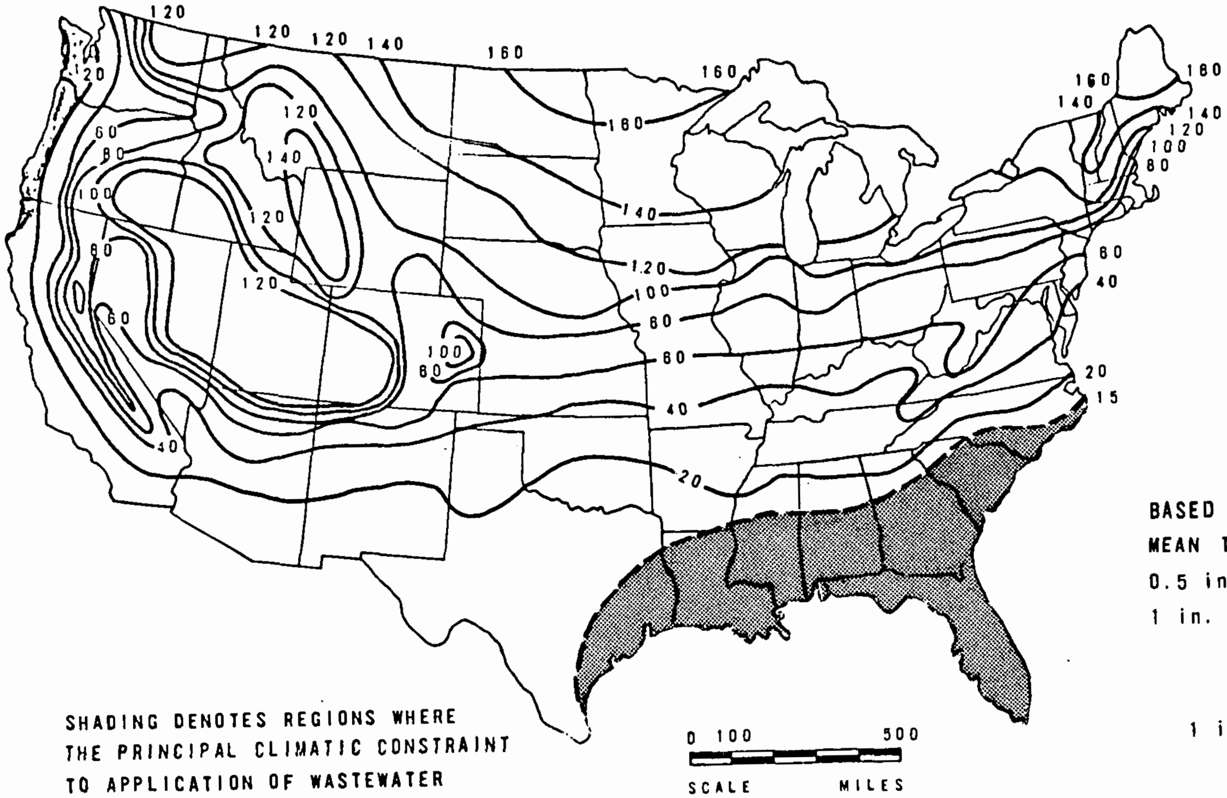
The application period is usually determined by climatic conditions. These conditions are similar to those for perennial grasses with slow rate systems. In general, Figure 6-2 can be used to estimate the number of days that overland flow cannot operate. Subtracting this period in weeks from 52 wk/yr will result in the application period. Using Equation 6-1, the wetted area can be computed.

6.2.5 Preapplication Treatment

Preapplication treatment is desirable for small scale systems to control nuisance and odor conditions during storage with slow rate and overland flow systems, and to lessen bed maintenance on rapid infiltration systems. Biological treatment is often employed with many forms of land treatment but may be avoided with overland flow. Also, rapid infiltration may be used with only primary level treatment but the application rate must be reduced somewhat over that of secondary level because of the clogging effect of suspended solids. The use of primary

FIGURE 6-2

ESTIMATED WASTEWATER STORAGE DAYS BASED ONLY ON CLIMATIC FACTORS [6]



SHADING DENOTES REGIONS WHERE THE PRINCIPAL CLIMATIC CONSTRAINT TO APPLICATION OF WASTEWATER IS PROLONGED WET SPELLS

BASED ON 32°F (0°C) MEAN TEMPERATURE
0.5 in./d PRECIPITATION,
1 in. OF SNOWCOVER

1 in. = 2.54 cm

effluent is recommended, but if land area is limited it may be necessary to provide a higher level of preapplication treatment. A suggested guide to the selection of preapplication treatment levels for each land treatment process is presented in Table 6-9.

TABLE 6-9

MINIMUM PREAPPLICATION TREATMENT PRACTICE

Process	Preapplication treatment
Slow rate	
Surface application	Primary sedimentation
Sprinkler application	Primary or biological[a]
Rapid infiltration	Primary
Overland flow	Bar screens and comminution

a. Typically oxidation ponds or aerated lagoons.

6.2.6 Storage Requirements

Storage volume estimates must include consideration of the total water balance for the year. However, the designer can approximate this storage by referring to Figure 6-2 and selecting the proper values for the geographical location in question. The values taken from the figure represent days of storage for the worst year in 20, based on severity of winter conditions. Storage requirements may be further reduced by seasonal discharges to surface waters if permitted by the state. Storage volume guidelines are summarized in Table 6-10.

TABLE 6-10

GUIDELINES FOR STORAGE VOLUMES

Land treatment process	Storage volume guidelines
Slow rate	
Annual crops	Up to 9 months of flow
Perennial crops	0.5-6 months of flow, see Figure 6-2
Rapid infiltration	7-30 days of flow
Overland flow	See Figure 6-2

6.2.7 Selection of Application Systems

In preliminary design for slow rate, the method of applying the water must be decided. Surface application is preferred where the site topography is quite flat or is suitable for application with a minimum amount of leveling. This method of application offers the least capital cost and the least operation and maintenance cost for most systems. Also, there should be no problems with aerosol transport or need for buffer zones.

Sprinkler application may be used for almost any topography, but preferably one having slopes of less than 15% to minimize difficulties with effluent runoff and erosion control. For small systems, the use of surface application systems is preferred for both rapid infiltration and overland flow treatment.

6.2.8 Postapplication Treatment

In those cases where effluent is collected for discharge to surface waters, discharge requirements must be met. Systems with overland flow may require postdisinfection. Disinfection may be accomplished using hypochlorinators or, in some cases, an erosion feeder type of chlorinator may be used. The latter units have not been widely accepted but may offer suitable reliability for very small systems.

6.3 Facilities Design

As in other parts of this manual, no attempt will be made to discuss the detailed design of preapplication treatment and storage facilities. The discussion is limited to the distribution and application systems. In addition to the comments contained in this chapter, the reader is directed to Section 5.8 for detailed design guidance. Distribution and application systems will be discussed for each land treatment process in the following section.

6.3.1 Slow Rate System

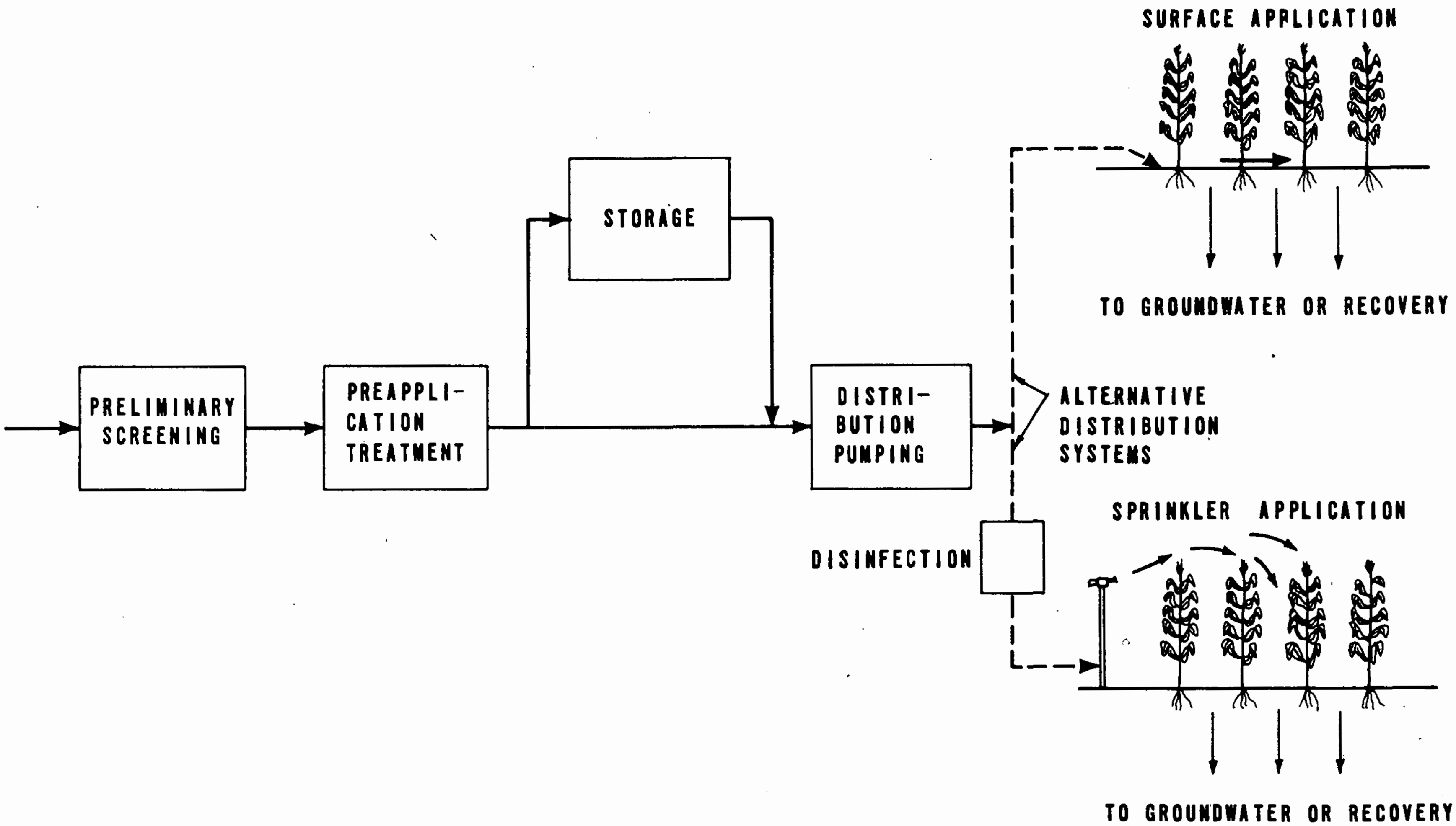
A schematic diagram showing the typical elements of a slow rate system is presented in Figure 6-3.

6.3.1.1 Surface Application Systems

Surface application systems require site-specific design, while sprinkler system design should be based on consultation with the

FIGURE 6-3

SCHEMATIC FOR TYPICAL SLOW RATE SYSTEM



equipment manufacturers. The general factors involved in the final layout and design of a surface application system are presented in Table 6-11. Most of the common surface irrigation systems are included in this table.

TABLE 6-11

FACTORS AFFECTING THE DESIGN OF SURFACE IRRIGATION SYSTEMS [7]

	Maximum slope, %				Water application rate of intake family, in./h[a]			Adaptable to		
	Humid areas		Arid areas							
	Nonsod crops	Sod crops	Nonsod crops	Sod crops	Minimum	Maximum	Shape of field	Row crops (row or bedded)	Sown, drilled, or sodded crops	Orchards and vineyards
Level										
Level border	Nearly level		Nearly level		0.1	2.0	Any shape	Yes	Yes	Yes
Contour levee	0.1	0.1	0.1	0.1	0.1	0.5	Any shape	Yes	Yes	Yes
Level furrow	Nearly level		Nearly level		0.1	2.0	Rows should be of equal length	Yes	Yes	Yes
Graded										
Graded border	0.5	2.0	2.0	4.0	0.3	2.0	Rectangular	No	Yes	Yes
Contour ditch	NA	4.0	4.0	15.0	0.1	3.0	Any shape	No	Yes	Yes
Graded furrow	0.5	NA	3.0	NA	0.1	3.0	Rows should be of equal length	Yes	Yes	Yes
Corrugation	NA	NA	4.0	8.0	0.1	1.5	Rectangular	No	Yes	Yes
Contour furrow	Cross slope 3.0	3.0	Cross slope 6.0	6.0	0.1	2.0	Rows should be of equal length	Yes	No	Yes

NA = not applicable.

a. Intake family is a grouping of soil by the SCS. It is based on the ability of the soil to take in the required amount of water during the time it takes to irrigate.

1 in./h = 2.54 cm/h

The most desirable design is one in which the furrows or border checks are so flat that failure to rotate the flow to the next field in the system would not result in wastewater escaping the property. In other words, the field would be flat enough to permit an enclosing or containment levee around the field. Alternative choices are tailwater control systems or a gravity return to the lagoon at the lower end of the site. If this is not possible, closer supervision of the operation will be necessary to minimize the risk of nuisance conditions occurring.

Any of the common low head or gravity design pipe materials should be suitable for transporting the wastewater to the field. Gated aluminum pipe is an effective means of distributing the water uniformly to border checks or furrows. Open concrete lined ditches with turnouts have been used effectively with small systems.

6.3.1.2 Sprinkler Application Systems

If a sprinkler irrigation system has been selected, the designer should work closely with one or more sprinkler manufacturer's vendors who will aid the designer in the use of their respective equipment. The availability of a knowledgeable local representative may weigh heavily in the final selection of equipment.

A list of most of the common types of sprinkler systems, guidelines for their application, and limitations is presented in Table 6-12. Additionally, some states have published regulations regarding preapplication disinfection, minimum buffer areas, and control of public access for sprinkler systems. These criteria should be reviewed for applicability.

Distribution systems may consist of any of the common pressure pipe materials, such as plastic, aluminum, asbestos-cement, lined and coated steel, or ductile iron.

The final design analysis should consider the following points:

1. Provision of adequate thrust blocks
2. Consideration of water hammer and surge conditions
3. Winter operation criteria
4. Provisions of sufficient valves and manifolding to permit proper agricultural management of the field
5. Automatic timers to limit the application in any one area
6. Alarms to signal system failures
7. Protection against plugging by algae

6.3.2 Rapid Infiltration Systems

Small scale rapid infiltration systems typically apply an annual application of 17 to 173 ft (5 to 53 m) of wastewater at rates of 4.0 to 40.0 in./wk (10 to 100 cm/wk). The application rate is determined from soil permeability data. It is preferable to dig at least one test pit (see Appendix F) and conduct three infiltration tests (see Appendix C). Multiple infiltration basins are required to permit intermittent application. A desirable basin design should provide sufficient flexibility to permit a 1 to 4 day application period followed by a 7 to 14 day drying period.

TABLE 6-12

FACTORS AFFECTING THE DESIGN OF SPRINKLER IRRIGATION SYSTEMS [7]

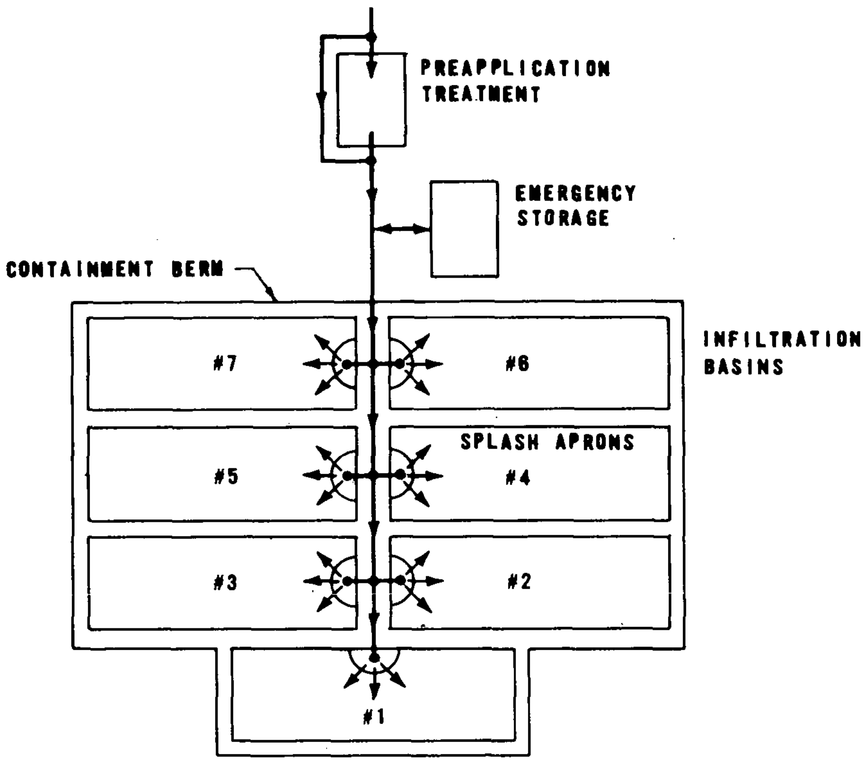
System	Maximum slope, %	Water application rate, in./h		Shape of field	Field surface conditions	Maximum height of crop, ft	Size of single system, acres
		Minimum	Maximum				
Multisprinkler							
Hand moved							
Portable set	20	0.10	2.0	Rectangular	No limit	No limit	1-40
Solid set	20	0.05	2.0	Any shape	No limit	No limit	1+
Tractor moved							
Skid mounted	5-10	0.10	2.0	Rectangular	Smooth enough for	No limit	20-40
Wheel mounted	5-10	0.10	2.0	Rectangular	safe tractor operation	No limit	20-40
Self moved							
Side wheel roll	5-10	0.10	2.0	Rectangular	Reasonably	4	20-80
Side move	5-10	0.10	2.0	Rectangular	smooth	4-6	20-80
Self propelled							
Center pivot	5-15	0.20	1.0	Circular	Clear of obstructions,	8-10	40-160
Side move	5-15	0.20	1.0	Square or rectangular	path for towers	8-10	80-160
Single sprinkler							
Hand moved	20	0.25	2.0	Any shape	No limit	No limit	20-40
Tractor moved							
Skid mounted	5-15	0.25	2.0	Any shape	Safe operation	No limit	20-40
Wheel mounted	5-15	0.25	2.0	Any shape	of tractor	No limit	20-40
Self propelled	No limit	0.25	1.0	Rectangular	Land for winch and hose	No limit	40-100
Boom sprinkler							
Tractor moved	5	0.25	1.0	Any shape	Safe operation of tractor	8-10	20-40
Self propelled	5	0.25	1.0	Rectangular	Lane for boom and hose	8-10	40-100
Permanent	No limit	0.05	2.0	Any shape	No limit	No limit	1+

1 in./h = 2.54 cm/h
1 ft = 0.305 m
1 acre = 0.405 ha

A typical rapid infiltration system is illustrated in Figure 6-4. The layout is based on a relatively level land surface. Sloping lands should utilize gravity flow to minimize pumping costs. Site layout should locate the maximum bed dimension perpendicular to probable groundwater flow. Steeper land would require smaller basins to minimize cut and fill and to avoid cross-basin subflows according to the following schedule: for 0 to 1% average cross-slope--1 basin; 2%--2 basins; 3%--4 basins. Slopes in excess of 3% should be graded to a 3% average before final basin construction.

FIGURE 6-4

SCHEMATIC FOR TYPICAL
RAPID INFILTRATION SYSTEM



Surface design details related to infiltration basins are similar to those for sludge drying beds with care being taken to minimize severe erosion at the point of application. Provision for access to the basin should be included to permit entry of a tractor with a disk, harrow, or other scarifying equipment. The need for underdrains should be determined based on Appendix C and details for design are presented in Chapter 5 (Sections 5.5.2 and 5.8).

6.3.3 Overland Flow System

Although the preferred method of applying wastewater to the field is by surface method, many industrial installations now exist with sprinkler systems. The typical overland flow system, with alternative application systems is illustrated in Figure 6-5. To provide the maximum treatment efficiency, wastewater must be applied at least once a day and in sufficient quantity to wet the entire terrace area.

A suggested method of supporting distribution piping is shown in Figure 6-6. In this case, the stone serves as a support, but it also serves as a means to convert a point discharge into sheet flow, minimizing erosion and maximizing treatment efficiency.

Where sprinklers are used, they should be placed downslope from the highest point on the terrace a distance equal to the radius of the sprinkler, unless one-half circle sprinklers are used.

Probably the most important feature of the overland flow system is the sloped terrace. This slope must be as nearly equal to a plane surface as possible and sloped in such a way as to prevent short-circuiting of the wastewater and standing water in the collection ditches. No swales, depressions, or gullies can be permitted; otherwise, water will pond and permit propagation of mosquitos or the production of odors.

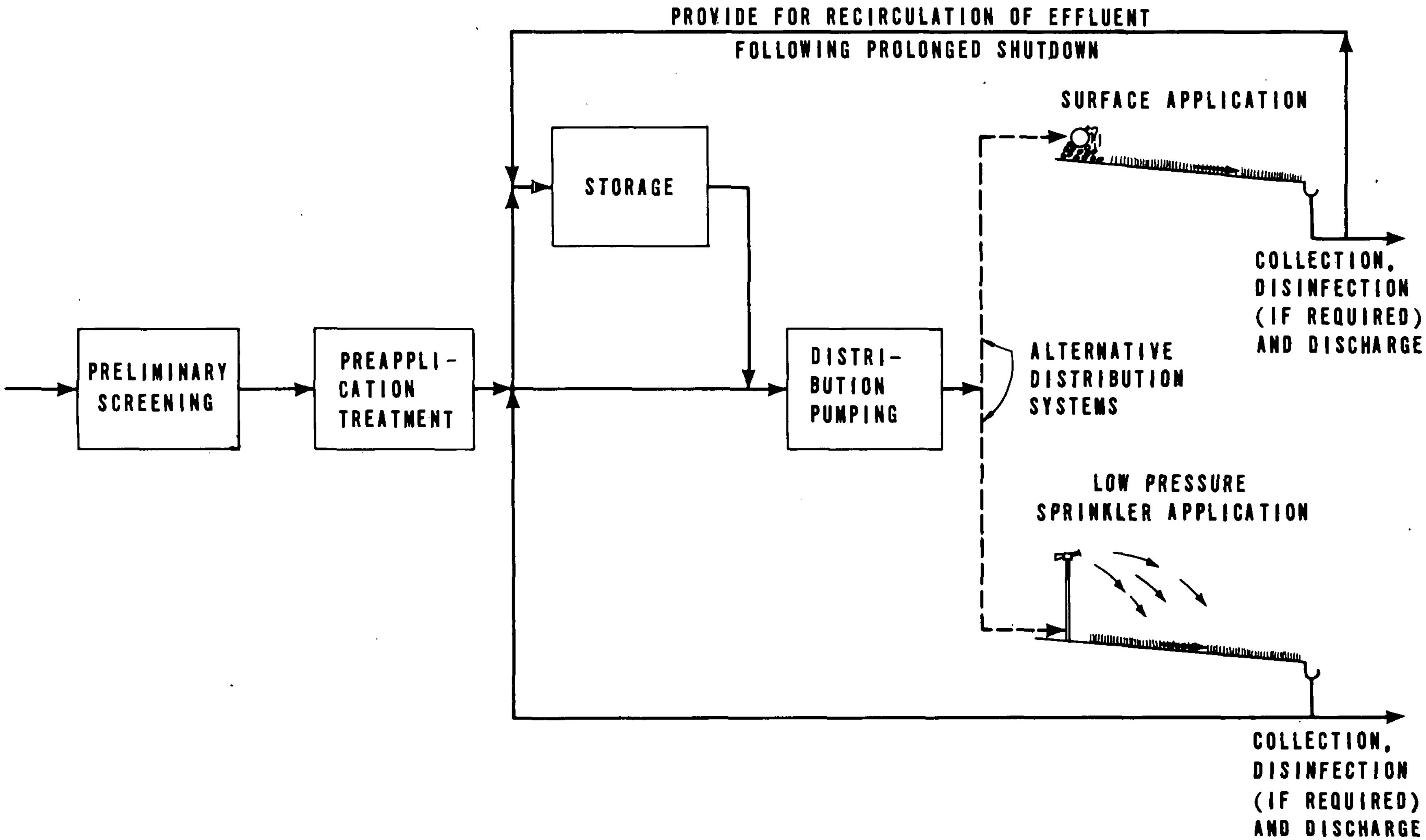
The second factor is the cover crop. Grasses must be selected for their resistance to continuously wet root conditions. Also, their growth should not be in clumps as this will result in the formation of rivulets of flow rather than a uniform sheet flow. Common grasses for this purpose have been Reed canary grass, Italian rye, red top, tall fescue, and Bermuda grass.

The distribution system should be designed to permit application on each portion of the field for from 6 to 12 h/d. This application period is based on convenience rather than for treatment reasons. The system must be valved and manifolded to permit a portion of the field to be taken out of service for grass mowing and/or harvesting. During that period, the remainder of the field must take the total flow or else it must be diverted to temporary storage. Following prolonged shutdown, the wastewater collected in the drainage ditches may have to be recirculated through the treatment system until discharge requirements are again being met. This would only be required where stringent discharge requirements are imposed.

Site access requires special equipment with broad tires having low pressure (less than 10 $lb/in.^2$ or 7 N/cm^2) to avoid creating ruts

FIGURE 6-5

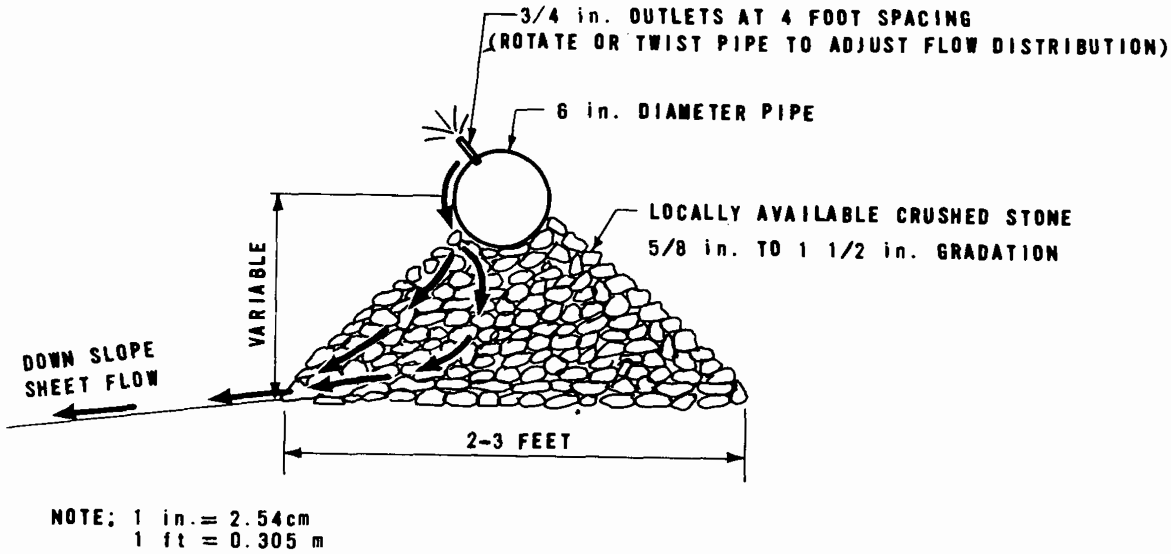
SCHEMATIC FOR TYPICAL OVERLAND FLOW SYSTEM



that would short-circuit the flow and the treatment process. Vegetation harvest and removal is not always necessary, since the vegetation can be cut with a chopping mower just prior to maturity (every 4 to 6 weeks) and allowed to decompose on the terrace [8]. Applications should be reduced for 3 or 4 days after winter shutdowns.

FIGURE 6-6

BUBBLING ORIFICE DISTRIBUTION
FOR OVERLAND FLOW



6.4 Small System Design Example*

6.4.1 Setting

The community of Angus, Washington, has decided to construct a land treatment system to meet its wastewater discharge specifications. The following information is known:

Present (1977) population - 2 234
Projected 1997 population - 3 400
Annual rainfall - 48 in. (120 cm), evenly distributed throughout the year
Warm season evaporation - 24 in. (60 cm), May 1 to October 1
Seasonal flow variations -
  Maximum month (August) - 1.5 of average
  Minimum month (January) - 0.8 of average

*Note: This is an example; it is intended for illustration only.

There is an elementary school of about 200 pupils and 15 staff. The high school is located in Hereford, about 10 miles (16 km) to the south. There are some small commercial establishments such as service stations and restaurants but the town's only industry, a sawmill, treats and recycles its own wastewater. As the town is presently sewered by individual systems, mostly septic tanks, a new collection system will be constructed. Water service is unmetered and estimated consumption is 100 gal/capita·d (378 L/capita·d).

6.4.2 Wastewater Quality and Quantity

Assume:

1. Design for 1997 population
2. Projected pupil and staff population will be 325
3. Due to unmetered water system, waste is relatively high and the wastewater will be of medium strength through the planning period (BOD = 200 mg/L).

Wastewater quality can be found in Table 6-2, second column. The flowrate for average design conditions, using an estimated daily wastewater flow of 75 gal/capita (284 L/capita), is calculated to be:

$$325\ (10)\ \text{gal/capita·d} + 3\ 400\ (75)\ \text{gal/capita·d}$$
$$= 258\ 250 \quad \text{say } 260\ 000\ \text{gal/d}$$

6.4.3 Locate Available Sites

By interpolation of the values in Table 6-5 for a flowrate of 260 000 gal/d the following preliminary site areas were determined:

- Slow rate, 26 to 52 acres (10.5 to 21 ha)
- Rapid infiltration, 3 to 13 acres (1.2 to 5.3 ha)
- Overland flow, 9 to 27 acres (3.6 to 10.9 ha)

As there is sufficient open space and farmland in the immediate area, it was decided to limit the search for available sites to a radius of 1 mile (1.6 km) from the lowest point in the collection system to minimize transmission costs.

6.4.4 Site Characterization

After the search, four potential tracts were located, all about 0.5 mile (0.8 km) from the designated point. The characteristics of the sites are summarized in Table 6-13.

TABLE 6-13

POTENTIAL LAND TREATMENT SITES FOR ANGUS, WASHINGTON

Site No.	No. of owners	Current use[a]	Size, acres[b]	Minimum depth to groundwater, ft[c]	Range of soil permeabilities, in./h[d]	Slope, average-maximum, %[d]	Remarks
A	5	Agriculture	200	15	0.6-20	2-10	15 acres at 5-10 in./h
B	1	Undeveloped land	95	10	0.2-2.0	Flat	Potential greenbelt
C	1	Nonirrigated pasture	300	20	0.06-0.5	5-15	Low permeability results in local wet spots
D	2	Low density housing	30	10	0.2-0.6	4-5	One house zoned residential

Sources: a. Local planning agency and site visit.
b. County assessor.
c. Well logs.
d. SCS report.

1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm

On the basis of acreage requirements, all sites except Site D would have sufficient acreage for all systems. Site D would be marginal for a slow rate system and would require additional soil testing if it is found to be the most desirable location.

Sites A, B, and C appear to have area in excess of the anticipated needs. The treatment site and facilities could be located in the most desirable location within the site and not all of the land would have to be purchased.

6.4.5 Select Land Treatment Process

Discharge would have to be either to the groundwater or to nearby White River, downstream of the water supply intake line. The groundwater is being used for some small irrigation wells near the sites and some individual domestic wells 1 to 2 miles (1.6 to 3.2 km) away. It is anticipated that Angus may some day use groundwater as well as the surface supply to meet future water demands. Therefore, discharges to the groundwater would have to meet existing requirements.

For discharge to White River, the Department of Ecology, State of Washington, has stipulated the following effluent quality limitations:

BOD - 10 mg/L
SS - 10 mg/L
Total N - 15 mg/L
Total P - 5 mg/L
Fecal coliforms - 200/100 mL
Maximum residual chlorine - 0.5 mg/L

Using Table 6-6, a slow rate or rapid infiltration system is applicable to Site A. Sites B and C could have either a slow rate or overland flow system, while a slow rate system appears to be the only choice for Site D. As all systems are capable of meeting or exceeding the discharge requirements, system selection (except for rapid infiltration) will be based on the soil permeability ranges.

For rapid infiltration rates on Site A, limited field tests are needed. Using the guidelines in Appendix F one test pit was dug down to 10 ft (3 m) to verify lack of restrictive layers in the soil profile. Using the procedures from Appendix C (Section C.3.1.4.3) three double ring infiltrometer tests were conducted. The resulting infiltration minimum rate was determined to be 5.0 in./hr (12.7 cm/h). Using Figure 3-3, the wastewater application rate is determined to be 42.0 in./wk (1.1 m/wk) which is 5% of the clear water rate (the range in Figure 3-3 is 3 to 10%). However, an upper limit of 40.0 in./wk (1.0 m/wk) is recommended for small rapid infiltration system design. Based on 52 wk/yr operation the annual rate is 173 ft/yr (52.7 m/yr).

To compute the area required to treat the wastewater, Equation 6-1 is used. In keeping with the conservative approach to small system design, the lower permeability values are used to obtain the equivalent application rate, R, from Table 6-8. Also, the average annual rainfall, including a reduction for evapotranspiration (assuming a colder month), is added to the rate of application R. As an example, a calculation to derive the area required for a slow rate system at Site A is shown:

L = 46 weeks (52 for rapid infiltration - Figure 6-2)
R = SCS permeability range plus (precipitation-evapotranspiration)
= (1.5 + 0.5) in./wk = 2.0 in./wk (5.0 cm/wk)
Q = 0.26 Mgal/d x 365 = 94.9 Mgal/yr

$$F = \frac{36.8 \times 94.9}{46 \times 2} = 38 \text{ acres } (15.4 \text{ ha})$$

The acreages calculated for each site and treatment process are shown in Table 6-14.

TABLE 6-14

REQUIRED ACREAGES FOR ANGUS LAND TREATMENT SYSTEMS

Site		Soil permeability, in./h	Application rate, in./wk	Treatment site requirement, acres
A	Slow rate	0.6	2.0	38
	Rapid infiltration	5.0	40.0	1.7
B	Slow rate	0.6	2.0	38
	Overland flow	0.2	8.0	10
C	Slow rate	0.2	1.5	51
	Overland flow	0.06	4.0	19
D	Slow rate	0.2	1.5	51

1 in. = 2.54 cm
1 acre = 0.405 ha

In comparing the required acreages with Table 6-13, Site D is eliminated on the basis of insufficient area.

6.4.6 Preapplication Treatment, Storage, and Application Methods

Preapplication treatment practices for various systems are indicated in Table 6-9. Unless surface application is selected for one of the slow rate systems, biological treatment with a minimum of 7 days detention time is normally practiced. This would most likely be an aerated lagoon designed to reduce BOD_5 to 60 mg/L or less.

From Figure 6-2, a storage of 40 days of flow is advised. The required storage volume is as follows:

260 000 gal/d x 40 days = 10.4 Mgal = 32 acre-ft (39 500 m^3)

Storage pond size, assuming 10 ft working depth (3 ft freeboard)
= 3.2 acres + 25% for levees, road = 4 acres (1.6 ha)

This will be required for either the slow rate or overland flow systems. The rapid infiltration system should not require any storage capacity because of the moderate climate and permeable soils. The methods of application for surface irrigation systems are summarized in Table 6-11 for various land conditions. By comparing the application rates from the third column of Table 6-14 and the average to maximum slopes from Table 6-13 to the values given in Table 6-11, the following surface application methods are chosen:

Site A - Graded border with any crop (minimum slope)
- Graded contour levee with sod crops (for maximum slopes)

Site B - Level border checks with any crop
Site C - Graded contour levee with sod crops

As the land preparation costs for Sites A and C would raise the development cost beyond that required for Site B, the use of sprinkler application should be investigated. Using the procedure outlined above, the remaining sites are screened using the values given in Table 6-12 for sprinkler systems. The preferred methods from this process are:

Site A - Hand or tractor moved solid set
Site C - Hand moved solid set, self propelled, and permanent set

The land at Site A, already in agricultural use, contains some uniform, rectangular fields. Site C would require grading and preparation (i.e., more cost) to enable the use of the more flexible, less expensive sprinkler systems.

The rapid infiltration system for Site A and the overland flow system for Site C are still feasible alternatives according to the criteria in Sections 6.3.2 and 6.3.3. The need to construct the 2 to 4% sloped terraces at Site B for effective overland flow would favor the less expensive grading required to prepare Site C.

A number of other considerations should be included in the cost-effectiveness analysis for small systems that are discussed in other sections of this manual. Recovery of renovated water from beneath rapid infiltration basins or slow rate sites to either provide relief drainage from perched groundwater or to recover water for sale is discussed in Section 5.5. For vegetation selection, and revenue generation by management of a cash crop, Section 5.6 should be reviewed. The systems that remain to be analyzed for cost effectiveness are summarized in Table 6-15.

TABLE 6-15

LAND TREATMENT ALTERNATIVES FOR ANGUS, WASHINGTON

Site	Land treatment system	Major feature
A	Slow rate	Tractor moved solid set sprinklers
	Rapid infiltration	Prepare the 15 acre-site
B	Slow rate	Level border strips
C	Slow rate	Hand moved solid set sprinklers
	Overland flow	Grade terraces and ditches

6.4.7 Other Considerations

On the basis of nonmonetary criteria, Site B holds a clear advantage--it is already owned by the city and would provide needed irrigation water for the future greenbelt.

If rapid infiltration is shown to be more cost effective than irrigation at Site A, the purchase or lease of the site would be necessary. For the irrigation system at Site A, purchase would not be necessary if a long-range contract could be negotiated with the owner. Since the land at Site A is presently being irrigated, the renovated water could be offered at an equal cost and the farmer would have the added advantage of the wastewater nutrients.

Site C would require the purchase or lease of a suitable area if the overland flow alternative were shown to be more cost effective. Sprinkler irrigation would convert nonirrigated pasture into more valuable land and should make a long-term lease more attractive to the owner than was the case at Site A.

If some alternatives appear to be very close to each other for cost effectiveness, the consideration of these nonmonetary items may be the basis for the final site selection.

6.4.8 Summary of Design Example

The total land requirement will be the sum of the acreage needed for pretreatment facilities, the actual area to be wetted, buffer zones (if required), access and service roads, and storage ponds. The major elements of each alternative and the total land requirement are summarized in Table 6-16.

6.5 References

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4. Francingues, N.R., Jr. and A.J. Green, Jr. Water Usage and Wastewater Characterization at a Corps of Engineers Recreation Area. U.S. Army Engineer Waterways Experiment Station, Environmental Effects Laboratory. Miscellaneous Paper Y-76-1. January 1976.

5. Minimum Design Standards for Community Sewerage Systems. U.S. Department of Housing and Urban Development. FHA G 4518.1. May 1968.

6. Whiting, D.M. Use of Climatic Data in Estimating Storage Days for Soil Treatment Systems. Environmental Protection Agency, Office of Research and Development. EPA-IAG-D5-F694. 1976.

7. Turner, J.H. Planning for an Irrigation System. American Association for Vocational Instructional Materials. Athens, Ga. 1971.

8. Parmelee, D.M. Personal Communication. May 1977.

TABLE 6-16

SUMMARY OF FEASIBLE LAND TREATMENT SYSTEMS FOR ANGUS, WASHINGTON

		Preapplication treatment		Wetted area		Buffer zones[b]		Storage				
Site	Land treatment system	Level[a]	Area, acres	Application rate, in./wk	Area, acres	Needed	Area, acres	Days of flow	Area, acres	Disinfection	Discharge[c]	Total area required, acres
A	Slow rate, sprinkler	S	3	2.0	38	Yes	8	40	4	Yes	Groundwater	53
	Rapid infiltration	P	2	40.0	1.7	No	0.2	0[d]	.	No	Groundwater	4
B	Slow rate, surface	S[e]	3	2.0	38	No	4	40	4	Yes	Groundwater	49
	Overland flow	P	2	8.0	10	No	1	40	4	No[f]	White River	17
C	Slow rate, sprinkler	S	3	1.5	51	Yes	10	40	4	Yes	Groundwater	68
	Overland flow	P	2	4.0	19	No	2	40	4	No[f]	White River	27

a. S = Secondary; P = primary.

b. May be required in some states. Average requirement based on 20% of wetted area and includes 10% for service roads.

c. If the discharge is to a groundwater, the nitrogen balance of the system should be checked using methods outlined in Chapter 5. Nitrate (NO_3) measured as nitrogen, should not exceed 10 mg/L in this case.

d. Includes extra freeboard for storage within basins.

e. Because of public contact. Under controlled conditions, primary treatment without disinfection would be sufficient.

f. Unless discharge coliform standard cannot be met--then post-treatment disinfection is necessary

1 acre = 0.405 ha
1 in./wk = 2.54 cm/wk

CHAPTER 7

CASE STUDIES

7.1 Introduction

Eleven case studies are presented in this chapter to illustrate the variety of existing land treatment systems. Six of the case studies are slow rate systems; three are rapid infiltration systems; and two are overland flow systems. Locations of the case studies and some system characteristics are presented for comparative purposes in Table 7-1.

TABLE 7-1

SUMMARY OF CASE STUDIES

Location	Avg flow, Mgal/d	Avg annual application rate, ft/yr	Degree of preapplication treatment	Application technique	No. of years in operation
Slow rate					
Pleasanton, California	1.4	8.5	Secondary (plus aerated holding ponds)	Sprinkler (portable pipe)	20
Walla Walla, Washington					
Industrial	2.1	1.7	Aeration	Sprinkler (buried pipe)	5
Municipal	6.8	...	Secondary	Sprinkler and surface (ridge and furrow)	78
Bakersfield, California (existing system)	14.7	6.9	Primary	Surface (border strip and ridge and furrow)	38
San Angelo, Texas	5.8	10.3	Primary	Surface (border strip)	18
Muskegon, Michigan	28.5	6.0	Aerated lagoons	Sprinkler (center pivot)	3
St. Charles, Maryland	0.6	10	Aerated lagoons	Sprinkler (surface pipe)	12
Rapid infiltration					
Phoenix, Arizona	13	364	Secondary	Surface (basin flooding)	3
Lake George, New York	0.7	140	Secondary	Surface (basin flooding)	38
Fort Devens, Massachusetts	1.3	94	Primary	Surface (basin flooding)	35
Overland flow					
Pauls Valley, Oklahoma	0.2	19-45	Raw (screened) and oxidation lagoon	Surface (bubbling orifice) and sprinkler (fixed and rotating nozzles)	2
Paris, Texas (industrial)	4.2	5.2	Raw (degreased and screened)	Sprinkler (buried pipe)	13

1 Mgal/d = 43.8 L/s
1 ft/yr = 0.305 m/yr

In addition to the mix of design flows and climates represented, the ages of the systems vary from several decades (Walla, Walla, Washington; Bakersfield, California; and Lake George, New York) to relatively new (Muskegon, Michigan; Pauls Valley, Oklahoma; and Phoenix, Arizona). The last two systems are principally demonstration projects with process optimization the major objective. Nearly all the cases have attracted some research interests and the ongoing research is discussed separately from the normal operation.

Capital and operating costs are included when reliable data are available. Costs for research projects are not comparable to design and construction costs for normal municipal systems.

7.2 Pleasanton, California

7.2.1 History

Pleasanton, California, with a population of approximately 35 000 is located 40 miles east of San Francisco. Wastewater irrigation has been practiced here since 1911, when the population was 2 000 and only 8 acres (3.2 ha) of land was utilized [1]. The agricultural land apparently has been in continuous use, although historical records are absent. The present system has been in operation since 1957, and consists of the sprinkler irrigation of 1.4 Mgal/d (61 L/s) of secondary effluent on pastureland for the grazing of beef cattle (see Table 7-2) [2]. Only 17 000 of the total population is served by the land treatment system. The remaining population is served by a separate treatment plant. The city is experiencing rapid growth; as a result, plans are underway to provide a regionalized sewer system, which calls for abandonment of the irrigation system in 5 years. It should be noted that abandonment in 5 years was also predicted in 1972 [3].

7.2.2 Project Description and Purpose

There are two primary objectives to be met in this land treatment system: (1) to provide proper management of wastewater, and (2) to produce a high quality forage crop for beef cattle grazing on the wastewater-irrigated fields. This system has proved to be successful in meeting both of these objectives.

The pastureland receiving wastewater is essentially level and sprinklers apply water consecutively to all parts. Soils range from gravelly loam to clay loam and are moderately to slowly permeable. There is a nonirrigated hill of 19 acres (7.7 ha) adjacent to the field area where cattle can be quartered when the fields become somewhat soggy during inclement weather. This is done to protect the soil from excessive compaction by the cattle during wet weather and to prevent the cattle from contracting hoof diseases as a result of the wetness.

TABLE 7-2

DESIGN FACTORS,
PLEASANTON, CALIFORNIA

Type of system	Slow rate
Avg flow, Mgal/d	1.4
Type of wastewater	Primarily domestic; some winery, cheese, and metal wastes
Preapplication treatment	Secondary (plus aerated holding ponds)
Disinfection	Not required
Storage	Not required
Field area, acres	184
Crops	Forage grass
Application technique	Sprinkler, portable
Routine monitoring	Yes
Buffer zones	No
Application cycle	
Time on, h	12-16
Time off, wk	5
Annual application rate, ft	8.5
Weekly application rate, in.	2.2
Avg annual precipitation, in.	18
Avg annual evaporation, in.	71
Annual nitrogen loading, lb/acre	325
Capital costs, $/acre	845
Operation and maintenance costs, ¢/1000 gal	10.4

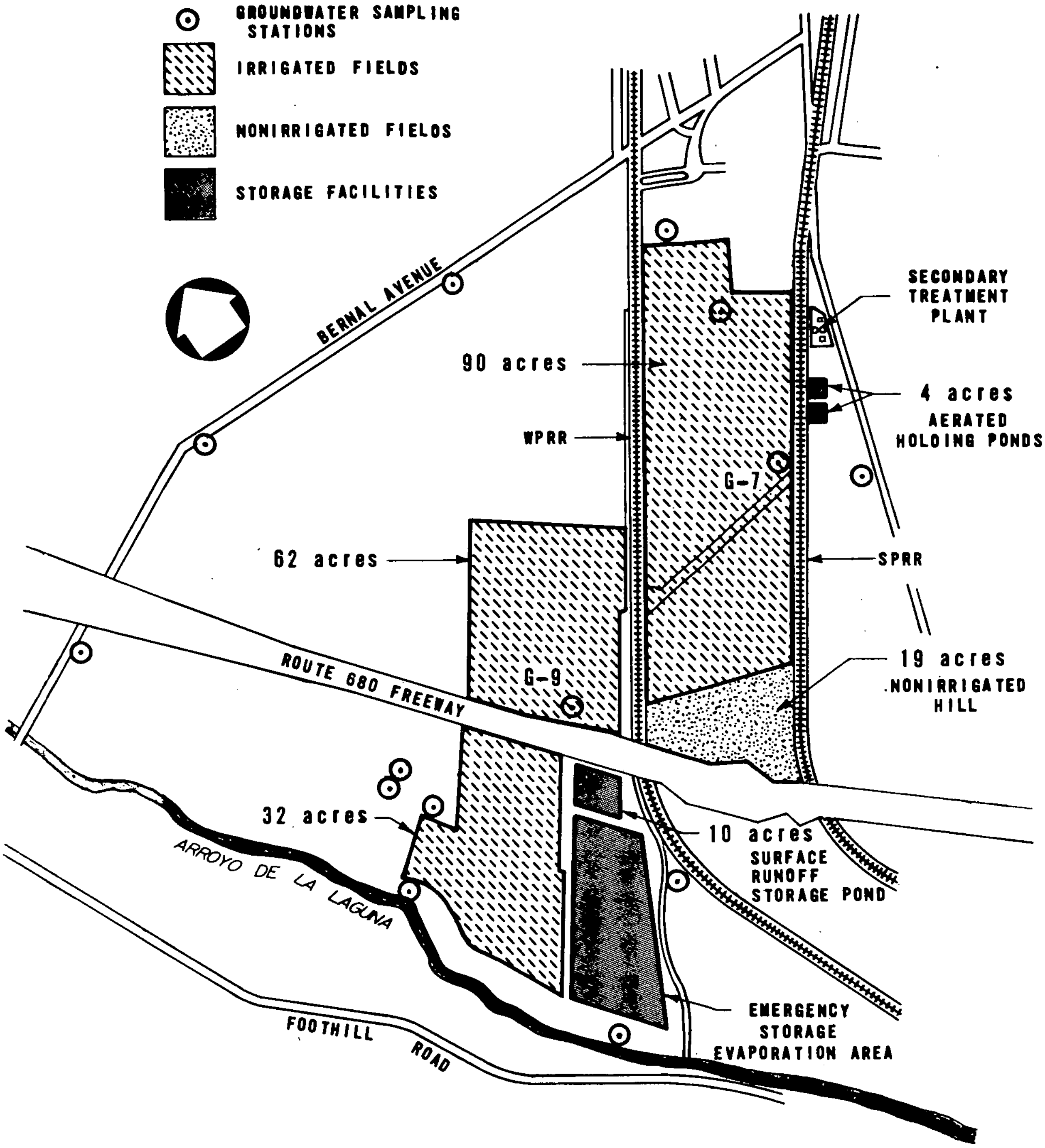
1 Mgal/d = 43.8 L/s
1 acre - 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha
1 $/acre = $2.47/ha
1 ¢/1000 gal = 0.264 ¢/m^3

7.2.3 Design Factors

The land treatment site is schematically depicted in Figure 7-1. Two holding ponds receive unchlorinated effluent from the undersized secondary treatment plant. The two ponds are aerated to prevent septicity prior to application and to provide further biological treatment. The two ponds total 4 acres (1.6 ha) and provide 5 Mgal (18 925 m^3) storage capacity, which equalizes the diurnal flow to the sprinkler irrigation system [1].

FIGURE 7-1

LAND TREATMENT SYSTEM,
CITY OF PLEASANTON



1 acre = 0.405 ha

A 75 hp (56 kW) pump, with an identical standby unit located at the end of the holding ponds, delivers wastewater to the field area via a 10 in. (25 cm) aluminum main line. A portion of the irrigated pastureland and the main line is shown in Figure 7-2. The portable lateral system consists of 30 ft (9 m) sections of 3 in. (7.5 cm) aluminum pipe, each containing a riser with an impact-type sprinkler head as shown in Figure 7-3. Each nozzle delivers 10 to 11 gal/min (0.7 L/s) and has a wetting radius of 30 ft (9 m) [2].

Tailwater and stormwater control is provided by peripheral drainage ditches that discharge into 18 in. (45 cm) and 10 in. (25 cm) steel lines and thence to a runoff collection pond. This 10 acre (4 ha) pond has a 26 Mgal (98 400 m^3) storage capacity and is provided with an overflow to an emergency storage evaporation area. The 40 acre (16 ha) emergency storage area is designed to handle the increased flows from a 50 year storm. Normal runoff water, occurring from about November to March, is recycled within the system by a 100 hp (75 kW) pump.

7.2.4 Operating Characteristics and Performance

The irrigation system is operated 7 days a week, year-round. A normal pumping schedule of 12 to 16 hours per day maintains the holding ponds at a fairly constant elevation. Pumping is by manual control with an automatic shutoff if the ponds drop to a certain level.

The irrigated pastureland supports a herd of 600 beef cattle. The cattle are rotated to fenced plots ahead of irrigation. The laterals containing the sprinklers are moved 60 ft (18.3 m) each day, which results in an application cycle of 5 weeks. The cattle are provided with a separate supply of drinking water at one end of the pasture. The cattle have experienced no ill effects from consumption of the grass; the marketing and sale of the beef occurs in a normal manner.

The pasture grass seed consisted of 44% tall fescue, 32% Italian rye grass, 20% orchard grass, and 4% mixed grasses [2]. Sudan grass is grown on the emergency storage field and is used as supplemental feed. The grasses are cut twice a year with a rotary mower in order to induce better growth and to control weeds such as star thistle. Fertilizers and pesticides have not been used.

Values for various wastewater constituents found in the irrigation water prior to application are presented in Tables 7-3 and 7-4. Although influent total suspended solids to the irrigation system are approximately 25 mg/L, no nozzle-plugging problems have been experienced; the nozzle diameter is 0.44 in. (1.1 cm).

FIGURE 7-2

IRRIGATED PASTURELAND, PLEASANTON, CALIFORNIA

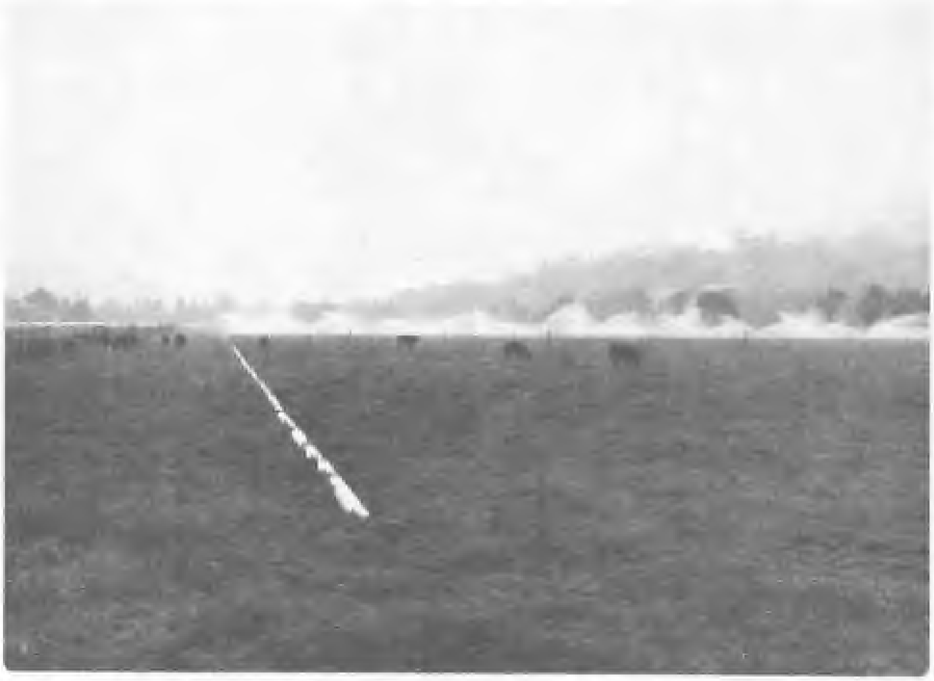


FIGURE 7-3

PORTABLE SPRINKLER SYSTEM, PLEASANTON, CALIFORNIA

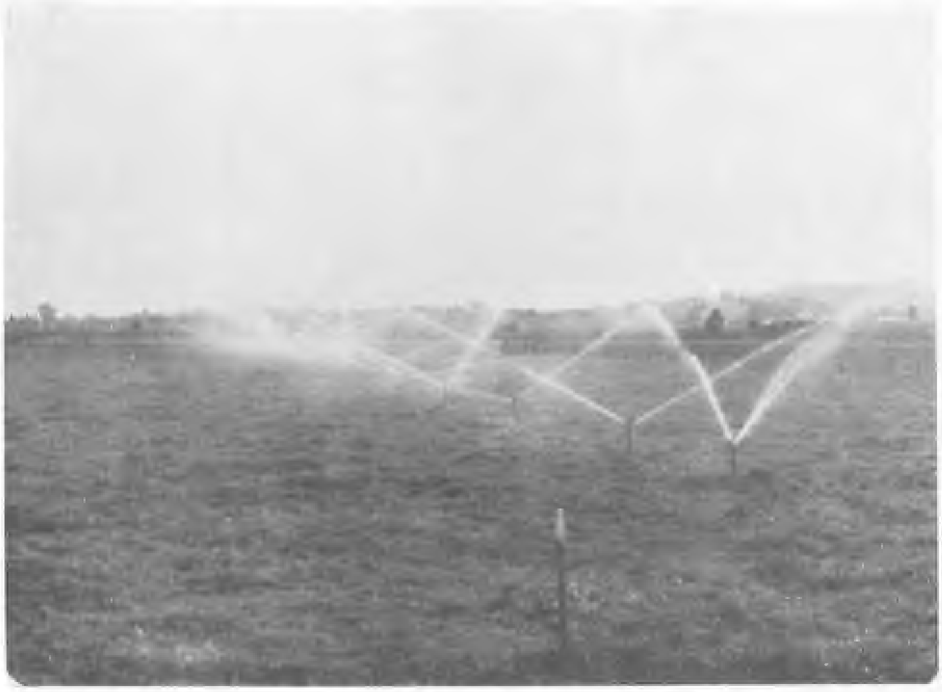


TABLE 7-3

CHARACTERISTICS OF HOLDING POND EFFLUENT AND GROUNDWATER QUALITY, PLEASANTON, CALIFORNIA

Constituent	Concentration, mg/L[a]		
	Holding pond effluent[b]	Groundwater quality[c]	
		G-7	G-9
BOD	22	....	....
COD (low level)	......	30	5
Total suspended solids (TSS)	25	....	....
Total dissolved solids (TDS)	702	980	708
Total organic carbon (TOC)	39	4.3	8.2
Nitrogen			
Organic	3.0	....	....
Ammonium (NH_4-N)	24.6	....	....
Nitrate (NO_3-N)	<0.02	0.13	4.9
Nitrite (NO_2-N)	0.01	0.02	0.01
Total phosphorus	4.8	0.02	0.02
pH	8.4	6.8	6.9
Temperature, °C	......	17.4	16.8
Boron (B)	0.73	0.0008	0.0007
Chloride (Cl)	97	125	111
Fluoride (F)	0.17	0.7	0.1
Sodium (Na^+)	130	150	110
Calcium (Ca^{++})	78	92	93
Magnesium (Mg^{++})	23	90	55
Potassium (K^+)	......	0.8	3.0
Bicarbonate (HCO_3)	520	897	491
Carbonate (CO_3)	3.6	....	....
Hardness (Ca, Mg)	......	600	460
Non-carbonate hardness	......	0	56
Alkalinity as $CaCO_3$	......	736	403
Specific conductance, µmhos/cm	972	1 640	1 190
Sulfate (SO_4)	......	53	120
Silica (SiO_2)	......	16	23
Iron (Fe)	......	0.0006	0.00001
Sodium adsorption ratio (SAR)	3.3	2.7	2.2
Sodium, %	......	35	34
Depth of groundwater below land surface, ft	......	7.5	37.3

a. Unless indicated otherwise.

b. Data for September 1975 [4].

c. Data for November 1975 to August 1976 [5]. See Figure 7-1 for location of wells G-7 and G-9.

1 ft = 0.305 m

Industrial inputs to the system include small flows from a highly acidic but seasonal waste from a winery, a pretreated cheese factory effluent, a pretreated metal waste, and a seasonal loading from the Alameda County Fair. There is no odor in the areas irrigated with wastewater and no odor from the holding ponds. Odors have been a problem at the treatment plant in the past, and as a result the trickling filter and settling tanks have been covered.

TABLE 7-4

TRACE WASTEWATER CONSTITUENTS OF HOLDING POND EFFLUENT, PLEASANTON, CALIFORNIA, SEPTEMBER 1975 [4]

Constituent	Holding pond effluent, mg/L
Arsenic (As)	<0.006
Barium (Ba)	<0.1
Cadmium (Cd)	<0.005
Chromium (Cr^{+6})	<0.005
Cyanide (Cn)	<0.05
Lead (Pb)	0.05
Mercury (Hg)	0.0003
Selenium (Se)	<0.005
MBAS	2.6
Aldrin	<0.00005
Chlordane	<0.00005
DDT	<0.00005
Dieldrin	<0.00005
Endrin	<0.00005
Heptachlor epoxide	<0.00005
Lindane	<0.00005
Methoxychlor	<0.00005
Toxaphene	<0.00005
2,4-D	<0.1
2,4,5-TP	0.001
Carbon chloroform extract (CCE)	2.52
Carbon alcohol extract (CAE)	17.24

7.2.5 Costs

The City of Pleasanton leases most of the farmland from the City of San Francisco which owns it as an underground water reserve. Grazing permits for the 184 acre (74 ha) irrigated pastureland are then allocated to local farmers by auction. This land is sublet at an annual rental fee of $100 to $110/acre ($247 to $272/ha) with a 2 year lease. The city furnishes the irrigation system and the labor to move the portable pipe. The farmer manages the cattle and the pasture grass.

Labor requirements to move the irrigation system involve 3 men at 2-1/2 hours per day, 7 days a week. Maintenance costs for repairs to pumps, pipes, nozzles, and fencing are about $5 000/year. Power consumption consists of about 27 000 kW·h per month for the 100 hp (75 kW) pump and 22 000 kW·h per month for the 75 hp (56 kW) pump, for a total of 49 000 kW;h per month. A breakdown of the approximate annual operating costs for this land treatment system is shown in Table 7-5.

TABLE 7-5

APPROXIMATE OPERATION AND MAINTENANCE COSTS,
LAND TREATMENT SYSTEM,
PLEASANTON, CALIFORNIA [2]

Expenditure	Annual cost
Labor[a]	$22 000
Taxes[b]	21 000
Power[c]	19 000
Material	5 000
Administration	500
Other	5 000
Subtotal	$72 500
Revenue from grazing rights[d]	-19 300
Total	$53 200
Operation and maintenance costs, ¢/1 000 gal[e]	10.4

a. Based on 3 men x 2-1/2 h/d x 7 d/wk x $8/h x 52 wk/yr.

b. Based on $113/acre.

c. Based on 49 000 kW·h/month and PG&E rate schedule A-12.

d. Based on $105/acre·yr x 184 acres.

e. Based on 1.4 Mgal/d avg flow.

1 acre = 0.405 ha
1 Mgal/d = 43.8 L/s

The most recent expansion in 1975-1976 of 20 acres (8 ha) incurred the following capital costs [2]:

Portable aluminum pipe and nozzles	$10 000
Barbed wire fencing to contain cattle	4 500
Drinking water tanks and corral for cattle	2 000
Seed	400
Total	$16 900

This represents a cost of $845/acre ($2 091 ha) not including cost of land.

7.2.6 Monitoring Programs

Since the land treatment operation is conducted over an important underground aquifer, careful monitoring of groundwater quality is mandatory. The California Regional Water Quality Control Board, San Francisco Bay Region, requires regular groundwater sampling at Pleasanton, and to meet this requirement a number of groundwater monitoring wells have been installed. These wells serve to ensure compliance with state regulations and are providing research data to assess overall groundwater impacts in the adjacent area. The location of these wells was shown in Figure 7-1. Groundwater quality data for two representative wells were presented in Table 7-3, covering the period from November 1975 through August 1976.

The Pleasanton land treatment site is located within a mile of a city development of more than 10 000 people. There are essentially no buffer zones; however, the site is totally enclosed by fences to limit public access. A health effects study is being conducted by the Southwest Research Institute. Scientists are performing measurements to determine the extent of aerosol dispersal and are analyzing irrigation water and aerosols for the presence of pathogenic microorganisms and chemicals. The results and conclusions of this study are not available at the time of this report.

7.3 Walla Walla, Washington

7.3.1 History

The use of wastewater as a source for irrigation water began in 1899 when the City of Walla Walla installed its first sanitary sewage collection system and discharged directly to Mill Creek without treatment. Irrigators still withdraw water from the creek for their truck crops. As the population increased and the system expanded, the wastewater became a larger portion of total stream flow, especially during the summer months.

In 1929, the city constructed a 7.5 Mgal/d (328 L/s) secondary treatment plant to treat domestic and industrial flows. In 1953, the industrial (food processing) wastes of about 3 Mgal/d (131 L/s) were separated from the plant and from 1953 to 1962, industrial wastewaters were treated at the source by the food processors. In 1962, industrial wastewaters began to receive treatment in an 8 Mgal/d (350 L/s) separate plant operated by the city. The industrial wastewater was screened, pH adjusted to 7.0, and directly discharged to Mill Creek.

In 1972, the domestic plant was upgraded to provide a higher quality effluent and now has an average treatment capability of 9.12 Mgal/d

(400 L/s) and a maximum hydraulic capacity of 13 Mgal/d (569 L/s). This same year, a sprinkler system for application of industrial wastewater was completed for all industrial effluent not required for stream flow augmentation. Stream flow augmentation is required to maintain a minimum flow of 11.25 ft^3/s (318 L/s) in Mill Creek and 1.77 ft^3/s (50 L/s) in the irrigation ditches. This source of irrigation water becomes essential in the summer, when upstream users divert all of the normal Mill Creek flow. Design factors for the industrial wastewaters (city operated) and municipal wastewaters (privately operated) slow rate systems are presented in Table 7-6.

TABLE 7-6

DESIGN FACTORS,
WALLA WALLA, WASHINGTON

	Industrial[a]	Municipal[b]
Type of system	Slow rate	Slow rate
Avg flow, Mgal/d	2.1	6.8 (municipal effluent to creek in winter)
Type of wastewater	Food processing	Domestic
Preapplication treatment	Aeration	Secondary
Disinfection	No	Yes
Storage	Not required	Not required
Field area, acres	700	940
Crops	Alfalfa	Vegetables
Application technique	Sprinkler (buried pipe)	Sprinkler and surface (ridge and furrow)
Routine monitoring	No	Yes
Buffer zones	No	No
Application cycle, wk		
Time on	1-2	Varies with crop
Time off	6-8	Varies with crop
Annual application rate, ft	1.7	Varies with crop
Weekly application rate, in.	0.7	Varies with crop
Avg annual precipitation, in.	15.5	15.5
Avg annual evaporation, in.	41	41
Capital cost, $/acre	2 500	--
Operation and maintenance costs, ¢/1 000 gal	4.0	4.3

a. City operated system.

b. Irrigation districts, privately operated.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha
1 $/acre = $2.47/ha
1 ¢/1 000 gal = 0.264 ¢/m^3

7.3.2 Project Description and Purpose

The treatment plants for all of the city's industrial and domestic wastewaters are located on an approximately 40 acre (16.3 ha) site 2 miles (3 km) east of the city. Additionally, the city owns approximately 1 000 acres (405 ha) of land 0.6 mile (1 km) north of this area for the sprinkler irrigation of effluent from the industrial treatment plant. Of the 1 000 acres, about 700 acres (285 ha) are presently being used with the remainder being held in reserve for future expansion.

7.3.2.1 Municipal System

Incoming domestic wastewaters are received in an aerated grit chamber, sent to the primary clarifiers, then to the three high-rate trickling filters. Next is intermediate clarification followed by a standard rate trickling filter and two final clarifiers. Sufficient chlorine is injected upstream of the final clarifiers to maintain a 0.1 to 0.5 mg/L residual in the final clarifier effluent. The final clarifiers double as chlorine contact tanks. The effluent is then discharged to a holding pond from which it flows to either the irrigation districts or Mill Creek. The city normally does not apply domestic effluent to its own land.

7.3.2.2 Industrial System

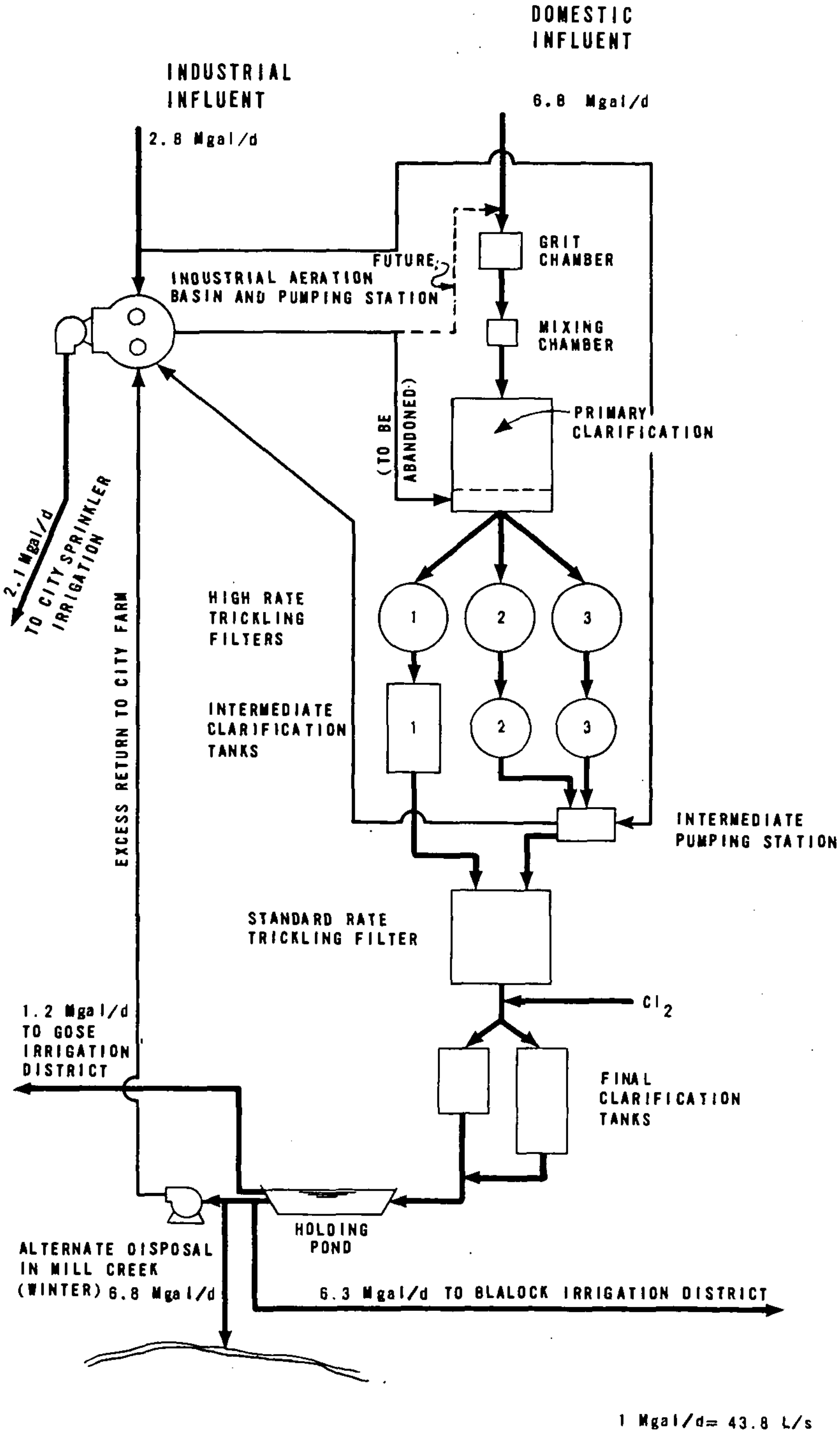
During the canning season (April through November) wastewater from the area's food processors (mostly locally grown vegetables) is pretreated at the packers. All solids above a #20 mesh (0.833 mm diameter) are screened from the waste stream before discharge to a separate collection system. The influent is then received at the plant in an aeration basin, aerated, and pumped to the city's sprinkler irrigation field.

7.3.2.3. Municipal/Industrial Interconnections

To maintain treatment flexibility, there are three operable interconnections between the two normally separated treatment systems. A schematic flow diagram for the municipal and industrial treatment systems is shown in Figure 7-4. During low industrial waste flow periods, when there is insufficient flow to operate the city (industrial) sprinkler system or when makeup water is needed to meet the irrigation commitment, a line from the industrial aeration basin connects directly to the end of the municipal primary clarifier. This has caused some problems with excessive vegetable oils at certain times of the year so this line will be rerouted to ahead of the aerated grit chamber, allowing complete primary treatment of these oils. Another line allows diversion of the raw industrial wastewater directly to the standard rate trickling filter

FIGURE 7-4

SCHEMATIC FLOW DIAGRAM, WASTEWATER TREATMENT PLANT, WALLA WALLA, WASHINGTON



if the industrial flow is extremely low relative to municipal influent. A third line, which is normally not used, allows municipal effluent in the final effluent holding pond to be pumped to the city sprinkler system. A fourth line, diverting raw industrial wastewater to the aerated grit chamber, is in place but inoperable due to leakage problems.

7.3.3 Design Factors

Operation of the city sprinkler system normally occurs from May to November each year for the food processing wastewater. Demand for irrigation water by the districts occurs from May through September so that a part of the industrial flow goes to the city tract with the rest being used for makeup to meet the irrigation demand. The monthly flow patterns for the sprinkler operation are presented in Table 7-7.

TABLE 7-7

AVERAGE DAILY FLOW OF WASTEWATER,
WALLA WALLA, WASHINGTON
Mgal/d

	May	Jun	Jul	Aug	Sep	Oct	Avg
Municipal wastewater[a]	7.1	7.3	7.0	7.0	6.4	6.0	6.8
Industrial wastewater	1.2	3.7	4.0	3.1	2.0	2.9	2.8
Total	8.3	11.0	11.0	10.1	8.4	8.9	9.6
Irrigation district demand	7.5	7.5	7.5	7.5	7.5	7.5	7.5
Net flow to city owned sprinkler irrigation fields	0.8	3.5	3.5	2.6	0.9	1.4	2.1

a. From wastewater treatment plant operations monthly report

1 Mgal/d = 43.8 L/s

The city's 700 acre (285 ha) irrigation site is divided into 8 separate subareas, the largest being 145 acres (69 ha); the smallest 60 acres (24 ha); with an average size of about 87 acres (36 ha).

There is no formal plan for operating the city sprinkler system. Each of the subareas is irrigated for a period of 1 to 2 weeks. The lack of an operational schedule and flow records for each plot means that application rates can only be estimated. The calculated average application rate based on 6 months flow to the field was presented in Table 7-6. Soils are principally well drained silt loams with slopes ranging from 2 to over 20%.

7.3.4 Operating Characteristics and Performance of City's Irrigation System

The industrial effluent for city operated sprinkler application is pumped against a maximum static head of 50 ft (15.2 m). At the farm, it is distributed to 8 separate fields through 187 laterals and 6 500 sprinkler heads. A gage pressure as high as 140 $lb/in.^2$ (96 N/cm^2) is maintained in the main distribution line with pressure at the heads in the 95 to 100 $lb/in.^2$ (67-70 N/cm^2) range. Even with some slopes on the site exceeding 20%, there is evidence of only minor erosion. The city sprinkler system employs two types of heads, the impact nozzle and the large diameter gun. The latter type, shown in Figure 7-5, distributes 325 gal/min (1 260 L/min) at 100 $lb/in.^2$ (70 N/cm^2) with 410 ft (125 m) diameter of coverage.

At the high operating pressures, there is a problem of breakage of joints between the risers and the lines and between the risers and the impact heads. There is also a problem of breakage of the heads and risers by the mower because of lack of visibility of the heads when the crops are highest prior to mowing. The type of mower used and an impact sprinkler are shown in Figure 7-6.

The principal crop being grown on the city's irrigated plot is alfalfa for hay. The farming operation is contracted with a local farmer who is paid on an acreage, bale, or weight basis according to the task being performed. For example, mowing and windrowing is paid for by the acre. The city stores the bales and sells them when markets are strong. Protein quality of the hay is good, averaging 14.5% by weight with a high of 21.0% and a low of 9.4%.

Although the BOD of the industrial effluent could be considered high (average = 965 mg/L), there were no indications of nuisance conditions at the application site nor have complaints been noted from the surrounding residents.

7.3.5 Irrigation of Municipal Effluent by Private Districts

Application rates of the municipal effluent used by the irrigation districts are difficult to estimate as no records are kept. The Blalock irrigation district consists of 840 acres (339 ha) and the Gose irrigation district is approximately 100 acres (40 ha) in size. District farmers withdraw water directly from the ditches or Mill Creek for sprinkler or flood irrigation, depending upon the time of year or the crop. On the average, sprinklers are of the 6 to 7 gal/min (23 to 26 L/min) type covering an area 40 ft by 60 ft (12 m by 18 m). Sprinklers are allowed to run 3 to 6 hours before being rotated to a different parcel. This gives application rates ranging from a potential minimum of 0.7 in./d (1.8 cm/d) to a maximum of 1.7 in./d (4.3 cm/d).

FIGURE 7-5

LARGE DIAMETER SPRINKLER GUN FOR INDUSTRIAL WASTEWATER APPLICATION USED AT WALLA WALLA, WASHINGTON

FIGURE 7-6

ALFALFA HARVESTING EQUIPMENT, SPRINKLER RISER, AND IMPACT HEAD AT CITY IRRIGATION SITE, WALLA WALLA, WASHINGTON

No distinction is made between water that is primarily or partly effluent and well or reservoir waters in terms of crop selection. Local farmers grow whatever has an attractive market price without regard to the water's source. Effluent irrigated crops sell for as high a price as noneffluent grown crops and there has been no case of market discrimination. This was confirmed by a representative of one of the area's food processing plants who purchases large amounts of both effluent and noneffluent irrigated vegetables.

Supplemental nitrogen (350 lb/acre [390 kg/ha] of 48% urea) and phosphorus are added to the wastewater to increase crop yields. Crops grown on domestic effluent irrigated land are (in decreasing order of acreage): onions, carrots, spinach, alfalfa (both for grazing and for harvesting), radishes, and tomatoes. Local farmers have not noted any decrease in yields nor deterioration of croplands over the years of effluent use. Nuisance conditions caused by slime buildup have occurred in the past, but separation of the industrial wastewater in 1972 appears to have solved the problem.

7.3.6 Costs

Cost data are difficult to compare between the municipal and industrial systems. For example, treatment costs for the municipal wastewater are borne by the irrigation districts and neither these costs nor the revenue from the farming are included in the treatment plant accounts. Likewise, much of the time required to maintain the city farm and to administer its agreement with the contracting farmer is not accounted for separately. Neither are records kept on the portion of industrial effluent used for makeup water in meeting the district's contracted irrigation demands. Last, credit for crops sold does not accrue to the cost of operating the farm but is returned to the revenue bond payers (the area's two food processing plants) to help them retire the debt for the sprinkler system construction. The estimated operation costs are summarized in Table 7-8.

The capital cost for construction of the pumping station, transmission lines, distribution system, and control system for the 700 acre (286 ha) sprinkler irrigation field was $1.7 million in 1971. This amounts to about $2 500 per acre ($6 000 per ha) construction cost.

7.3.7 Monitoring Programs

The only continuing monitoring program is carried out by the City of Walla Walla for the treatment plant operating records [6]. A specific soils monitoring program was conducted in 1974 and repeated in 1976. The soils in the city's sprinkler irrigation operation were sampled to determine the effects of irrigation. The adjacent 500 acre (204 ha) city tract is a nonirrigated dry land farming area on which wheat and barley are grown. Soils tests taken in 1974 of the nonirrigated and irrigated parcels provide little indication of any differences in conditions after 2 years of wastewater irrigation.

TABLE 7-8

AVERAGE ESTIMATED OPERATIONS COSTS,
WALLA WALLA, WASHINGTON
¢/1 000 gal

	Municipal system[a]	Industrial system[b]
1973	3.4	...
1975	3.7	7.6
1976[c]	4.3	4.0

a. Treatment costs only.

b. Operates May-November, treatment and distribution costs.

c. To July 1976.

1 ¢/1 000 gal = 0.264 ¢/m^3

7.3.8 Conclusion

The division of the industrial and municipal wastewaters into separate treatment and application streams is unique and offers some real advantages at Walla Walla. This method should be investigated for other areas having problems associated with the high seasonal loads and high BOD of food processing wastes.

7.4 Bakersfield, California

7.4.1 History

Land application of wastewater has been practiced for over 60 years at Bakersfield in the San Joaquin Valley of central California. Beginning in 1912, untreated wastewater was used for crop irrigation. Since 1939, city-owned lands have been continuously utilized for irrigation of forage, fiber, and seed crops with treated municipal wastewater. Primary treatment plants constructed in 1939 (plant No. 1) and 1952 (plant No. 2) service about half of the metropolitan area (population 200 000) and supply the wastewater for the 2 400 acre (960 ha) city farm. The farm is leased to a grower who irrigates year-round with surface distribution methods. Year-round irrigation is possible because of the warm arid climate.

Although the farming operation has been successful in containing all wastewater within the boundaries of the site and producing crop yields consistent with local averages, certain deficiencies have developed, and upgrading and expansion of existing facilities are needed. A summary of principal design factors for both the existing and proposed land treatment systems is presented in Table 7-9.

TABLE 7-9

DESIGN FACTORS, BAKERSFIELD, CALIFORNIA

	Existing system	Proposed system
Type of system	Slow rate	Slow rate
Avg flow, Mgal/d	14.7	19.0
Type of wastewater	Primarily domestic, some poultry processing waste	Primarily domestic, some poultry processing waste
Preapplication treatment	Primary	Aerated lagoons
Disinfection	Not required	Not required
Storage		
Time, d	4	90
Capacity, Mgal	60	1 710
Field area, acres	2 400	4 800
Crops	Forage, fiber, and seed	Forage, fiber, and seed
Application technique	Surface (border strip and ridge and furrow)	Surface (border strip and ridge and furrow
Routine monitoring	No	Yes
Buffer zones	No	No
Application cycle, d		
Time on	1-2	0.5-1
Time off	7-15	10-15
Annual application rate, ft	6.9	4.5
Maximum weekly application rate, in.	4	4
Avg annual precipitation, in.	6.4	6.4
Avg annual evaporation, in.	60	60
Annual nitrogen loading, lb/acre	466	280
Capital cost of proposed system, $/acre[a]	--	2 960

a. Not including preapplication treatment or storage.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha
1 $/acre = $2.47/ha

7.4.2 Existing System Characteristics, Design Factors, and Performance

The existing land treatment system, depicted in Figure 7-7, consists of a network of ditches and equalizing reservoirs supplying the fields with wastewater for border strip and ridge and furrow methods of irrigation. Although the topography is very flat, the drainage is from north to south with sump pumps along the southern end to return tailwater to storage ponds (see Figure 7-7). Soils range from fine sandy loam to clay loam. The soils are generally alkaline and poorly drained with dense clay lenses at depths ranging from 10 to 15 ft (3 to 4.5 m) below the surface. This clay barrier produces perched water in areas where it is continuous and reduced percolation in areas where it is not.

Two permanent groundwater aquifers exist at approximate depths of 100 to 200 ft (30 to 60 m) and at 300 ft (90 m). They are separated by a clay barrier, and the confined lower aquifer is used for water supply. The deep wells on the farm, as shown in Figure 7-7, produce water for supplemental irrigation water. The quality in this region, however, is inadequate for potable uses as a result of naturally occurring high total dissolved solids and nitrates.

The wastewater is primarily domestic in nature, with only a few poultry-processing plants discharging high-BOD wastes to plant No. 1. The characteristics of effluents from plant No. 1 (3.8 Mgal/d [166 L/s]) and plant No. 2 (10.9 Mgal/d [477 L/s]) have been combined and a typical blend of constituents found in the irrigation water is given in Table 7-10.

The quality of the combined primary effluent is quite suitable for irrigation. The sodium adsorption ratio is relatively high at 7.5; however, it is not critical. The total dissolved solids concentration is not a problem for any of the crops grown.

Liquid and nitrogen loading rates, and nitrogen requirements for the principal crops grown on the farm are shown in Table 7-11. As can be seen, the nitrogen applied meets the nitrogen uptake of all crops. For cotton, the nitrogen loading is more than twice that which can be utilized. Applying excess nitrogen to cotton promotes excess vegetative growth at the expense of fruitive growth, resulting in decreased yields. Yields for all other crops are approximately equal to, and in some cases higher than, the countywide averages. Crop yields resulting from irrigation with primary effluent and the economic return per acre are presented in Table 7-12.

FIGURE 7-7

EXISTING WASTEWATER IRRIGATION SYSTEM,
BAKERSFIELD, CALIFORNIA

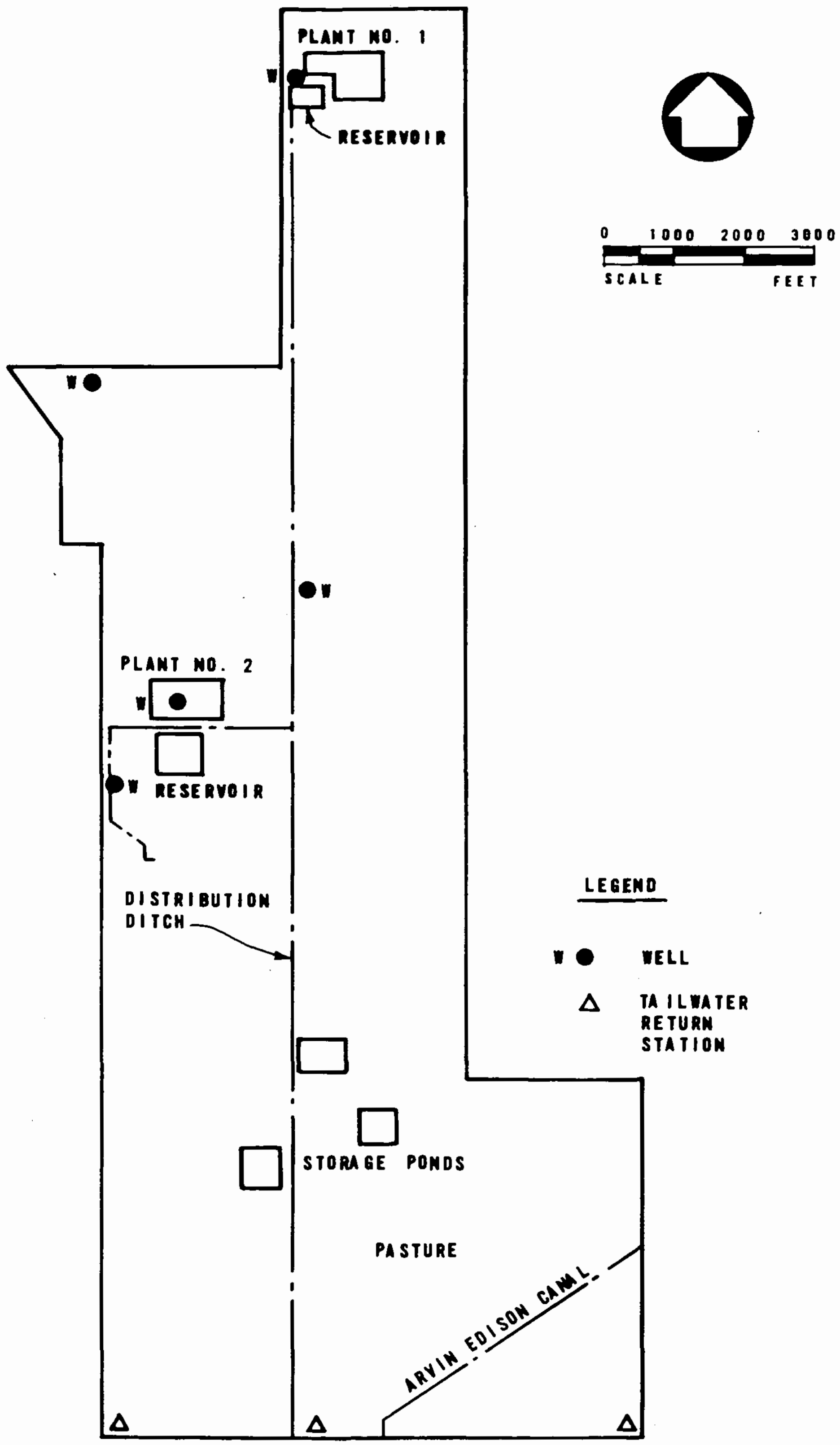


TABLE 7-10

COMPOSITE WASTEWATER CHARACTERISTICS FOR CITY PLANTS NOS. 1 AND 2, BAKERSFIELD, CALIFORNIA[a]

BOD, mg/L[b]	150
Suspended solids, mg/L	48
pH, units	7.0
EC, mmhos/cm	0.88
TDS, mg/L	477
SAR	4.1
SAR(adj)	7.5
Calcium, meq/L	2.30
Magnesium, meq/L	0.41
Sodium, meq/L	4.74
Potassium, mg/L	26
Carbonate, meq/L	0
Bicarbonate, meq/L	3.57
Chloride, meq/L	3.01
Sulfate, meq/L	1.54
Boron, mg/L	0.38
Cadmium, mg/L	<0.01
Total nitrogen as N, mg/L	20-25
Phosphorus as P, mg/L[b]	6.2

a. Based on 1976 tests, except as noted.

b. Based on 1973 tests.

TABLE 7-11

LOADING RATES IN 1973 AND TYPICAL NITROGEN UPTAKE REQUIREMENTS, BAKERSFIELD LAND TREATMENT SYSTEM [7]

Crop	Liquid loading rate, ft/yr	Nitrogen loading rate, lb/acre·yr	Typical nitrogen uptake, lb/acre·yr
Alfalfa	4.9	371	360-480
Barley	1.8	139	75
Corn	3.3	252	150
Cotton	3.7	277	100
Pasture grass	4.9	371	150-250

1 ft = 0.305 m
1 lb/acre = 1.12 kg/ha

TABLE 7-12

EXISTING CROP YIELDS AND ECONOMIC RETURN, BAKERSFIELD, LAND TREATMENT SYSTEM

Crop	Yield, lb/acre	Typical price, $/lb[a]	Economic return, $/acre
Alfalfa	16 000	0.025	400
Barley	3 000-5 000	0.045	135-225
Corn	36 000-60 000	0.0075	270-450
Cotton	600-800	0.35	210-280

a. Based on 1973 prices.

1 lb/acre = 1.12 kg/ha
1 $/lb = $2.2/kg
1 $/acre = $2.47/ha

Management of water has become a problem due to lack of storage and increasing flows. Ponding of excess water has occurred on some areas of the pastureland in winter. Although flies and mosquitos are attracted to the stagnant water, no diseases have been traced to effluent use. The equalizing reservoirs and the storage pond for tailwater are periodically sprayed to control mosquito propagation.

Public health regulations for irrigation of fodder, fiber, and seed crops are such that the quality of reclaimed water shall not be less than that of primary effluent. No disinfection of the effluent is required, and none is provided at the two treatment plants. Normally, both corn and barley are green chopped (not harvested for grain) for cattle fodder. Dairy cows are not allowed to graze pastures irrigated with nondisinfected effluent so they are fed with green chop and hay. Beef cattle are allowed to both graze pastures and be fed on the green chop and hay.

7.4.3 Proposed System Characteristics and Design Factors

Although primary effluent is suitable according to the California Department of Health, the Central Valley Regional Water Quality Control Board has set limits of 40 mg/L on BOD and suspended solids prior to forage crop irrigation. Their rationale is that such an effluent can be stored without causing a nuisance and will reduce the potential for odors in system management.

The proposed system consists of an upgrading of the existing preapplication facilities and inclusion of a concrete pipe distribution system for continued surface irrigation of crops. City plant No. 1 will be abandoned and its flow redirected to city plant No. 2, where new primary clarifiers and aerated lagoons will be constructed. Chlorination will not be provided since surface application to forage, fiber, and seed crops does not require disinfection.

The 4 800 acre (1 944 ha) land area of the proposed system will be twice that of the existing system because the principal objective of the proposed system is crop production rather than land treatment. Thus, double cropping with corn and barley is proposed and this combination requires less water than is presently applied. A schematic of the portion of the system located within the limits of the existing city farm and an additional 320 acre (129 ha) parcel northeast of the farm is presented in Figure 7-8. The remainder of the system (not shown in the figure) is located to the south, including 960 acres (389 ha) of undeveloped land which will be reclaimed for irrigation.

7.4.4 Proposed System Operation

Flexibility of operation will be provided by storage reservoirs, which will hold flows during periods of low irrigation demand. In addition, automatically operated tailwater return stations will control runoff from irrigated fields. Outlets from the distribution laterals will consist of orchard-type valves which are adaptable to gated surface pipe, open ditches with siphon pipe, or direct flooding of border strips. Telemetered alarms will continuously scan the operation of the system to alert the operator of malfunctions at any of the pumping stations.

7.4.5 Costs

City revenues from the lease of the existing farm amount to about 20% of the operating and maintenance costs for the two treatment plants [7]. Detailed operating and maintenance costs for the existing farm were not available.

The estimated construction costs for the proposed system on the basis of summer 1977 construction startup are summarized in Table 7-13. These costs do not include land acquisition costs or engineering, administration, and legal expenses. The City of Bakersfield will lease the property to the highest responsible bidder for management of the system, and the city will be responsible only for maintenance of the main pipeline and all pumping equipment.

FIGURE 7-8

PROPOSED WASTEWATER IRRIGATION SYSTEM,
BAKERSFIELD, CALIFORNIA

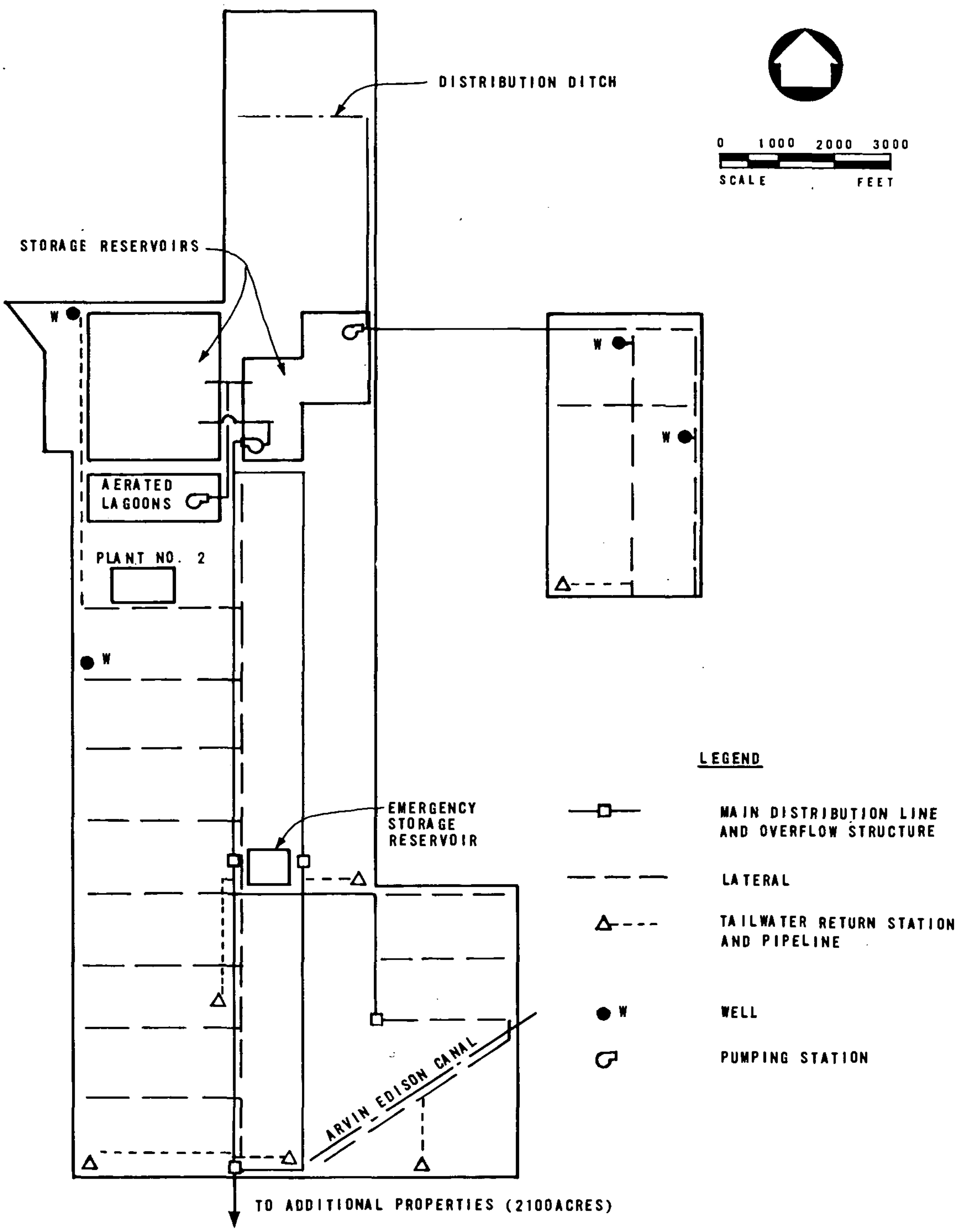


TABLE 7-13

ESTIMATED CONSTRUCTION COSTS,
PROPOSED WASTEWATER IRRIGATION SYSTEM,
BAKERSFIELD, CALIFORNIA[a]

Land reclamation and site development	$ 1 126 000
Storage and equalizing reservoirs	5 385 000
Distribution pipelines	4 697 000
Distribution pumping stations	621 000
Tailwater return systems	513 000
Bonds and insurance	154 000
Total	$12 496 000
Construction costs, $/gal of capacity[b]	0.66

a. Based on summer 1977 construction startup.

b. Based on 19 Mgal/d average flow.

1 $/gal = $3.785/L
1 Mgal/d = 43.8 L/s

7.5 San Angelo, Texas

7.5.1 History

San Angelo, a city of about 65 000 in west central Texas, has treated its wastewater by primary sedimentation followed by land application since 1928. The system was operated at one site for the first 30 years and has been operated at the present site for 18 years. Pressure from development around the first site led to its abandonment in 1958. The present slow rate system consists of 630 acres (255 ha) of city-owned pasture and cropland irrigated by the border strip method (see Table 7-14). Preapplication treatment will soon be upgraded from primary to secondary to meet state requirements for wastewater irrigation of areas accessible to the public.

7.5.2 Project Description

Wastewater is currently given primary treatment prior to land application. The effluent can be used directly to irrigate the pastureland, shown in Figure 7-9, or it can be directed through four holding ponds. The detention time in the ponds at an average flow of 5.8 Mgal/d (254 L/s) is about 30 days. The 330 acres (134 ha) of pasture receive somewhat more effluent annually than the 300 acres (121 ha) of cropland, which is rotated between oats, rye, and grain sorghum.

TABLE 7-14

DESIGN FACTORS,
SAN ANGELO, TEXAS

Type of system	Slow rate
Avg flow, Mgal/d	5.8
Type of wastewater	Domestic and industrial
Preapplication treatment	Primary
Disinfection	No
Storage	
Capacity, Mgal	174
Time, d	30
Field area, acres	630
Crops	Coastal Bermuda grass, fescue, oats, rye, and grain sorghum
Application technique	Surface (border strip)
Routine monitoring	Yes
Buffer zones	No
Application cycle, d	
Time on	1
Time off	10-14
Annual application rate, ft	10.3
Avg weekly application rate, in.	2.4
Avg annual precipitation, in.	18.6
Avg annual evaporation, in.	60
Annual nitrogen loading, lb/acre	800
Operation and maintenance costs, ¢/1 000 gal	0.9

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.13 kg/ha
1 ¢/1 000 gal = 0.264 ¢/m^3

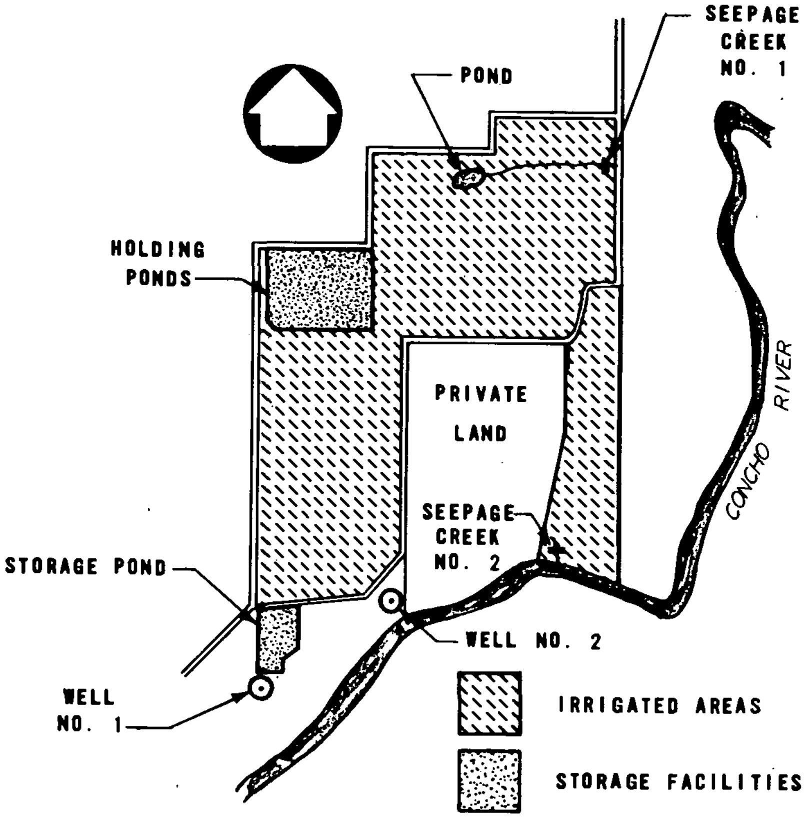
7.5.3 Design Features

Distribution consists of underground pipelines and outlet valves at the head end of the borders. The pasture is irrigated year-round with the storage ponds providing additional treatment as well as detention to allow crop rotation. Effluent is pumped from the primary treatment plant to the storage ponds and flows by gravity to the fields.

As can be seen in Table 7-14, the evaporation exceeds precipitation by an average of 41.4 in./yr (1.1 m/yr). Thus, the 123.6 in./yr (3.1 m/yr) wastewater application results in an excess of 82.2 in./yr (2.1 m/yr). The percolating water emerges as groundwater seeps at several drainage points on the farm and flows into the Concho River.

FIGURE 7-9

SLOW RATE LAND TREATMENT SYSTEM
AT SAN ANGELO, TEXAS



7.5.4 Operating Characteristics and Treatment Performance

The pastureland is planted in coastal Bermuda grass that is both grazed by beef cattle and harvested as hay. Grazing rights are sold to cattlemen and the hay, shown in Figure 7-10, is sold to the public by the bale. The oats, rye, and sorghum are used for cattle feed.

The wastewater is applied by the border strip method as shown in Figure 7-11. The borders vary in width and follow the slope of the land. The principal soils are Angelo and Rio Concho clay loams.

The treatment performance can be estimated by comparing the quality of the applied wastewater to the quality of groundwater that emerges out of

FIGURE 7-10

COASTAL BERMUDA GRASS HAY AT SAN ANGELO, TEXAS

FIGURE 7-11

BORDER STRIP IRRIGATION OF PASTURE AT SAN ANGELO, TEXAS

seepage creek No. 1. The treatment performance data presented in Table 7-15 are for October 1973 [8]. The apparent nitrogen removal of 52% is for an area currently in pasture that receives over 800 lb/acre·yr (900 kg/ha·yr). The crops grown in that area in 1973 are unknown.

TABLE 7-15

TREATMENT PERFORMANCE,
SAN ANGELO, TEXAS [8]
mg/L

Constituent	Pond effluent	Seepage creek No. 1
BOD	54.2	1.0
Ammonia nitrogen	28.0	0.2
Nitrate nitrogen	0.8	13.0
Total nitrogen	28.8	13.7
Total phosphorus	5.9	0.09
Total dissolved solids	1 704	1 900

The high total dissolved solids value apparently does not adversely affect crop growth. The pasture that is grazed supports 10 head of cattle per acre (25 head/ha) which is an order of magnitude greater than comparable densities for conventional irrigated pasture in central Texas.

7.5.5 Costs

Capital costs for the construction of the system in 1955 and purchase of the land are not available. In 1972, the value of the land was estimated to be \$500/acre (\$1 250/ha) [3]. The new activated sludge plant, which will have a capacity of 8.5 Mgal/d (372 L/s), will cost \$4 million (April 1977). This plant will be capable of supplying effluent as irrigation water to nearby farmers. The present land treatment system may be expanded when the city can purchase additional land in the area.

Operations require three farm employees and a manager at a budget of about \$60 000 to \$70 000/yr. Grazing rights are sold at \$5.50/month for each head of cattle. The baled hay is sold at \$1.50 to \$2.00 per bale. In all, the revenue from the farm amounts to \$80 000 to \$90 000/yr for a net profit of around \$20 000/yr. Revenues amount to 3.8¢/1 000 gal (1.0¢/m^3).

7.5.6 Monitoring

Normal monitoring includes periodic analyses of groundwater in several wells. In October 1973, an intensive monitoring survey was conducted to determine the effects of the land treatment system on the Concho River [8]. The findings were that while seepage from the system was significantly increasing the flow of the river, it was having a negligible effect on the water quality. A sample groundwater analysis from the normal monitoring program is presented in Table 7-16.

TABLE 7-16

SAMPLE OF GROUNDWATER QUALITY, SAN ANGELO, TEXAS [9, 10] mg/L

Constituent	Well No. 1	Well No. 2
Total dissolved solids	1,659	1,628
Total alkalinity as $CaCO_3$	352	394
Total hardness as $CaCO_3$	1,080	676
pH, units	7.3	7.3
Calcium	192	162
Magnesium	146	66
Sodium	130	265
Sulfate	140	120
Chloride	596	500
Bicarbonate	429	481
Iron	2.7	0.1
Phosphorus	0.015	0.025
Ammonia nitrogen	0.0	0.0
Nitrate nitrogen	9.0	22.3
Total nitrogen	9.1	22.4

7.5.7 Long-Term Effects Research

Research on the chemical and microbiological effects of 18 years of operation is being conducted by researchers at Texas A & M University. The 2 year effort is expected to be finished in 1977. Sampling will include soil, plant tissue, wastewater, groundwater, and water emerging as seeps. The heavy metals, nutrients, and organics will be measured in the water and soil samples. Crops within specially fenced areas will be checked for yields and tested for nutrients and accumulation of metals.

7.6 Muskegon, Michigan

7.6.1 History and Objectives

The need for an alternative wastewater management program for the Muskegon County area became apparent in the late 1960s because of deterioration in the water quality of local surface waters. Fourteen municipalities and five major industries were required to achieve an 80% phosphorus reduction and produce effluent that would not result in the degradation of the water quality of Lake Michigan.

Areawide solutions were explored and the most cost-effective solution was to divert all the wastewater discharges from surface waters and make use of undeveloped land as a major component of an areawide treatment system. The decision to undertake such a plan was based in part on economics and in part on a commitment to recycle nutrients as resources rather than discharge them to the environment in a nonbeneficial manner. Construction of the facilities commenced in 1972 and operation was begun in stages starting in May 1974. The first full year of operation was 1975.

7.6.2 Project Description and Design Factors

The Muskegon County Wastewater Project consists of two independent systems: the Muskegon Project and the smaller Whitehall Project. Both systems make use of the slow rate process of land treatment. Because of the much larger size and quantity of information available for the Muskegon Project, this section will be limited to a discussion of that system. A summary of the principal design factors for the Muskegon Project is presented in Table 7-17.

Industrial wastewaters discharged to the system constitute over 60% of the present flow. The largest single discharger, S.D. Warren Company, a Kraft papermill, contributes approximately 15 Mgal/d (0.7 m^3/s) [11].

The treatment system consists of biological treatment in aerated lagoons followed by sprinkler irrigation of land on which corn is presently grown. While there are many interesting features of the system, its uniqueness lies primarily in the size of the facility. With a design capacity of 42 Mgal/d (1.8 m^3/s) and over 5 000 acres (2 025 ha) of land under irrigation, it is the largest operating facility in the United States designed specifically for land treatment of wastewater. Other features include the low overall operation costs. During 1976, crop revenues offset 60% of the total operating costs of the system.

TABLE 7-17

DESIGN FACTORS, MUSKEGON PROJECT, MUSKEGON, MICHIGAN

Type of system	Slow rate
Avg flow, Mgal/d	28.5
Type of wastewater	Domestic and industrial (papermill)
Preapplication treatment	Aerated lagoons
Disinfection	As required
Storage	
Capacity, Mgal	5 323
Time, d	187
Field area, acres	5 350
Crops	Corn and rye grass
Application technique	Sprinkler (center pivot)
Routine monitoring	Yes
Buffer zones	Yes
Application cycle	
Time on	Varies
Time off	Varies
Annual application rate, ft	Varies; 1-9, avg 6
Avg weekly application rate, in.	3.0
Avg annual precipitation, in.	32
Avg annual evaporation, in.	30
Annual nitrogen loading, lb/acre	130
Capital costs, \$/gal of capacity[a]	1.01
Operation and maintenance costs, ¢/1 000 gal	12.5

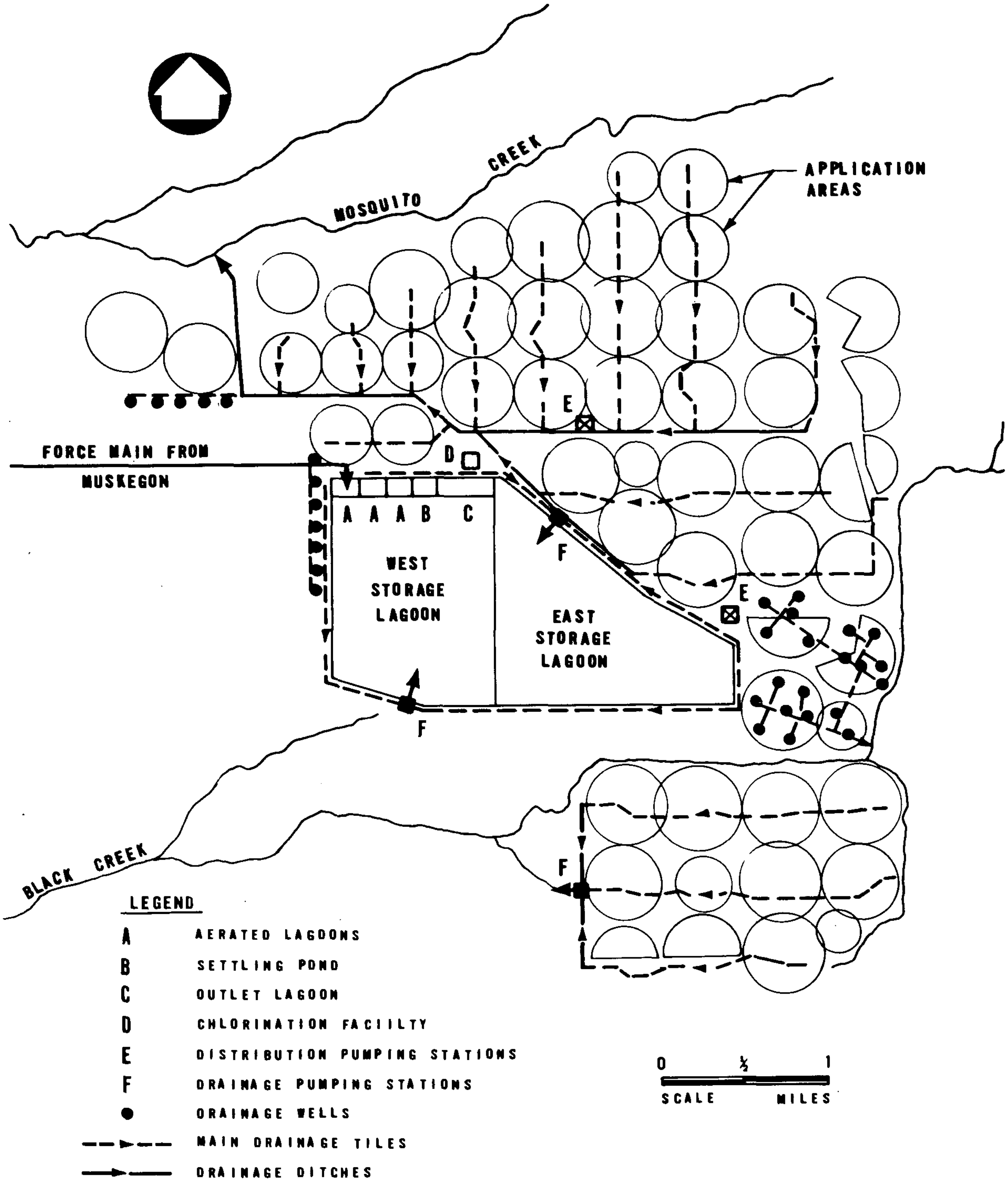
a. Includes transmission, preapplication treatment, storage, distribution system, and underdrainage.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.13 kg/ha
1 \$/gal = 3.785/L
1 ¢/1 000 gal = 0.264 ¢/m^3

A plan of the facilities is shown in Figure 7-12. Incoming wastewater first enters one of three 8 acre (3.2 ha) aerated lagoons, which may be operated in parallel or series. From the treatment cells, wastewater enters the two 850 acre (344 ha) storage lagoons shown. A separate settling pond is also provided which can serve as a bypass to the storage lagoons. During the irrigation season (April through November), water for irrigation is drawn from either the storage lagoons or from the settling pond into a 14 acre (5.6 ha) outlet lagoon. The treated effluent released from the outlet lagoon can be chlorinated in a mixing chamber prior to delivery via open channels to the two main distribution

FIGURE 7-12

MUSKEGON PROJECT LAND TREATMENT SITE PLAN



pumping stations. These pumping stations deliver the wastewater through a series of buried pipes to the irrigation equipment. Because of bacteria die-off during the long storage period, chlorination is not required at all times to meet discharge requirements.

Wastewater is applied to the land by 54 center pivot irrigation machines which utilize low pressure nozzles and roll on pneumatic tires (see Figure 7-13). Vertical turbine pumps at two main pumping stations discharge to an asbestos-cement pipeline distribution network. Major design data for the distribution system are presented in Table 7-18.

Soil types at the application site include sandy soils with infiltration rates from 5 to more than 10 in./h (12.7 to 25.4 cm/h), loamy soils with rates from 2.5 to 10 in./hr (6.4 to 25.4 cm/h), and clay soils with rates ranging from 0.02 to 2.5 in./h (0.05 to 6.4 cm/h). The majority of the soils, however, are sands and sandy loams. The maximum design application rate is 4 in./wk (10 cm/wk), which includes an allowance for 0.74 in./wk (1.9 cm/wk) of precipitation.

A combination of drainage tiles, drainage wells, and natural drainage collects the subsurface water and discharges it to adjacent receiving surface waters. The majority of the site is underlain with drainage tiles at approximately 500 ft (153 m) intervals and from 5 to 8 ft (1.5 to 2.4 m) deep. The laterals, constructed of perforated

FIGURE 7-13

CENTER PIVOT BOOM WITH LOW PRESSURE NOZZLE,
MUSKEGON PROJECT

polyethylene filtered by fiberglass socks, conduct the water to main concrete drainage pipes. The concrete pipes carry the water to open ditches which in turn discharge to two receiving streams. Drainage tiles were largely installed using a continuous plow machine as shown in Figure 7-14.

TABLE 7-18

DISTRIBUTION SYSTEM DATA [12]
MUSKEGON PROJECT

Pumping	
No. of vertical turbine pumps	17
Peak capacity, Mgal/d	91.7
Piping size range, in.	8-36
Center pivot irrigation rigs	
No. of rigs	54
Radius, ft	700-1 400
Coverage range, acres	35-141
Operating pressure, $lb/in.^2$	35-84
Nozzle pressure, $lb/in.^2$	3-10
Application rate (continuous operation)	
in./h	0.0239
in./wk	4.0
Application season, months	8

1 Mgal/d = 43.8 L/s
1 in. = 2.54 cm
1 ft = 0.305 m
1 acre = 0.405 ha
1 $lb/in.^2$ = 0.69 N/cm^2

7.6.3 Operating Characteristics and Performance

Irrigation with wastewater at Muskegon commenced in May 1974, and numerous temporary startup problems were encountered. Most of the problems, such as dike damage, breaks in irrigation pressure pipes, and electrical cable failures, have been resolved. A persistent, significant odor problem occurred at the treatment site and was attributed to the high volume of papermill waste. The inlet structure has been modified to reduce the release of odor.

An operational problem was the plugging of the irrigation rig nozzles with a mixture of sand and weeds, which are blown into the storage lagoons and main irrigation ditches. During the first two irrigation seasons, ten full-time "nozzle cleaners" were hired in an attempt to minimize plugged nozzles. Even with this effort, the degree of uniform water application was not acceptable. To alleviate this problem, settling basins and screening systems have been installed ahead of both irrigation pumping stations.

FIGURE 7-14

INSTALLATION OF DRAINAGE TILES,
MUSKEGON PROJECT

Another operational problem during the first season was downtime due to the irrigation rigs becoming stuck in soft, wet soil in one area. This problem has been greatly alleviated by increasing tire size from 11 by 24 in. to 14.9 by 24 in. (28 by 60 cm to 38 by 60 cm). To further alleviate the problems of stuck rigs, machine speed has been doubled [12].

In 1975, 4 700 acres (1 900 ha) of the 5 400 acres (2 182 ha) irrigated was planted in corn and irrigated with up to 4 in./wk (10 cm/wk) of wastewater. The remaining 700 acres (283 ha) was fallow or in rye grass. Total wastewater applied ranged from zero to over 100 in./yr (254 cm/yr) per field, with the majority of the fields receiving from 50 to 75 in./yr (127 to 190 cm/yr) [13]. Representative yields of corn grain from various fields at the Muskegon land treatment site for the 1975 season are presented in Table 7-19.

TABLE 7-19

REPRESENTATIVE YIELDS OF CORN GRAIN, FOR VARIOUS SOIL TYPES, MUSKEGON LAND TREATMENT SITE, 1975 [13]

Field soil type	Wastewater application, in./yr	Supplemental nitrogen fertilizer, lb/acre·yr	Corn grain yield, bu/acre·yr
Roscommon sand	57	65	90
Rubicon sand	106	63	83
AuGres sand	59	70	71
Roscommon sand	69	40	69
Granby loamy sand	14	27	61
Rubicon sand	93	44	53
AuGres sand	14	10	36
Roscommon sand	14	0	31
Project average	54	44	60

1 in./yr = 2.54 cm/yr
1 lb/acre·yr = 1.12 kg/ha·yr
1 bu/acre·yr = 2.47 bu/ha·yr

The wastewater provides an adequate amount of phosphorus and potassium for the corn crop [13]. However, the low levels of nitrogen in the wastewater would not be adequate without supplemental additions. In the sandy soil, there is little organic nitrogen and even less in a soluble form usable by plants, as sandy soils do not retain much nitrogen. Therefore, during the 2 months of the 6 month irrigation period in which the corn is actively growing, it is necessary to inject nitrogen fertilizer into the wastewater on a daily basis. It is important that the application rate of the soluble nitrogen be adjusted so that the corn plants absorb and use all of the available nutrients as fast as their metabolism permits. From 0 to 89 lb/acre·yr (0 to 100 kg/ha·yr) of nitrogen fertilizer was added to the different irrigated fields, depending upon the amount of wastewater applied and crop requirement needs.

Corn planted in 1976 yielded an average of 81 bu/acre·yr (200 bu/ha·yr), significantly greater than the 45 to 50 bu/acre·yr (111 to 123 bu/ha·yr) average corn grain yield on operating farmland in Muskegon County. This is quite remarkable in light of the fact that most soils at the treatment site are very poor and that wastewater renovation is the primary purpose of the system. The agricultural productivity of the Muskegon land treatment system has steadily increased over its first 3 years of existence, as shown in Table 7-20. Sale of the corn has proceeded with the grain commanding full market value.

TABLE 7-20

INCREASED AGRICULTURAL PRODUCTIVITY, MUSKEGON LAND TREATMENT SITE [13]

	1974	1975	1976
Corn yield, bu/acre·yr			
Land treatment site	28	60	81
Muskegon County average	55	65	45-50
Gross crop revenue, $ millions	0.35	0.7	1.0[a]

a. Estimated.

1 bu/acre·yr = 2.47 bu/ha·yr

The entire Muskegon wastewater treatment operation is being handled by 40 full-time people and an additional part-time labor force of up to 10 workers. A part of this work activity is associated with the Muskegon EPA Research and Development Grant. The normal staffing of the treatment operation during the day shift is 2 people on the northern irrigation rigs and 2 people on the southern irrigation rigs with another 2 people providing maintenance as needed [11]. The other 2 shifts are staffed with 1 person per shift.

The average treatment results for 1975 are presented in Table 7-21. BOD, suspended solids, and phosphorus levels are well below the discharge permit requirements, which are 4 mg/L for BOD, 10 mg/L for suspended solids, and 0.5 mg/L for phosphorus.

7.6.4 Costs

The construction cost for the Muskegon wastewater treatment system was $42.7 million, of which $12.0 million was for collection (force mains and sewer lines) and transmission (pumping and lift stations) [13]. The net operating costs for the total wastewater treatment system (Muskegon and Whitehall sites) incurred during the 1975 season was $1 232 000. Gross operating costs by system component and revenues gained are presented in Table 7-22.

Operating experience based on observations of storage lagoon treatment performance has shown that the actual biological treatment system was overdesigned. It has been demonstrated that proper treatment can be obtained by running a smaller percentage of the aerators [12]. This has resulted in reduced operating costs for aeration. As additional cost-effectiveness measures of this nature become apparent, further reduction in net operating costs can be expected.

TABLE 7-21

SUMMARY OF TREATMENT PERFORMANCE, 1975 AVERAGE RESULTS, MUSKEGON PROJECT [11][a]

Parameter	Influent	Average storage lagoon effluent	Drain tiles	Mosquito Creek
BOD	205	13	1.2	3.3
pH, units	7.3	7.8	......	7.5
Specific conductance, μmhos	1 049	825	599	574
Total solids	1 093	691	......	466
Suspended solids	249	20	......	7
COD	545	118	......	33
TOC	107	38	11.6	15
Ammonia-N	6.1	2.4	0.29	0.6
Total Kjeldahl nitrogen	8.2	4.5	......	......
Nitrate-N	Trace	1.1	2.2	1.9
Total P	2.4	1.4	......	0.05
Chloride	182	154	60	78
Sodium	166	144	42	66
Calcium	73	58	72	61
Magnesium	14	16	23	18
Potassium	11	9	2.6	4
Iron	0.8	1.0	7.7	1
Zinc	0.6	0.11	0.01	0.07
Manganese	0.28	0.16	0.20	0.11
Total coliforms, colonies/100 mL	....	$100\text{-}1.2 \times 10^8$	<1-170	$<1\text{-}9.6 \times 10^4$
Fecal coliforms, colonies/100 mL	....	$4\text{-}1.2 \times 10^6$	<1-32	$<1\text{-}4.8 \times 10^3$
Fecal streptococci, colonies/100 mL	....	$2\text{-}3.8 \times 10^3$	<1-47	....

a. mg/L unless otherwise noted.

7.6.5 Monitoring

Operation of the Muskegon wastewater treatment system includes an extensive monitoring program to determine the efficiency of the system and to ensure that the quality of the discharged water meets present discharge standards. The monitoring program at Muskegon is designed to evaluate influent, biological treatment, storage, postchlorination, postirrigation, lagoon seepage, groundwater, surface water, soil, and crop characteristics. Samples are taken for chemical and biological analyses once or twice daily at each step of the treatment process. On a weekly basis, a total of 2 883 samples are analyzed for one of 25 wastewater constituents [11]. In addition, groundwater is sampled monthly to twice yearly from over 300 wells for analysis [13]. This massive monitoring program requires the services of nine laboratory personnel and the results, thus far, have been that no significant effects on the groundwater or surface water of the area have occurred.

TABLE 7-22

OPERATING COSTS
MUSKEGON WASTEWATER SYSTEM, 1975 [13]

Operating costs by component	
Collection and transmission	$ 431 000
Aeration and storage	191 000
Irrigation and drainage	475 000
Farming	474 000
Laboratory and monitoring	236 000
Other	77 000
Subtotal, Muskegon	$1 884 000
Total, Whitehall	62 000
Total, gross operating	$1 946 000
Revenues	
Crop	
Corn (4 500 acres x 60 bu x $2.58/bu)	$ 698 000
Wheat (270 acres x 10 bu x $3.10/bu)	8 000
Laboratory services	8 000
Total	$ 714 000
Net operating cost	$1 232 000
Unit operating cost, ¢/1 000 gal[a]	12.5

a. Based on an average flow of 27 Mgal/d. Does not include debt retirement.

1 acre = 0.405 ha
1 ¢/1 000 gal = 0.264 ¢/m^3
1 Mgal/d = 43.8 L/s

7.7 St. Charles, Maryland

7.7.1 Project Description and History

This slow rate woodlands system was developed as a private utility for the new community of approximately 8 500 people of St. Charles in eastern Maryland in 1965. It consists of preapplication by aerated lagoons, storage, and sprinkler application of approximately 0.6 Mgal/d (26 L/s) in a wooded area (see Table 7-23). The land application portion of the system has been operated primarily as a means of disposal with little attention given to either treatment performance or crop production benefits.

TABLE 7-23

DESIGN FACTORS,
ST. CHARLES, MARYLAND [3, 14]

Type of system	Slow rate
Avg flow, Mgal/d	0.6
Type of wastewater	Domestic
Preapplication treatment	Aerated lagoons
Disinfection	Yes
Storage	
Capacity, Mgal	90
Time, d	150
Field area, acres	67
Crops	Wooded field (oak-pine)
Application technique	Sprinkler (surface pipe)
Routine monitoring	No
Buffer zones	Provided but not required
Application cycle	
Time on, h	8-15
Time off, d	4-7
Annual application rate, ft	10
Avg weekly application rate, in.	2-3.5
Avg annual precipitation, in.	40
Avg annual evaporation, in.	27
Capital costs, $/acre[a]	1 100
Unit operation and maintenance costs, ¢/1 000 gal	6

a. 1966.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 $/acre = $2.47/ha
1 ¢/1 000 gal = 0.264 ¢/m^3

The decision to build a land application system rather than a surface discharge system was based on the availability of ample undeveloped land in the 8 000 acre (3 250 ha) St. Charles plot. Soil conditions appeared to be favorable with loamy sand the predominant soil, but depth to groundwater is only 5 to 6 ft (1.5 to 1.8 m). The two major alternatives were either: (1) a 5 to 6 mile (9 to 11 km) interceptor to the Potomac River at a cost of approximately $17 million, or (2) a surface discharge to nearby Zekiah Swamp, which was declared environmentally unacceptable [3].

At this time, the Charles County Sanitary Commission and neighboring Prince George's County are constructing an interceptor and regional treatment plant in the Mattawoman Creek basin. Plans (1976) call for St. Charles to join this system when completed, abandoning this site.

7.7.2 Design Factors

The preapplication treatment originally consisted of an oxidation pond system covering approximately 10 acres (4 ha). The combination of preapplication treatment and storage lagoons now totals 40 acres (16 ha), and six floating aerators have been added to the influent cells. Effluent quality from the lagoons averages about 40 mg/L BOD and 75 mg/L suspended solids [15]. The effluent is then chlorinated to 20 MPN/100 mL fecal coliforms (maximum allowable by State of Maryland).

The land application system consists of 11 woodland plots which are sprinkled independently. Because wastewater disposal has been emphasized over treatment or tree production, only enough field area has been developed to preclude ponding and runoff. The plots receive differing application rates, depending on their ability to take the water.

Aboveground aluminum pipe is used for the distribution system. The 6 in. (15 cm) diameter distribution mains are designed to discharge the effluent when pressure is released, thus providing cold weather protection. The 2 in. (5 cm) laterals are supported above ground so that they will drain back into the mains; this appears to have caused problems in that they are easily knocked over by deer. Sprinkler spacing is mostly 90 by 80 ft (27 by 24 m), which seems to produce adequate distribution coverage at 40 $lb/in.^2$ (28 N/cm^2). The sprinklers are primarily the impact type, although it has been found that the stationary umbrella-type sprinklers reduce tree icing.

Buffer zones were not specifically required for the system, although adequate buffering is provided by the secluded nature of the site. The nearest structure is approximately 0.5 mile (0.8 km) and the nearest well is approximately 1.5 miles (2.4 km) [3].

7.7.3 Operating Characteristics

The system has continued to perform adequately for the purposes originally intended. Operation is straightforward and requires a minimum of control. A crew of two workers and a supervisor operate the system in addition to their other duties, and usually visit the site 3 to 5 times per day. Pumps and distribution lines are manually controlled. Decisions regarding application periods and cycles are based on visual appearances of the fields.

The groundwater table has risen significantly throughout most of the site and is at or near the surface in many areas [15]. In many of these areas, the original tree species have been replaced by pokeweed as a result of the high groundwater.

7.7.4 Monitoring Program

Operational monitoring of the lagoon systems at St. Charles has been reported, but the land application portion of the system has not been closely monitored. A research project is currently being conducted through a combined effort by the Departments of Agricultural Engineering, Agronomy, Botany, and Civil Engineering at the University of Maryland [15]. The study program hopes to provide information on the fluctuation of groundwater levels, the effects on water quality in groundwater and nearby water courses, the fate of materials applied, and the effects of vegetation.

7.8 Phoenix, Arizona

7.8.1 History

Rapid infiltration of municipal wastewater at Phoenix, Arizona, started in 1967 when the U.S. Water Conservation Laboratory, in cooperation with the Salt River Project and the City of Phoenix, constructed the Flushing Meadows experimental pilot project. The Flushing Meadows project demonstrated the feasibility of renovating secondary effluent for unrestricted use for irrigation and recreational purposes. Wastewater was applied to the rapid infiltration basins to evaluate the quality improvement of the effluent as it moved through the soil and the hydraulics of the groundwater recharge system. The effect of basin management on infiltration rates was examined by altering surface conditions and flooding schedules. The results for the first 5 years are well documented [16, 17]. Operation of the project is continuing and reports on the second 5 year study period will be prepared in 1978.

The 23rd Avenue Project, a large scale rapid infiltration system, was designed based on engineering criteria developed at Flushing Meadows. This project, constructed in 1974, is described in the sections which follow. Design factors for both the Flushing Meadows and the 23rd Avenue Project are presented in Table 7-24.

TABLE 7-24

DESIGN FACTORS,
PHOENIX, ARIZONA

	23rd Avenue Project	Flushing Meadows
Type of system	Rapid infiltration	Rapid infiltration
Avg flow, Mgal/d	13	0.6
Type of wastewater	Municipal	Municipal
Preapplication treatment	Secondary	Secondary
Disinfection	Yes	Yes
Storage	Not required	Not required
Field area, acres	40	1.9
Crops	None	None[a]
Application technique	Surface (basin flooding)	Surface (basin flooding)
Routine monitoring	Yes	Yes
Buffer zones	No	No
Application cycle, d		
Time on	3-21	4
Time off	3-21	10-20
Annual application rate, ft	364	365
Avg annual precipitation, in.	7	7
Avg annual evaporation, in.	70	70
Annual nitrogen loading, lb/acre	35 400	35 400

a. Several vegetated basins were experimented with, but results were inconclusive.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha

7.8.2 Purpose

A need to supplement the present water resources in the Phoenix area exists. The purpose of the 23rd Avenue Project is to demonstrate the feasibility of rapid infiltration on a scale that could partially meet this water need. If the initial demonstration project shows that wastewater can be economically renovated, the system could be expanded to reclaim all of the effluent discharged in the Phoenix area. A significant portion of the treated flow would be used for nuclear power plant cooling water. The rest could be made available for irrigation and an extensive aquatic park development (Rio Salado Project) proposed along the Salt River channel.

7.8.3 Project Description

The rapid infiltration site is located on the north side of the bed of the Salt River and east of 35th Avenue. The layout of the 23rd Avenue Project is shown in Figure 7-15. Secondary effluent from the 23rd Avenue wastewater treatment plant flows through a concrete channel to the site. The soil profile at the site is similar to the Flushing Meadows site, consisting of loamy sands, sand, gravel, and boulders to a depth of over 200 ft (60 m) [16]. On the basis of results learned at that project, infiltration rates of at least 2.5 ft/d (76 cm/d) were expected [16]. Initially, the effluent was routed through an 80 acre (32 ha) oxidation pond before application to the infiltration basins. However, the extra detention time in the oxidation pond (approximately 4 days) stimulated dense algae growths in the effluent applied to the basins and reduced the average infiltration rate to about 0.5 ft/d (15 cm/d). The algae remain in suspension and accumulate on the basin bottom as a cake. The algae cake does not decompose or shrink during drying and, consequently, the infiltration rate is not restored significantly when flooding is resumed. Pilot studies have shown that the infiltration rate can be expected to at least double when the oxidation pond is bypassed, as shown in Figure 7-15, and secondary effluent is applied directly to the basins.

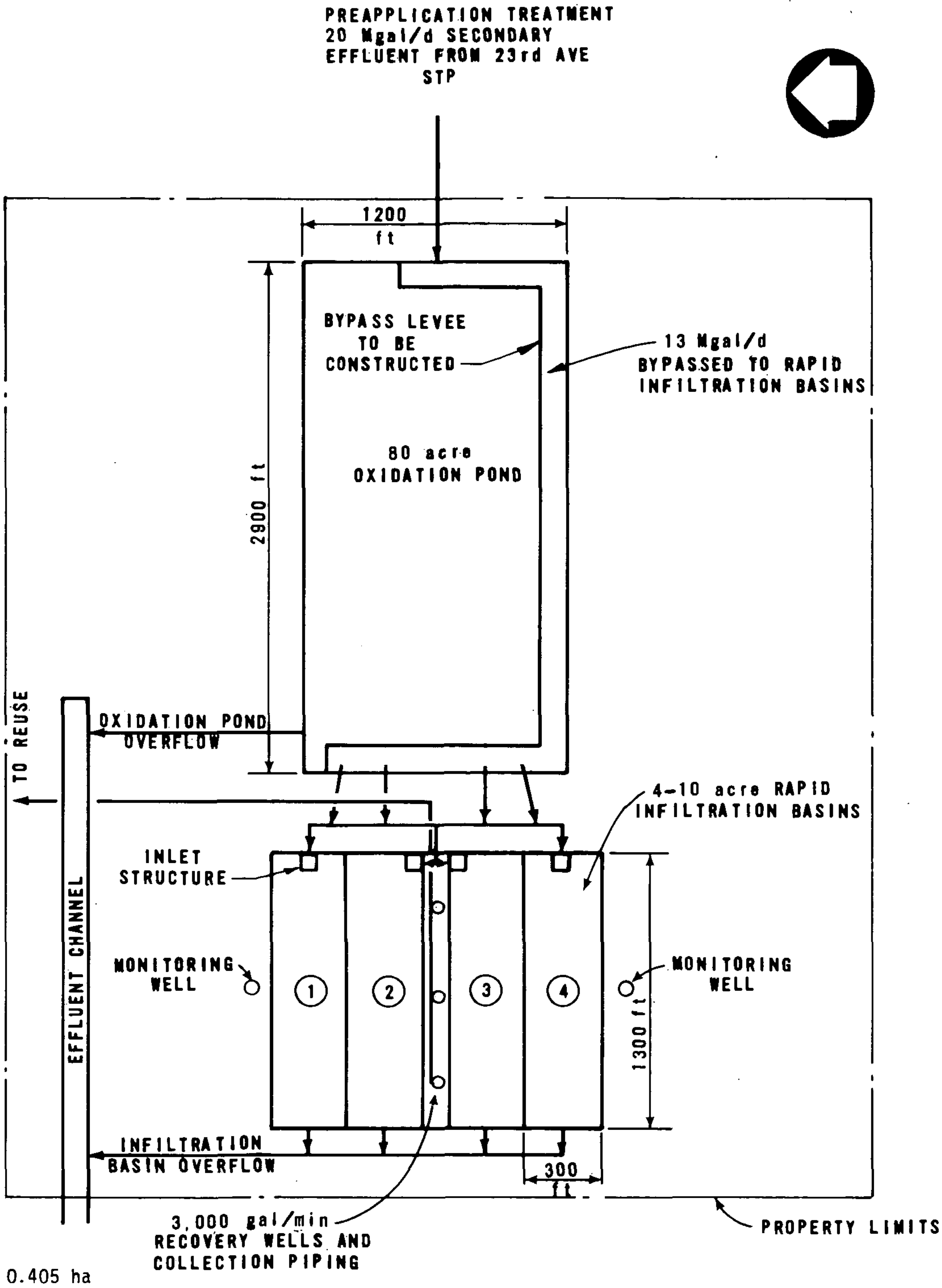
At the time of the site visit (1976), there was no reuse of the renovated water taking place. The electric rate for pumping the renovated water initially was higher than that of the local irrigation district, thus economically prohibiting its use. The City of Phoenix has negotiated with the local power company to lease the wells to the irrigation district so that it can take advantage of the lower electric rate. Resolution of this problem has made the costs of renovated water competitive with those of native groundwater sources.

7.8.4 Design Factors

At the time when the 23rd Avenue project is in full operation, about 13 Mgal/d (0.57 m^3/s) of secondary effluent will be applied to the four rapid infiltration basins. The renovated water will be recovered by a series of three 24 in. (60 cm) diameter wells (1 existing and 2 future) equipped with electric driven pumps of 3 000 gal/min (189 L/s) capacity. The static water table depth is about 80 ft (24 m). The first well is 200 ft (60 m) deep and perforated from 100 to 120 ft (30 to 36 m). A pump discharge and collection piping system will be constructed to the point of reuse. Two 6 in. (15 cm) diameter observation and sampling wells have been constructed, one each on the north and south side of the rapid infiltration basin. The wells will be used to sample renovated water quality and monitor the groundwater level. The project will be operated so that the groundwater level will be the same as that in the aquifer adjacent to the project to preclude the movement of renovated water away from the site.

FIGURE 7-15

LAYOUT OF THE 23RD AVENUE
RAPID INFILTRATION AND RECOVERY PROJECT



1 acre = 0.405 ha
1 ft = 0.305 m
1 Mgal/d = 0.043 m^3/s
1 gal/min = 0.063 L/s

7.8.5 Operating Characteristics and Performance

When the 23rd Avenue Project becomes fully operational, the effects of various wastewater cycling schedules will be studied. Inundation and dry-up periods ranging from several days each to several weeks each will be employed. During inundation, a constant depth of 1 to 3 ft (0.3 to 1.0 m) will be maintained. Inflow and outflow will be measured to evaluate infiltration rates in each basin. Basins will be drained at the end of inundation periods into the next basin to be flooded so that the dry-up period can start immediately. One of the four inlet structures to the infiltration basins is shown in Figure 7-16.

FIGURE 7-16

INLET STRUCTURE, 23RD AVENUE PROJECT, PHOENIX, ARIZONA

The unconfined groundwater table occurs at a depth of 60 to 80 ft (18 to 24 m) beneath the site and is continuous to a depth of 230 ft (69 m) where a clay layer may be located. The percolated wastewater will move toward the recovery wells at the center of the basins. The two outer basins will be inundated while the two inner basins are drying, and vice-versa. This will provide travel distances in the range of 100 to 500 ft (30 to 150 m) and 400 to 900 ft (120 to 300 m) which may yield two different qualities of reclaimed water.

By pumping only as much reclaimed water as has been infiltrated, equilibrium should be established so that no flow between the recharge system and the native groundwater takes place. The equilibrium will be checked by measuring the water levels in the observation wells. A schematic of the infiltration and the recovery process is shown in Figure 7-17.

The renovated water from the one existing recovery well has been sampled and analyzed to determine the performance of the system. Measured levels of BOD, SS, and fecal coliforms have always been far below the specified limit for unrestricted irrigation and recreation use and the state health department has certified the renovated water for these purposes. Data on the quality of renovated water at the 23rd Avenue Project are presented in Table 7-25 [18].

TABLE 7-25

RENOVATED WATER QUALITY
THE 23rd AVENUE PROJECT IN PHOENIX, ARIZONA [18][a]

Constituent	Average[a]
Suspended solids	0.8
Nitrate nitrogen	6.7
Ammonia nitrogen	0.1
Phosphorus	0.16
Fluoride	0.7
Boron	0.5
Total dissolved solids	910.0
Fecal coliforms, colonies/100 mL	0-30

a. mg/L unless otherwise noted.

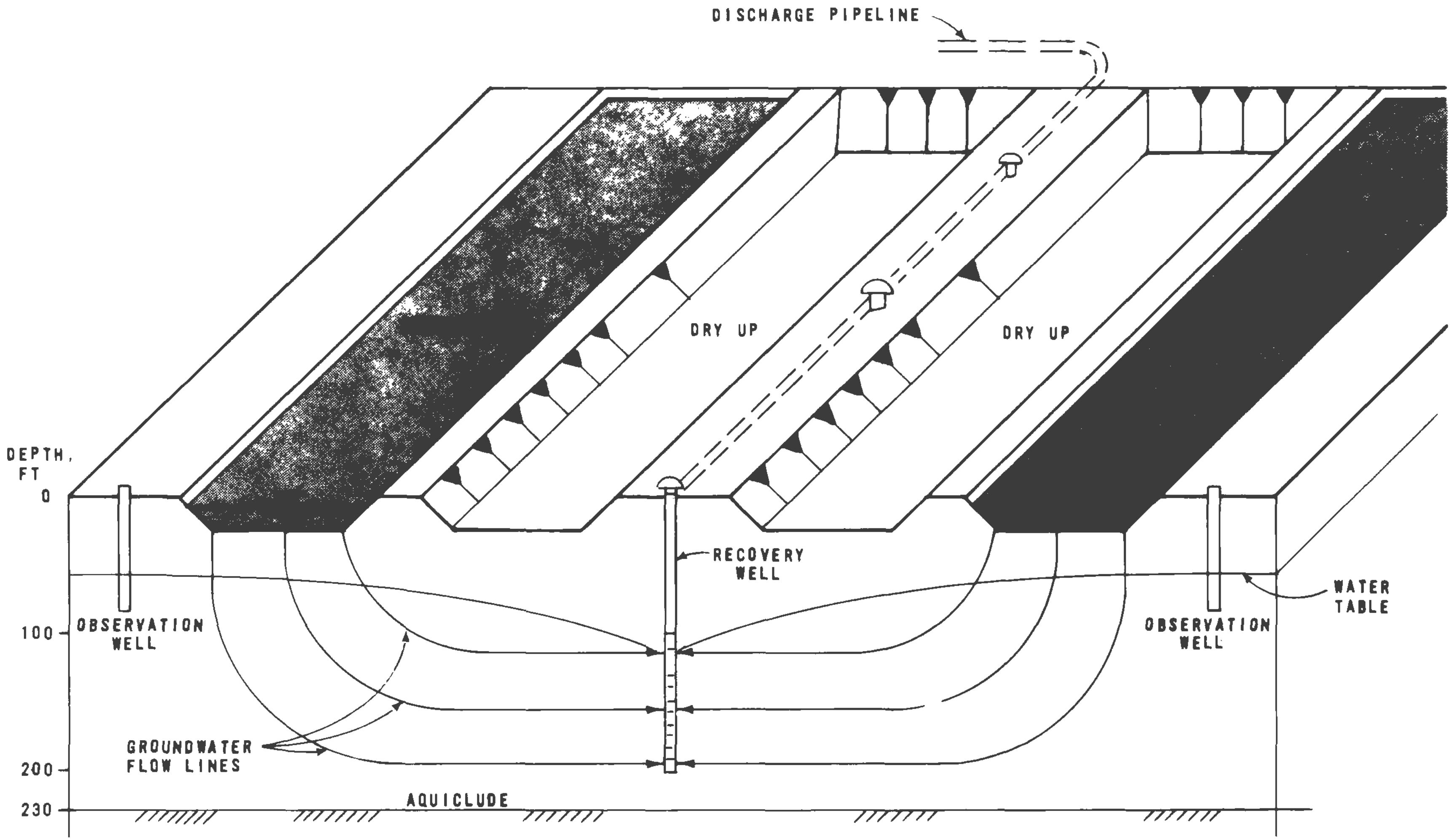
7.8.6 Monitoring Program

In addition to the quality and level of groundwater, the direction of groundwater movement will also be checked by monitoring the total dissolved solids concentration in the observation wells. If no native groundwater enters the project, the total dissolved solids of the reclaimed water should be approximately the same as that of the wastewater effluent.

Continuous 24 hour samples will be taken of the effluent entering the infiltration basins. Characteristics of the secondary effluent of the 23rd Avenue Project were not available at the time of the site visit. However, the secondary effluent from the 91st Avenue activated sludge

FIGURE 7-17

SCHEMATIC OF THE 23RD AVENUE RAPID INFILTRATION AND RECOVERY SYSTEM



renovated water from the Flushing Meadows project are shown in Table 7-26.

7.8.7 Other Research at the 23rd Avenue Project

A special objective of the 23rd Avenue Project is to determine how air pressure buildup in the soil beneath large infiltration basins affects the infiltration rate. The effect of basin size on air pressure buildup beneath the advancing wet front in the soil will be studied by comparing infiltration rates measured by two methods. Measurements will be made with cylinder infiltrometers or by comparing small inundated areas within the recharge basin to the infiltration rate when the entire basin is inundated. Piezometers have been installed to measure air pressures down to a depth of 40 ft (12 m). Reductions in infiltration rates from air pressure buildup have proved to be insignificant for small basins. Additionally, the depth of water during inundation of the basins will be varied to reduce or increase hydraulic head to determine what effect these factors have on infiltration rates.

The effects of the high algae loading on surface clogging are also being studied before the oxidation pond bypass channel is completed. Also, research is being conducted to determine whether inundation depth can be used to limit algae growth. Tensiometers have been installed to measure the increase in hydraulic impedance of the surface layer over time in the infiltration basins.

7.8.8 Other Research at Flushing Meadows

Research at Flushing Meadows has dealt with the fate of viruses in wastewater as they enter the soil. Secondary effluent and renovated water from four observation wells were assayed every 2 months in 1974 for viruses during flooding periods. The number of viruses detected in the sewage effluent averaged 2 118 per 100 litres. However, no viruses were detected in any well samples. These results indicate that viruses are reduced by a factor of at least 10^4 (99.99%) during percolation of the wastewater through 10 to 30 ft (3 to 9 m) of the basin soil [19].

The emphasis of the most recent research at Flushing Meadows is aimed at maximizing nitrogen removal. Increased nitrogen removal has been realized by reducing the hydraulic loading rate to the basins and using optimum flooding and drying periods. Preliminary results indicate that by reducing the annual hydraulic loading to the basins from 300 ft (100 m) to 173 ft (52 m), nitrogen removal increased to about 60% (from 30%) and phosphate removal increased to 90% (from 70%). These values are from samples taken from a well in the center of the spreading basins at a depth of 30 ft (10 m). The application schedule was 9 days flooding and 12 days drying.

TABLE 7-26

CHARACTERISTICS OF SYSTEM INFLUENT AND RENOVATED WATER, FLUSHING MEADOWS, PHOENIX, ARIZONA [17]

Constituent	Concentration, mg/L	
	System influent	Flushing Meadows renovated water
BOD	15	0-1
COD	45	15
Suspended solids	20-100	0
Total dissolved solids	1 100	1 100
Total organic carbon	20	5
Total nitrogen[a]	36	25
Ammonium nitrogen (NH_4^-N)	30	5-20
Nitrate nitrogen (NO_3^-N)	1	0.1-71[b]
Nitrite nitrogen (NO_2^-N)	2	1
Organic nitrogen	3	1
Phosphate (PO_4)-phosphorus	10	0.1-3[c]
Fecal coliforms per 100 mL	10^6	0[d]
Viruses per 100 mL	2 118	0
pH	7.6-8.1	7.0
Boron (B)	0.75	0.75
Fluoride (F)	4.1	2.6
Sodium (Na^+)	200	200
Calcium (Ca^{++})	82	82
Magnesium (Mg^{++})	36	36
Potassium (K^+)	8	8
Bicarbonate (HCO^3)	381	381
Chloride (Cl^-)	213	213
Sulfate (SO_4)	107	107
Carbonate (CO_3)	0	0
Cadmium (Cd)	0.008	0.007
Copper (Cu)	0.12	0.017
Lead (Pb)	0.082	0.066
Mercury (Hg)	0.002	0.001
Zinc (Zn)	0.19	0.035-0.108[e]

a. Overall nitrogen removal during sequences of long flooding and drying periods was about 30%.

b. Nitrate peaks occurred when flooding was resumed after long dry-up periods as a result of incomplete denitrification.

c. Phosphate removal increased with the underground travel time.

d. Fecal coliforms were between 0 and 200 per 100 mL in water sampled at 30 ft below the basins. Renovated wastewater from a well 200 ft away from the basins had a zero fecal coliform count.

e. High zinc level may have been the results of using galvanized plumbing in sampling wells.

7.9 Lake George, New York

7.9.1 History

Lake George, located in the eastern part of the State of New York, is a recreational lake 32 mi (52 km) long and from 1 to 3 mi (1.6 to 4.8 km) wide. The discharge of any wastewaters, treated or untreated, directly into the lake or into any tributary thereof has been strictly prohibited for at least 90 years. This has preserved the pristine quality of the lake which is still used as a public drinking water supply with no treatment other than chlorination.

By the late 1930s, Lake George Village, located at the southern end of the lake, had grown large enough to require a wastewater treatment plant. Since septic tank systems had been allowed, the regulation restricting the discharge of wastewater into the drainage basin area was interpreted to mean surface discharges. Thus, it was decided that discharge into the soil would be a satisfactory means of disposal of the treated effluent from the proposed wastewater treatment plant.

Although most of the Lake George watershed is underlain by rock consisting of pre-Cambrian gneisses, a small natural delta sand deposit created by outwash from the receding glaciers was discovered at the southwest corner of the Lake George Village area. Advantage was taken of this mass of delta sand and the wastewater treatment plant was constructed at this location to utilize this sand as a rapid infiltration area for the secondary effluent. The original treatment plant was completed and put into operation in 1939 and has been in continuous operation ever since. Design factors for the rapid infiltration system are presented in Table 7-27. A view of a rapid infiltration basin is shown in Figure 7-18. During the winter ice forms on the basin surface (Figure 7-19) and the applied wastewater floats the ice and infiltrates into the sandy soil.

7.9.2 Project Description

The Lake George Village wastewater treatment plant receives wastewater from two force mains, one from the Village and one from the Town of Lake George. There are five pumping stations, including two located in town which lift the wastewater approximately 200 ft (60 m) from the collection point at the lake to the treatment plant. Primary treatment is provided by one circular Imhoff tank and two mechanically cleaned circular Clarigesters (similar to Imhoff tanks), all operating in parallel. Secondary treatment consists of two high-rate rotating arm trickling filters and one covered standard-rate fixed nozzle trickling filter. The latter is used exclusively in the winter and is covered to prevent icing of the sprayed wastewater. Secondary sedimentation is accomplished by two rectangular and two circular settling tanks. After secondary sedimentation, the unchlorinated effluent is passed onto the natural delta sand beds for infiltration into the soil. At present, there are 14 north and 7 south sand beds. Sludge from the secondary settling tanks is returned to the Clarigesters, and digested sludge is applied to 3 sludge drying beds. The general layout of the treatment plant and the location of the sand beds and sampling wells are shown in Figure 7-20.

TABLE 7-27

DESIGN FACTORS,
LAKE GEORGE, NEW YORK

Type of system	Rapid infiltration
Avg flow, Mgal/d	1.1 (summer) 0.4 (winter)
Type of wastewater	Domestic
Preapplication treatment	Secondary
Disinfection	No
Storage	None
Field area, acres	5.4
Crops	None
Application technique	Surface (basin flooding)
Routine monitoring	No
Buffer zones	No
Application cycle	
Time on, h	8-24
Time off, d	4-5 (summer); 5-10 (winter)
Avg annual application rate, ft	140
Avg annual precipitation, in.	34
Avg annual evaporation, in.	26
Avg nitrogen loading, lb/acre	6 700

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha

7.9.3 Design Factors

It is estimated that the Lake George Village wastewater treatment plant, with a design capacity of 1.75 Mgal/d (76.7 L/s), presently serves a population of approximately 2 100 in the winter and 12 300 in the summer [20]. In 1965, the plant underwent major expansion with the addition of eight sand beds, and in 1970 one additional bed was put on line to bring the total to 21. The material in the beds ranges from coarse to fine sand, with a few beds having some clay content. Depth to water table and bedrock varies, but is generally deeper in the old north sand beds

FIGURE 7-18

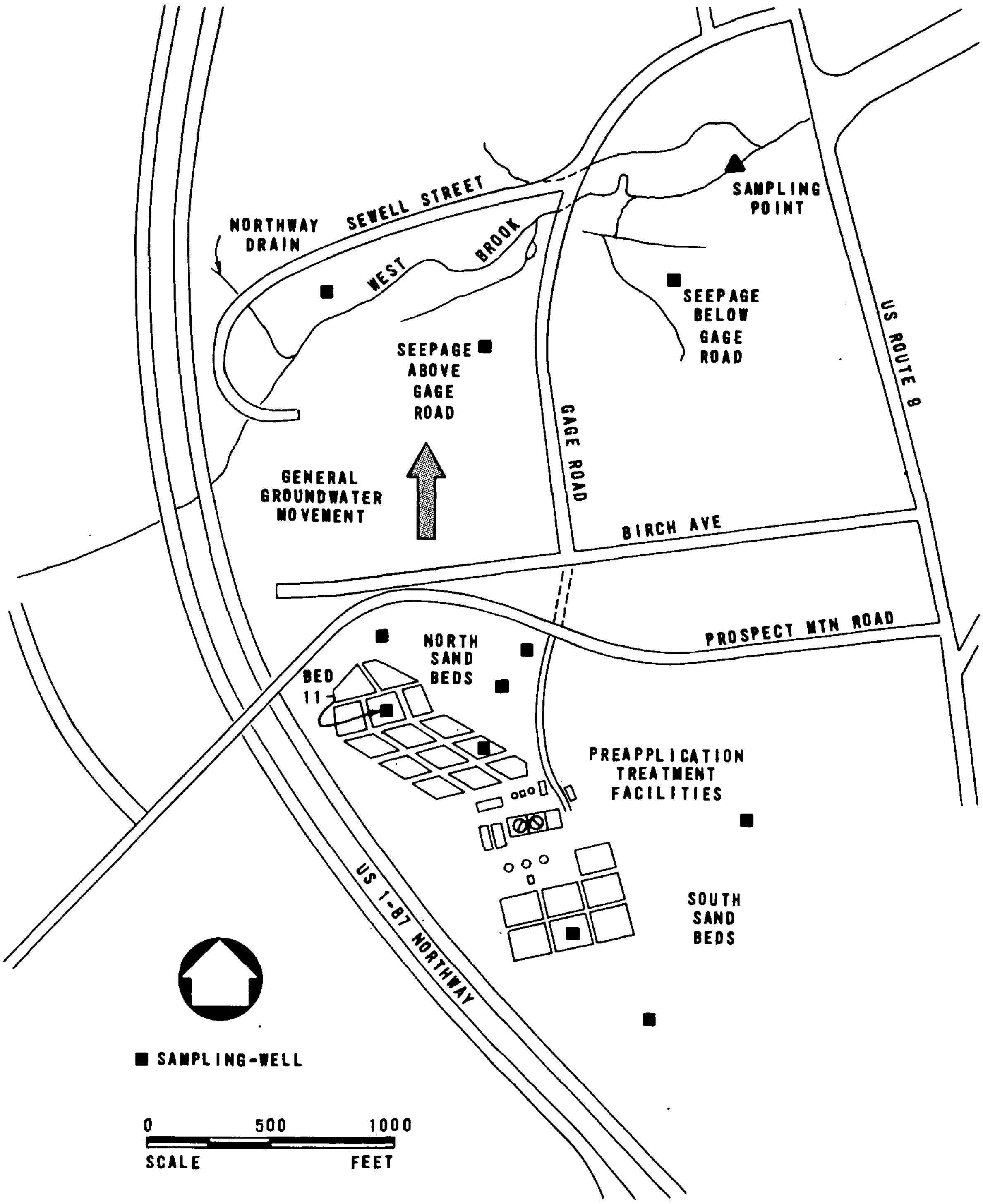
RAPID INFILTRATION BASIN, LAKE GEORGE, NEW YORK

FIGURE 7-19

OPERATIONAL BASIN COVERED WITH ICE,
LAKE GEORGE, NEW YORK

FIGURE 7-20

LAKE GEORGE VILLAGE WASTEWATER TREATMENT
PLANT AND SAMPLING LOCATIONS



than in the new south sand beds. Well points driven in bed 11 of the north sand beds have found the water table to be at a depth of approximately 65 ft (20 m) below the surface, and bedrock to be approximately 90 ft (27 m) below the surface. The sand beds vary in size, ranging from 0.16 acre to 0.42 acre (0.06 to 0.17 ha), and combine for a total surface area of 5.4 acres (2.2 ha) [21].

The unchlorinated secondary effluent is discharged onto the natural delta sand beds by surface flooding. In order to prevent erosion of the sand at the point of discharge, concrete splash pads with brick baffles are provided. Individual sand beds are dosed by adjusting the gates within the distribution chambers. The beds have 3 to 5 ft (1 to 1.5 m) dikes around them, and each bed has a control valve for individual flooding. The 14 lower (north) beds are fed by gravity, while effluent from the secondary settling tank must be pumped up to the 7 upper (south) beds. A float control in the wet well automatically operates the intermittent pump.

Vertical movement of the infiltrated effluent through the sand ranges from 15 to 75 ft (4.5 to 20 m), depending on the sand bed and the season. Horizontal underground movement is approximately 2 000 ft (600 m) before the renovated effluent emerges as seepage near West Brook, a tributary to Lake George.

7.9.4 Operating Characteristics and Performance

Normal weekday operation of the sand beds is to dose one north and one south bed with 8 to 10 in. (20.4 to 25.4 cm) of effluent over an 8 hour period during the day. A similar pair of beds are flooded throughout the remaining 16 hours. On weekends, two north and two south beds are dosed for a period of 24 hours each. During the high flow months of the summer, more than 2 beds are flooded at a time.

There is no set schedule as to which rapid infiltration bed will be used on any one day. Plant personnel make daily decisions based on visual inspection of the status of the beds. Most of the sand beds dry in 1 to 3 days. Generally, the beds are rested for 5 to 10 days prior to the next application. The frequency of application increases with the increase in flow due to the influx of tourists during the summer months. During the peak flows of August, it is often necessary to flood the sand beds before they have fully dried. This practice is avoided if at all possible, as the surface of the beds must remain aerobic in order to restore the renovative capacity of the system.

It has been found that the rapid infiltration basins perform well and clog slowly under conditions of 1 day dosing followed by several days of drying. The rest period, providing complete or partial drying, has a renewing effect on the infiltration capacity of the sand beds. In addition, the sand beds are occasionally reconditioned by raking or scraping the surface. The top few centimetres of sand are removed, which include a mat of algae and other organic material, and the sand bed is regraded. This cleaning operation is generally restricted to spring or autumn, when weather is mild and flows are not at a peak. Weeds are removed for aesthetic purposes. There have been no serious problems with the operation of the sand beds.

Application continues year-round without storage, regardless of severe winter weather. In winter, part of the water freezes and forms an ice layer which may attain 1 ft (0.3 m) in thickness. This does not interfere with the operation. The warm effluent flows under the ice, simultaneously melting the ice above it and the ground below it, and in effect, floats the ice layer. The ice is actually beneficial to the process, as it serves as an insulating layer for the soil surface.

7.9.5 Environmental Studies

In an effort to evaluate the environmental effects of the Lake George Village wastewater treatment plant, numerous studies have been conducted by Rensselaer Polytechnic Institute, the New York State Health Department, and the New York State Department of Environmental Conservation. The Rensselaer Fresh Water Institute was organized and studies were begun in 1968. A number of well points were placed in the sand beds and at the periphery of the treatment plant grounds, as shown in Figure 7-20. Two additional well sites are located between the sand beds and West Brook and one is located across West Brook.

Analysis of water samples from the wells has shown that there is almost complete removal of BOD, coliforms, ammonia nitrogen, and organic nitrogen in the top 10 ft (3 m) of passage through the sand beds [22]. Ammonia and organic nitrogen are converted to nitrate-nitrogen and, at least partly, the nitrate is reduced to nitrogen gas by denitrification. Phosphorus removal is a function of the frequency of sand bed use, with a bed in constant use having considerably less phosphate removal than an infrequently used bed for the same distance of downward percolation.

A resistivity survey has indicated that the most probable direction of flow of the wastewater discharged onto the sand beds is northerly along Gage Road toward West Brook [23]. The seepage which occurs above and below Gage Road is tributary to West Brook and has been estimated to be approximately 0.6 Mgal/d (26 L/s), or 10% of the total flow of West Brook [24].

Water quality data of the plant effluent and seepage above Gage Road and West Brook are given in Table 7-28. The water which emerges from the ground in the area of West Brook contains considerably higher concentrations of dissolved solids, alkalinity, and chloride than the

TABLE 7-28

WATER QUALITY DATA, SEASONAL MEANS, LAKE GEORGE, NEW YORK [25]

	Temperature, °C	Dissolved oxygen, mg/L	Dissolved solids, mg/L	pH	Alkalinity, mg/L as $CaCO_3$	Chloride, mg/L	Nitrate-nitrogen, mg/L	Ammonia nitrogen, mg/L	Total Kjeldahl nitrogen, mg/L	Soluble phosphate, µg/L	Total phosphorus, µg/L
Plant effluent applied to sand beds											
Spring	13.0	5.3	177	7.2	96	44	1.8	3.8	8.0	750	1 555
Summer	22.4	2.1	224	6.9	218	46	1.6	15.9	18.4	2 950	3 950
Fall	10.0	4.5	197	7.0	93	32	3.5	3.0	9.1	700	1 650
Winter	4.5	6.8	234	7.0	109	52	1.1	5.1	12.5	488	1 425
Seepage above Gage Road											
Spring	10.8	10.3	160	7.9	106	37	2.3	0.0	0.2	8	16
Summer	14.3	8.2	173	7.8	99	49	1.6	0.0	0.1	14	16
Fall	8.9	10.1	220	7.9	111	42	3.5	0.1	....	<2	.....
Winter	4.8	11.3	212	7.8	118	40	3.8	0.0	0.1	10	10
West Brook downstream of seepage											
Spring	10.7	11.0	85	7.4	35	15	0.7	0.0	0.2	1	6
Summer	12.6	10.2	120	7.8	68	29	1.8	0.1	0.1	3	3
Fall	8.0	11.5	93	7.5	56	25	1.5	0.0	0.0	1	2
Winter	2.0	13.0	79	7.3	39	14	0.6	0.0	0.1	2	6

natural groundwater in the area. This is evidence that the seepage does in fact originate from wastewater effluent. From the data, it can be seen that the total phosphorus content of the applied wastewater is reduced by greater than 99% in its passage through the approximately 2 000 ft (600 m) of sand before it emerges and runs off into West Brook and ultimately into Lake George. It also can be seen that the applied nitrogen is oxidized to nitrate prior to its emergence from the ground. The nitrate content of the seepage is about 1.6 to 3.8 mg/L and increases the nitrate content of West Brook. However, the nitrate-nitrogen concentration in West Brook downstream of the seepage is about 0.6 to 1.8 mg/L, which is well below the EPA drinking water standard of 10 mg/L.

Based on numerous studies and extensive sampling and analyses, the land treatment system at Lake George is doing an adequate job of purifying the wastewater to a drinking water quality [22]. The soil system is satisfactorily removing essentially all of the phosphorus and is providing a nitrified effluent which appears to have no deleterious effect upon the quality of Lake George.

7.10 Fort Devens, Massachusetts

7.10.1 History

Fort Devens is a U.S. Army military installation located in the Nashua River basin about 32 mi (52 km) northwest of Boston, Massachusetts. A rapid infiltration system at Fort Devens has received an unchlorinated primary sewage effluent for over 35 years. The total population and wastewater flows have fluctuated over the years, but are presently on the decline. In 1973, the daytime population was about 15 000 of which 10 400 were permanent residents [26]; whereas the 1976 population has been estimated to be 10 000 and 7 000, respectively. The present wastewater treatment facility has been providing continuous service since its construction in 1942. Selected design factors are presented in Table 7-29.

7.10.2 Project Description and Design Factors

The Fort Devens wastewater treatment facility has a design capacity of 3.0 Mgal/d (131 L/s), but has been receiving from 1.0 to 1.3 Mgal/d (43 to 57 L/s) for the last several years. Comminuted, degreased wastewater is pumped from a central pumping station to three Imhoff tanks which provide primary treatment. Settleable solids accumulate on the bottom of the Imhoff tanks and are withdrawn to sludge drying beds in April and in November of each year. These dewatering beds are underdrained and discharge to an adjacent wetland area [27].

TABLE 7-29

DESIGN FACTORS,
FORT DEVENS, MASSACHUSETTS

Type of system	Rapid infiltration
Avg flow, Mgal/d	1.3
Type of wastewater	Domestic
Preapplication treatment	Primary (Imhoff tank)
Disinfection	No
Storage	Not required
Field area, acres	16.6
Crops	None (weeds)
Application technique	Surface (basin flooding)
Routine monitoring	No
Buffer zones	No
Application cycle, d	
Time on	2
Time off	14
Avg annual application rate, 1960 to 1973, ft	94
Avg annual precipitation, in.	44
Avg annual evaporation, in.	26
Annual nitrogen loading, lb/acre	11 200

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha

Final treatment of the unchlorinated primary effluent is achieved by discharging to 22 rapid infiltration basins. These 22 basins provide a total field area of 16.6 acres (6.7 ha) or an average of 0.76 acre (0.31 ha) per basin [28]. They are situated on the top of a steep-sided hill composed of a 200 ft (60 m) thick layer of unconsolidated stratified sand and gravel deposited by receding glaciers. This flat, oval-shaped hilltop rises approximately 70 ft (21 m) above the floodplain of the Nashua River [26]. The soil formation in which the treatment beds were constructed is primarily poorly graded sands or gravelly sands with interspersed lenses of silty sand and sandy gravels. Particle size distribution differs appreciably between the various soil horizons in the beds. The layout of the Fort Devens land treatment facility is schematically depicted in Figure 7-21.

7.10.3 Operating Characteristics

Effluent is distributed within each treatment bed by discharging onto a tapered concrete trough with slotted wooden splashboards, as shown in

Figures 7-21 and 7-22. A view of several grass-covered basins with accumulated organic material on the surface is shown in Figure 7-23.

Under normal operating conditions, the application cycle consists of flooding three treatment beds concurrently with effluent for a 2 day period, then allowing a 14 day recovery or dry-up period. On a yearly basis, each bed receives effluent for a total of 52 days [27].

After the 2 days of flooding, effluent has normally accumulated on the surface of the beds to a depth of 0.5 to 1.6 ft (15 to 50 cm). This standing water infiltrates the beds within the initial 2 or 3 days of the recovery period, restoring aerobic conditions to the surface of the beds. Winter conditions, while reducing infiltration rates somewhat, do not interfere with normal operations. The effluent is sufficiently warm, 46 to 54°F (8 to 12°C) during the winter to melt any accumulated ice and snow cover and to infiltrate and move through the sand beds.

Operation of the Fort Devens rapid infiltration basins normally involves no routine maintenance. Solids build up on the surface, dry and crack during the recovery period, and are degraded under the prevailing aerobic conditions. During the summer, the sand beds have a good stand of naturally occurring annual grasses and weeds (see Figure 7-22 and 7-23). No attempt is made to remove this vegetation as there is no apparent detrimental effect. However, renovation of the bed surface has been performed. This renovation consists of excavation to a depth of 1 ft (0.3 m) depth to 1.5 to 4.0 ft (0.45 to 1.22 m) in order to remove a an area adjacent to the treatment beds. The exposed surface is scarified or raked prior to replacement of the excavated material. It should be pointed out that this renovation procedure is not required very often. The only cleaning operation was completed in October 1968 [26]. At this time it was necessary to excavate below the specific 1 ft (0.3 m) depth to 1.5 to 4.0 ft (0.45 to 1.22 m) in order to remove a tarlike layer about 1.5 ft (0.45 m) thick which had formed below the surface of the beds. Since the discovery of this tarlike layer, there has been more surveillance of the dumping of oils and grease into the system. Grease traps, installed at various locations in the collection system to remove kitchen grease and fats and various oils from the wastewater, are cleaned more frequently, and the materials collected are deposited in sanitary landfills [27].

Normal operation and maintenance of the Fort Devens treatment facility is carried out by two full-time employees. The application of daily flows to various combinations of treatment beds is based on the continued capacity of the beds to accept the effluent and from operational experience developed over the years.

FIGURE 7-21

LAND TREATMENT SYSTEM, FORT DEVENS, MASSACHUSETTS

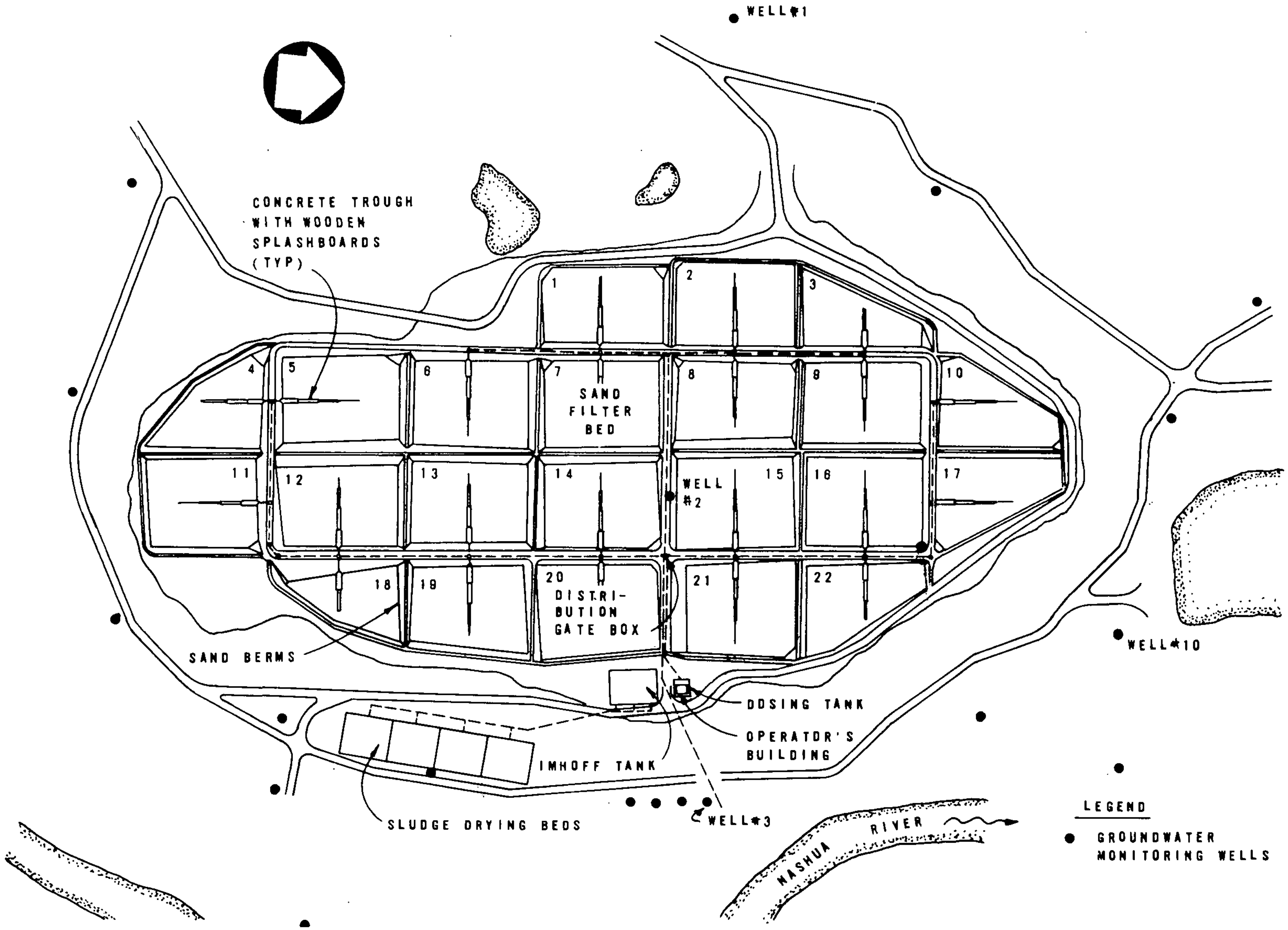


FIGURE 7-22

DISTRIBUTION CHANNEL INTO RAPID INFILTRATION BASIN,
FORT DEVENS, MASSACHUSETTS

FIGURE 7-23

GRASS COVERED INFILTRATION BASINS,
FORT DEVENS, MASSACHUSETTS

7.10.4 Treatment Performance

During 1973 and 1974, the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL) conducted extensive studies to determine the effectiveness of the rapid infiltration basins at Fort Devens to renovate unchlorinated primary sewage effluent. Groundwater quality beneath the application site and the surrounding area was monitored by collecting and analyzing bi-weekly samples from 21 observation wells (Figure 7-21). Results of the chemical and bacteriological analyses of the primary effluent and selected observation wells are summarized in Table 7-30.

Analysis of the data has proved that the rapid infiltration system serving Fort Devens is treating unchlorinated primary sewage effluent to a quality comparable to that achieved by conventional tertiary wastewater treatment facilities. The treatment basins were found to greatly reduce the levels of BOD_5, COD, organic and ammonia nitrogen, phosphorus, and total coliform bacteria in the applied effluent. Although most wastewater constituents were increased in the native groundwater, the quality of the groundwater peripheral to the treatment sites continues to meet EPA drinking water standards, with the exception of nitrate-nitrogen and coliform bacteria. While fecal coliform determinations proved negative, total coliforms showed a mean value of 200 per 100 mL in the peripheral groundwater wells [26].

7.10.5 Research Studies

In 1974, further studies by CRREL were undertaken in an attempt to optimize nitrogen removal. The objectives were to remove greater amounts of nitrogen by management of the treatment system to enhance the nitrification-denitrification processes. In an effort to achieve this, the application cycle was modified from inundating 3 beds for 2 days, followed by a 14 day recovery period, to inundating 9 beds for 7 days, followed by a 14 day recovery period. Results of this study have shown that an increase in inundation period continued to renovate the primary effluent to a degree comparable to before. The total nitrogen levels of the groundwater continued to be 20 mg/L. However, when the treatment basins were inundated for 7 days, the percentage of total nitrogen removal was greater than when the basins were inundated for 2 days. By increasing the inundation period, total nitrogen additions were increased by 54% from about 32 to 50 lb/acre·d (36 to 55 kg/ha·d). Although total nitrogen additions were larger during the 1974 study, a proportional increase in groundwater nitrogen levels was not observed, indicating a greater percentage of nitrogen removal. However, after 6 months of increased inundation period, the infiltration capacity had been reduced so much that the basin surfaces were still wet at the beginning of the next cycle of inundation and recovery. This gradual decline in the basin infiltration capacity over several months was attributed to clogging of the surfaces of the basins by accumulating organic matter. It was found that an occasional extended recovery period of 60 consecutive days will rejuvenate the infiltration capacity of the treatment basins so that the 7 day application/14 day recovery cycle can once again be used. The restoration of infiltration capacity during the extended recovery period is attributed to the aeration of the surface and the subsequent oxidation of accumulated organics [29].

TABLE 7-30

CHEMICAL AND BACTERIOLOGICAL CHARACTERISTICS OF PRIMARY EFFLUENT AND GROUNDWATER IN SELECTED OBSERVATION WELLS, FORT DEVENS LAND TREATMENT SITE (1973, Average Values) [27]

Constituent[b]	Primary effluent	Well[a] 1[c]	2	3	10
BOD_5	112	3.5	12	2.5	0.9
COD	192	42	26	19	10
Total nitrogen	47	1.3	14.5	19.5	20.3
Organic nitrogen	23	0.5	8.3	2.3	1.2
Ammonium nitrogen (NH_4-N)	21	0.6	5.3	1.3	0.5
Nitrate nitrogen (NO_3-N)	1.3	0.2	0.9	15.6	18.6
Nitrite nitrogen (NO_2-N)	0.02	0.01	0.03	0.3	0.02
Total phosphate (PO4-P)	11	0.4	5.9	0.9	1.3
Ortho phosphate (PO_4-P)	9	0.1	5.6	0.2	0.1
Chloride	150	20	85	230	257
Sulfate	42	9	48	39	35
Total coliforms, MPN/100 mL	3.2×10^7	335	3 900	210	620
ph, units	6.2 - 8.0	7.3	6.8	6.3	6.1
Conductivity, µmhos	511	133	371	360	333
Alkalinity (as $CaCO_3$)	155	29	120	28	14
Hardness (as $CaCO_3$)	41	12	23	44	30
Depth of well below ground level, ft	--	40	64	9.5	23

a. Well locations are shown in Figure 7-20.

b. mg/L unless otherwise noted.

c. Indicative of native groundwater quality.

1 ft = 0.305 m

A tracer study conducted by the U.S. Army Medical Bioengineering Research and Development Laboratory has demonstrated that viruses are capable of movement past the upper soil layers [28]. The wastewater was artificially spiked to provide a continuous virus concentration of 10^5 PFU/mL of wastewater applied to the treatment beds. This is a much greater virus concentration than that normally found in domestic wastewaters. The field studies at Fort Devens have shown that viruses at this concentration are not impeded in the local soil strata and can readily penetrate to the groundwater. In addition to poor adsorption, other removal mechanisms such as filtration or straining were not a factor, mainly because of the size of the sandy, silty, and gravelly soils in relation to the extremely small virus particles. The virus stabilized in the groundwater beneath the treatment basins at almost 50% of the artificially high applied virus concentration.

The bacteriological indicator organisms were reported to behave differently than the viruses at the rapid infiltration site. Total coliform, fecal coliform, and fecal streptococcus organisms were readily concentrated on the soil surface. Unlike the viruses, the bacteria were filtered or strained at the soil surface. However, it was reported that significant numbers of bacteria are capable of migration into the groundwater [28].

7.11 Pauls Valley, Oklahoma

7.11.1 History

Pauls Valley is a community of 6 000 in south central Oklahoma. In 1962, a 4 cell, 33 acre (13 ha) lagoon was constructed to treat 0.7 Mgal/d (31 L/s) of wastewater, with some effluent used for irrigation. In 1975, an experimental overland flow system was constructed to treat a portion of the flow. Much of the experimental system was patterned after the EPA research project at nearby Ada, Oklahoma [30, 31]. The principal design factors are summarized in Table 7-31.

7.11.2 Project Description and Objectives

The purpose of the experimental system is to demonstrate the treatment of both oxidation lagoon effluent and untreated municipal wastewater by overland flow. The system consists of 32 terraces, each 0.25 (0.1 ha), for a total of 8 acres (3.2 ha). Lagoon effluent is supplied to 8 terraces and screened untreated wastewater is supplied to the remaining 24 terraces. Lagoon effluent is taken from the second cell where it has received approximately 30 days of detention.

Half the terraces are sloped at 2% and half at 3%. A typical terrace is 75 ft wide by 150 ft long (23 m by 45 m). Three distributor mechanisms

TABLE 7-31

DESIGN FACTORS,
PAULS VALLEY, OKLAHOMA

Type of system	Overland flow
Avg flow, Mgal/d	0.2
Type of wastewater	Domestic
Preapplication treatment	Raw (screened) and oxidation lagoon
Disinfection	No
Storage	Not required
Field area, acres	8
Crops	Fescue, annual rye, and Bermuda grass
Application technique	Surface (bubbling orifice) and sprinkler (fixed and rotating nozzle)
Routine monitoring	Yes
Buffer zones	No
Application cycle, h	
Time on	8-12
Time off	12-16
Annual application rate, ft	
Screened untreated wastewater	19
Oxidation lagoon effluent	45
Avg weekly application rate, in.	
Screened untreated wastewater	4.3
Oxidation lagoon effluent	10.3
Avg annual precipitation, in.	36
Avg annual evaporation, in.	58.5
Capital costs, \$/acre[a]	8 500

a. Includes construction costs of preapplication treatment and engineering, 1975.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha
1 \$/acre =\$2.47/ha

are used: (1) rotating booms with fan nozzles, (2) fixed fan nozzles at the top of the slope as shown in Figure 7-24, and (3) the bubbling orifice method as shown in Figure 7-25. The rotating boom is patterned after those used at the Ada, Oklahoma research project [30].

The purpose of the multiple terraces is to compare the treatment efficiencies and the operating conditions for: (1) screened untreated wastewater versus oxidation lagoon effluent, (2) slopes at 2% versus slopes at 3%, and (3) the three types of distributors.

FIGURE 7-24

FIXED FAN NOZZLE, PAULS VALLEY, OKLAHOMA

7.11.3 Design Factors

The original seeding was 30 lb/acre (34 kg/ha) fescue and 15 lb/acre (17 kg/ha) annual rye. During the summer the annual rye dies out and subsequent seeding with Bermuda grass has begun to grow. The application is year-round; however, the oxidation lagoon acts as a backup system and would provide storage if needed. The soil is a slowly permeable red clay.

The bubbling orifice consists of a 6 in. (15 cm) PVC manifold with 0.75 in. (1.9 cm) outlets. The manifold is cradled in readily available crushed limestone that is 0.6 in. to 1.50 in. (1.6 to 3.8 cm) in diameter. The flow of wastewater spreads out and slows down as it contacts the limestone and begins to travel down the slope.

FIGURE 7-25

BUBBLING ORIFICE FOR WASTEWATER APPLICATION,
PAULS VALLEY, OKLAHOMA

7.11.4 Operation and Performance

Two related operating problems have taken much of the first year to solve. The screening device used was not successful initially and large solids were pumped into the system. This has resulted in frequent clogging of the sprinkler nozzles. Improved screening has reduced the clogging. Second, the grasses suffered from the heat and the occasional dry periods of the first summer. Bermuda grass may become the principal vegetation because of its tolerance for heat and water.

7.11.5 Costs

The construction cost for the research system in 1975 was approximately $68 000, including roads, fencing, seeding, preapplication treatment, earthwork, distribution, runoff piping, and engineering. Unit costs of the three distributor systems are presented in Table 7-32. Each unit supplies wastewater to a 0.25 acre (0.1 ha) terrace along the top of the slopes.

TABLE 7-32

UNIT COSTS OF OVERLAND FLOW APPLICATION,
PAULS VALLEY, OKLAHOMA

Item	Unit	Number	Unit cost, $	Cost, $ per acre
Fixed nozzle systems	each	8	200	800
Rotating boom systems	each	16	375	1,500
Bubbling orifice systems	each	8	140	560

7.12 Paris, Texas

7.12.1 History

In 1960, the Campbell Soup Company began to construct an overland flow system at Paris, Texas. When the food processing plant began operating at the end of 1964, there were 300 acres (120 ha) of prepared slopes with a vegetative cover of mixed grasses ready for wastewater treatment. The system has been expanded in three increments to the present 900 acres (360 ha). In 1968, a 12 month intensive monitoring program was conducted and the results have been widely published [32, 33, 34, 35]. The principal design factors are presented in Table 7-33.

7.12.2 Objectives and Description

The objective of the overland flow system is to treat the food processing wastewater in an efficient and cost-effective manner [36]. The construction of the overland flow system also resulted in the reclamation of the heavily eroded rolling terrain.

Wastewater from the heat processing of soups, beans, and spaghetti-type products is collected in two drainage systems. The first, containing grease from cooking, is routed through a gravity grease separator before it joins the second waste stream from the vegetable trimming area. The combined stream passes through revolving drum-type #10-mesh screens prior to being pumped to the sprinklers [34].

TABLE 7-33

DESIGN FACTORS,
PARIS, TEXAS

Type of system	Overland flow
Avg flow, Mgal/d	4.2
Type of wastewater	Food processing
Preapplication treatment	Grease removal and screens
Disinfection	No
Storage	Not required
Field area, acres	900
Crops	Reed canary, tall fescue, redtop, and perennial rye
Application technique	Sprinkler (buried pipe)
Routine monitoring	Yes
Buffer zones	No
Application cycle, h	
Time on	6-8
Time off	16-18
Annual application rate, ft	5.2
Avg weekly application rate, in.	2-3
Avg annual precipitation, in.	45
Avg annual evaporation, in.	36
Annual nitrogen loading, lb/acre	240
Capital costs, \$/acre[a]	1 500
Unit operation and maintenance cost, ¢/1 000 gal[b]	4.8

a. Excluding land, 1976.

b. 1971.

1 Mgal/d = 43.8 L/s
1 acre = 0.405 ha
1 ft = 0.305 m
1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha
1 \$/acre = \$2.47/ha
1 ¢/1 000 gal = 0.264 ¢/m^3

Wastewater is applied to the overland flow terrace by impact-type sprinklers. The original sprinkler system consisted of 4 in. (10 cm) aluminum irrigation pipe as laterals laid on the surface, but the more recently constructed terraces have buried laterals. The treated wastewater is collected as runoff in grassed waterways and is discharged into a creek.

7.12.3 Design Features

While the current hydraulic loading is 5.2 ft/yr (1.6 m/yr), the system has operated effectively at higher rates. In the 1968 research program, the total annual application was measured at 11 ft (3.4 m) for the 11.4 acres (4.6 ha) monitored and rainfall was 4.7 ft (1.4 m). Of this total amount of water, 18% was accounted for as evapotranspiration, 61% as runoff, and 21% assumed as percolation [34].

The rolling terrain was graded into terraces with slopes ranging from 1 to 12%. In the more recently added fields, slopes of from 2 to 6% are used. Slope lengths range from 200 to 300 ft (61 to 92 m). The slopes are seeded to a mixture of Reed canary grass, tall fescue, red top, and perennial rye grass [36]. Reed canary grass has become the predominant grass on the mature slopes.

7.12.4 Operation and Performance

The treatment performance documented in 1968 is compared to recent effluent quality in Table 7-34. BOD and COD removals on a concentration basis have improved and are relatively consistent throughout the year, as shown in Table 7-35. The suspended solids removals are not as high as BOD removals and are not as consistent. Despite the wide range in pH of the wastewater, the runoff is consistently between 6.6 and 7.5.

TABLE 7-34

TREATMENT PERFORMANCE DURING 1968
COMPARED TO EFFLUENT QUALITY IN 1976, PARIS, TEXAS [35, 37]

	1968 values		June 1976
Constituent	Influent	Treated effluent	Treated effluent
BOD, mg/L	572	9	1.9
COD, mg/L	806	67	45
Suspended solids, mg/L	245	16	34
Total nitrogen, mg/L	17.2	2.8	....
Total phosphorus, mg/L	7.4	4.3	....
Chloride, mg/L	44	47	43
Electrical conductivity, μmhos/cm	449	490	....
pH, unit	4.4-9.3	6.2-8.1	6.6

TABLE 7-35

SEASONAL QUALITY OF TREATED EFFLUENT
PARIS, TEXAS
mg/L

Month	BOD	COD	Suspended solids
1975			
Jul	3.1	44	34
Aug	3.4	43	17
Sep	1.9	38	15
Oct	2.7	32	15
Nov	2.7	36	23
Dec	3.1	34	15
1976			
Jan	6.5	38	15
Feb	3.6	40	19
Mar	3.4	44	37
Apr	4.6	50	76
May	2.3	43	38
Jun	1.9	45	34
Average	3.3	41	28

The grass was cut but not removed in 1965 and 1966. In 1967, the hay was harvested and in 1968 three cuttings were made for a total yield of 3.65 tons/acre (8.2 Mg/ha) [32]. Currently, the grass is cut once a year and it is harvested green, dried in a hay dryer, and converted to pellets for animal feed [36]. The grassed terraces are shown in Figure 7-26. Because the slopes are nearly always wet, access is restricted to vehicles with high-flotation tires.

7.12.5 Costs

Construction and operating costs reported in 1971 are shown in Table 7-36. It is estimated that the $1 007/acre ($2 483/ha) construction cost (excluding land) has increased to about $1 500/acre ($3 700/ha) by 1976 [36].

In 1976, 10 men (3/shift) and a supervisor were required to operate the system. Maintenance includes checking and replacing sprinkler heads (which have a service life of 4 to 5 years).

FIGURE 7-26

OVERLAND FLOW TERRACES AT PARIS, TEXAS

TABLE 7-36

CONSTRUCTION AND OPERATING COSTS, PARIS, TEXAS [34], 1971

Construction costs, $/acre	
Site clearing, grading, and drainage ditches	362
Planting and fertilizing	108
Pipeline and sprinkler system	348
Engineering, surveying, and equipment	188
Total	1 007
Operating costs, ¢/1 000 gal	
Labor	3.2
Maintenance	1.4
Power	0.2
Miscellaneous	0.4
Subtotal	5.2
Revenue	0.4
Total	4.8

1 $/acre = $2.47/ha
1 ¢/1 000 gal = 0.264 ¢/m^3

7.12.6 Monitoring

In addition to the constituents listed in Table 7-35, the regular monitoring program includes analyses of temperature, pH, total residue, chlorides, sulfates, oil and grease, color, and dissolved oxygen. The total runoff flow is monitored continuously and samples are taken every 3 days for analyses.

7.12.7 Microbiology

Research on the soil microbiology at Paris has been reported by Vela [38] and Vela and Eubanks [39]. Populations of heterotrophic soil bacteria ranged from 10^6 to 10^8 organisms/gram of soil [39]. Large populations (1.5×10^5 to 7.4×10^5 organisms/gram of soil) of psychrophilic bacteria that are capable of actively growing at 2°C were also found, although the soil reaches this low temperature only a few days of the year [38]. This large microbial population sustains a high level of treatment even when low temperatures occur.

7.13 Other Case Studies

Many existing case studies of land treatment were necessarily excluded in this chapter. Lubbock, Texas, is an example of a slow rate system where a farmer is contracting for municipal effluent for irrigation on his land [40, 41]. At Tallahassee, Florida, research on nutrient removal has preceded full scale plans for treatment [42]. Case studies of operations at Quincy, Washington; and Manteca, California [43]; and Livermore, California [44], have also been reported.

For rapid infiltration the studies at Santee [45] and Whitter Narrows, California, [46] are available. The Calumet, Michigan, rapid infiltration system, probably the oldest rapid infiltration system in the United States, is being studied. Untreated, undisinfected wastewater at a flow of 1.2 Mgal/d (53 L/s) has been treated on 12 acres (4.8 ha) since 1887 [47, 48].

The most prominent overland flow system that is not included as a case study is at Melbourne, Australia. It has been operating successfully for several decades [49].

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Chapter 8

DESIGN EXAMPLE

8.1 Introduction

The design of a land treatment system is highly dependent on conditions, such as climate, soil, topography, and many others. As a consequence, no design example can be universal; however, the example should be illustrative of a design procedure in which the feasible alternatives are developed and assessed according to the methods in this design manual.

This presentation is adapted from a design example prepared by Mr. Sherwood Reed of USA CRREL for use in Corps of Engineers training courses. It is intended to present the development and evaluation of land treatment alternatives. As such, the design is not intended to be complete, since many components of a complete system, such as a transmission system and pumping stations, are omitted. The elimination of these components from this example will not allow a complete cost-effective comparison between land treatment and conventional treatment alternatives. Cost data used in this example were taken from sources described in Chapter 3.

The approach here is to present a statement of the problem and the data from which preliminary design alternatives, based on annual loadings, and process performance estimates are developed. A relative cost comparison between the developed process alternatives is presented from which the most cost-effective alternative can be chosen for final design. The final process design is based on more detailed analyses, including monthly loading distributions.

8.2 Statement of Problem

The problem is to provide adequate wastewater treatment for a community that has an existing primary treatment plant and surface water discharge. The recommended design must be the most cost-effective alternative and adapted to local conditions.

8.3 Design Data

8.3.1 Location

The problem area is located in the northeastern United States. The existing community has a present population of 70 000, with a 20 year design population of 90 000. The design wastewater flow is 10 Mgal/d (438 L/s). The existing treatment facilities for the community consist of a primary treatment plant with disinfection and sludge digestion. At present, the effluent is discharged to a river, and the digested sludge is applied to the land. The system was constructed in the early 1940s and is in very poor structural and mechanical condition, so it will be abandoned.

8.3.2 Climate

The climatic influences on land treatment are an important aspect in determining storage and length of the application season for slow rate and overland flow systems. The climatic data for the site was obtained from the National Oceanic and Atmospheric Administration's Climatic Summary of the United States for 20 years of record, and are presented in Table 8-1. For the worst year in 10, there are 142 days (mostly between November and March) in which the mean air temperature is less than 32°F (0°C). As indicated in Section 5.3, this necessitates storage for slow rate and overland flow systems. The annual precipitation of 50.2 in. (128 cm) occurs fairly uniformly throughout each month of the year. A total evapotranspiration of 25.1 in. (64 cm) occurs from late March to early November. The difference between precipitation and evapotranspiration, as given in Table 8-1, is used in computing monthly nitrogen and hydraulic balances.

TABLE 8-1

CLIMATIC DATA FOR THE WORST YEAR IN 10

Month	Temperature, °F		Days with mean temperature ≤32°F	Precipitation (Pr)		(ET) Evapo-transpiration, in.	Monthly net water excess (Pr - ET), in.
	Mean	Mean daily minimum		Total, in.	Days with mean ≥0.5 in.		
Nov	41.6	31.0	16	4.8	4	0.8	4.0
Dec	29.4	20.8	28	4.2	3	0	4.2
Jan	26.0	16.7	30	4.3	3	0	4.3
Feb	28.4	16.0	26	3.5	2	0	3.5
Mar	34.3	25.0	26	5.0	4	0.2	4.8
Apr	47.3	35.7	7	4.6	3	1.4	3.2
May	57.5	46.2	1	3.9	3	3.2	0.7
Jun	66.3	55.3	0	3.3	2	4.6	-1.3
Jul	72.0	60.7	0	3.8	2	5.4	-1.6
Aug	69.8	58.3	0	4.0	3	4.3	-0.3
Sep	62.2	51.4	1	4.2	2	3.3	0.9
Oct	51.8	40.4	7	4.6	3	1.9	2.7
Annual	48.9	38.1	142	50.2	34	25.1	25.1

1 °F = 1.8 x °C + 32
1 in. = 2.54 cm

8.3.3 Wastewater Characteristics

The characteristics of the wastewater are important in determining hydraulic and wastewater component application rates. To avoid nuisance conditions during winter storage, biological treatment in lagoons will be provided. The characteristics of the mostly domestic wastewater are presented in Table 8-2 along with the anticipated quality of the wastewater applied to the land after storage. Limited information on the quality of the Susanna River and native groundwater is also presented. The concentrations of trace metals are low, and mass application criteria for them are presented in Section 8.7.1.3.

TABLE 8-2

WATER QUALITY CHARACTERISTICS[a]

Parameter	Raw wastewater[b]	Wastewater to be applied to land[c]	Susanna River[d]	Groundwater[e]
BOD_5, mg/L	240	40	3.9	...
Suspended solids, mg/L	240	45	......	...
Total dissolved solids, mg/L	500	470	250	400
Total nitrogen as N, mg/L	40	28	6.0	...
Ammonia as N, mg/L	20	10	......	...
Organic as N, mg/L	20	4	......	...
Nitrate as N, mg/L	0	14	......	6
Total phosphorus as P, mg/L	10	8	......	...
Chloride, mg/L	40	37	20	35
Dissolved oxygen, mg/L	...	.....	5.0	...
CCE	...	.....	0.16	...
Total coliforms, MPN/100 mL	...	2 000	......	...

a. Trace metal concentrations are within the typical range for municipal wastewaters. Discussion is included in Section 8.7.1.3.

b. Data obtained from existing wastewater treatment plant records.

c. Assumed preapplication treatment by aerated lagoon plus storage.

d. Data obtained from State Water Quality Control Board.

e. Data obtained from USGS.

8.3.4 Discharge Limitations

The Susanna River, which is used as a public drinking water supply, has an average flow of 60 ft^3/s (1.7 m^3/s), a low flow of 44 ft^3/s (1.2 m^3/s), and a minimum dissolved oxygen concentration of 5.0 mg/L. Five miles (8 km) downstream from the existing wastewater treatment plant, the Susanna River flows into an estuary which is widely used for recreation. The State Water Quality Regulatory Agency has imposed the following limits on surface discharges (expressed as mg/L, 30 day averages):

BOD_5	4.0
Suspended solids	1
Phosphorus as P	0.1
Total Kjeldahl nitrogen	1
Nitrate-nitrogen	5
Total nitrogen as N	6
Maximum total chlorine residual	0.1
Total coliforms, organisms/100 mL	3

The groundwater aquifer is a potential drinking water source and fits Case I (see Section 5.1.1) so the EPA drinking water criteria for chemical and pesticide levels would therefore apply to discharges to groundwater. The most critical groundwater criterion would be a nitrate-nitrogen concentration not to exceed 10 mg/L (at site boundary).

8.3.5 Site Investigation

A preliminary investigation (see Section 3.5) of the lands adjacent to the community has determined that about 11 000 acres (4 450 ha) is available. The general topography of the area is shown in Figure 8-1. The area is bounded on the south by the Susanna River, which flows westerly. The existing water treatment plant and intake, and wastewater treatment plant and outfall are in the southwestern corner. The land increases in elevation from about 100 ft (30 m) above mean sea level near the Susanna River to a maximum elevation of 450 ft (136 m) at Clyde's Saddle. The surface slopes in the range of 1 to 4%, although a relatively flat area of 0 to 2% occurs in the eastern portion.

8.3.5.1 Soil Description

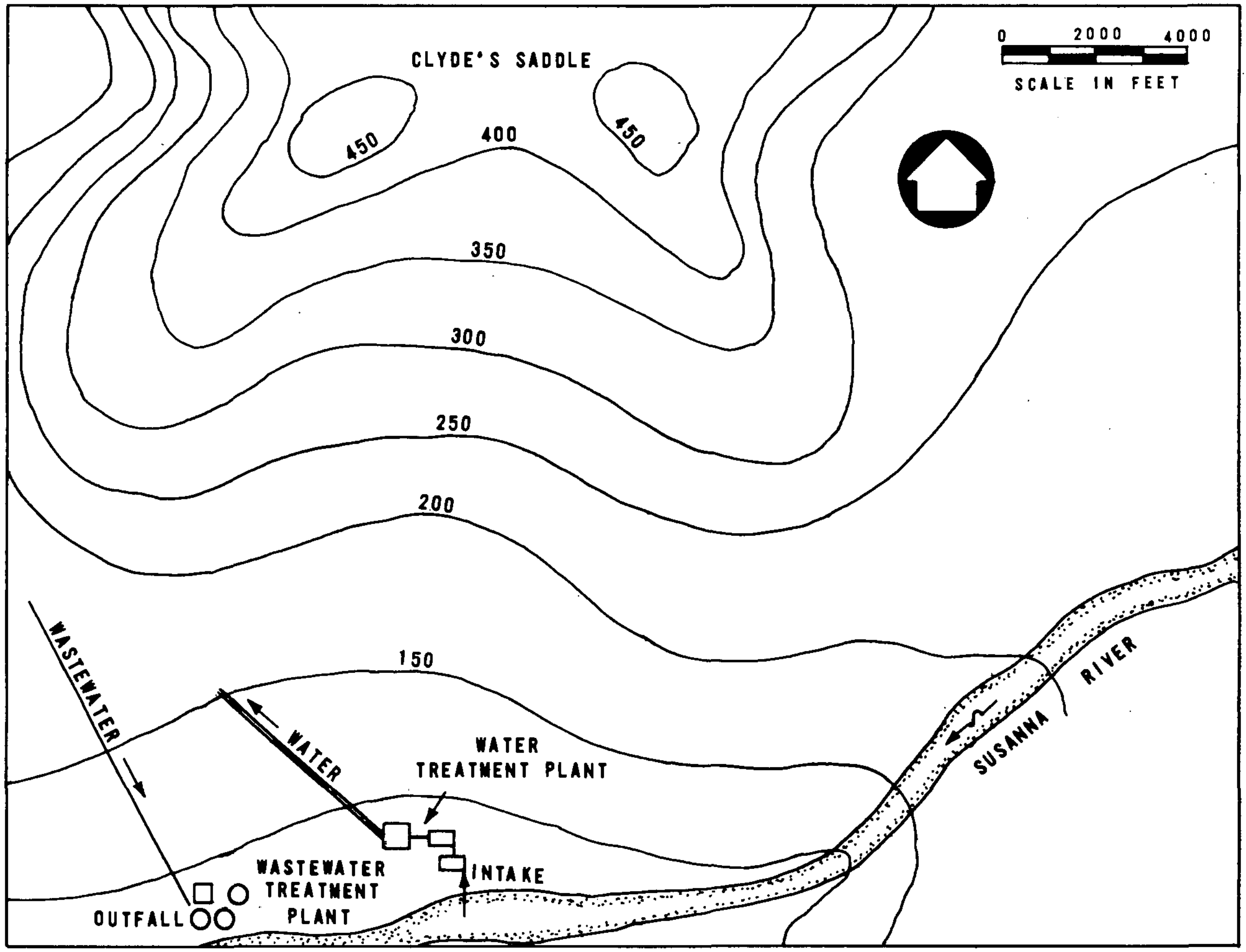
The type and location of agricultural soils as described in the SCS report for the study area include Hunt clay (HpG), Hanover loamy sand (Hn), and Bomoseen sandy clay loam (BsN), as shown in Figure 8-2.

The Hunt clay is a red-brown clay with a thin surface mantle of silt loam. Drainage is very poor with permeability of less than 0.2 in./h (0.5 cm/h). It is fair to good for grasses and legumes; poor for grain and seed crops and hardwood trees; and not suited for coniferous trees.

The Hanover loamy sand is a well-drained soil with a distance of 10 ft (3 m) or more to the water table. The permeability is at least 3 in./h

FIGURE 8-1

GENERAL TOPOGRAPHY

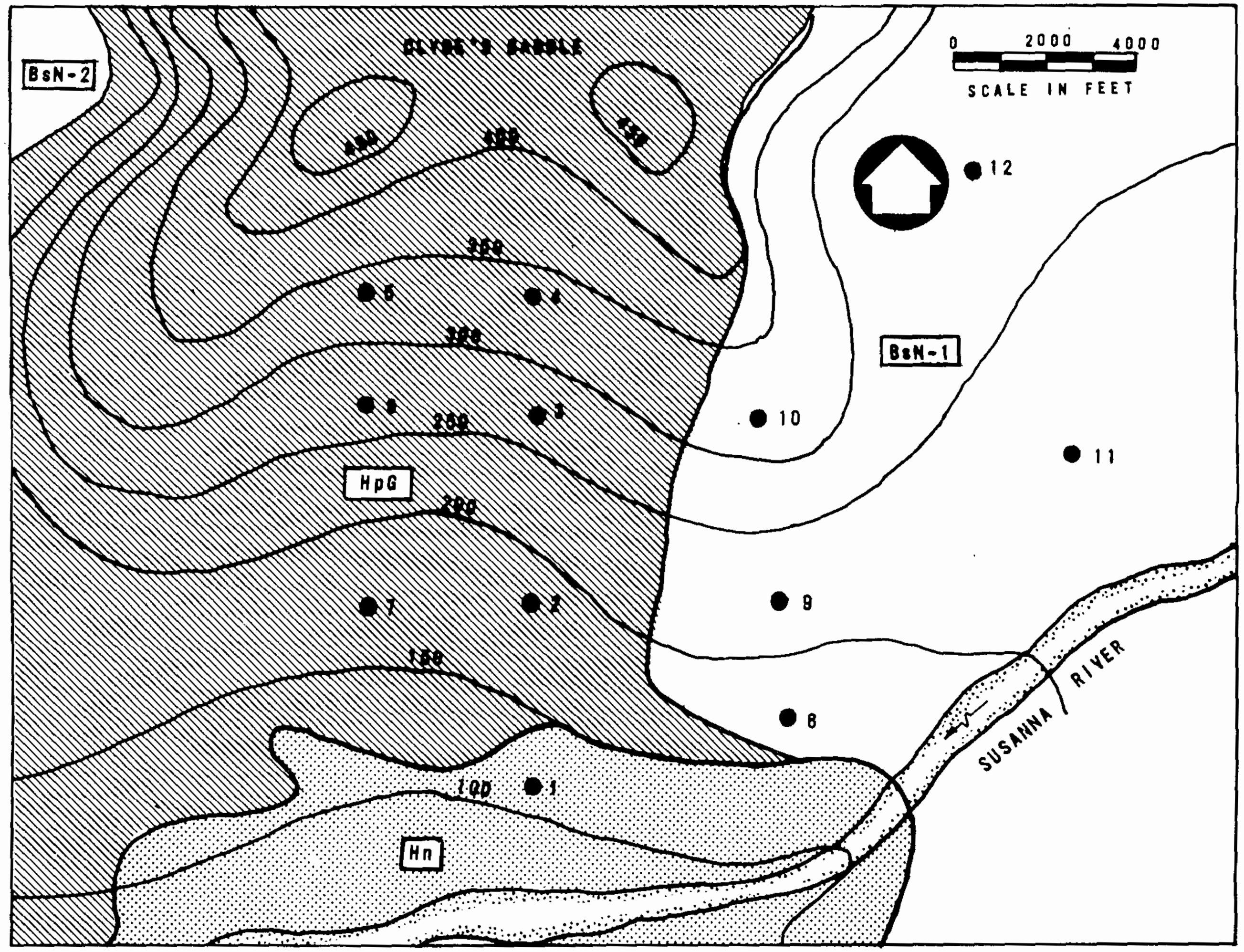


1 in. = 2.54 cm

1 ft = 0.305 m

FIGURE 8-2

AGRICULTURAL SOIL MAP



● = SOIL BORINGS, BsN = BOMOSEEN SANDY CLAY LOAM, Hn = HANOVER LOAMY SAND, HpG = HUNT CLAY.

(8 cm/h). It is fair to good for grain, seed crops, grasses, and legumes; good for hardwoods, and fair for coniferous trees.

The Bomoseen sandy clay loam is well drained, underlain by fine sands with 10 ft (3 m) or more to groundwater. The permeability is 0.6 in./h (1.5 cm/h). It is good for grain, seed, grass, legumes, and hardwoods; and fair for conifers. A descriptive summary of the soil types, including system suitability and available area, is presented in Table 8-3.

TABLE 8-3

AVAILABLE LAND AREAS BY SOIL TYPE[a]

Soil type	Soil description	Maximum slope, %	Permeability, in./h	System suitability	Available acres
BsN-1	Sandy clay loam	2	0.6	Slow rate	4 240
BsN-2	Sandy clay loam	3-4	0.6	Slow rate	330
Hn	Loamy sand	3	3	Rapid infiltration and slow rate	1 340
HpG-1[b]	Clay	2	<0.2	Overland flow	1 230
HpG-2[c]	Clay	3-4	<0.2	Overland flow	4 020
				Total	11 160

a. Data from SCS report.

b. Area between 100 and 200 ft contours (half clear, half brush, and woodland).

c. Area above 200 ft contour (all brush and woodland).

1 acre = 0.405 ha
1 ft = 0.305 m

The general soils evaluation shows that within the study area, there exist soil types that appear to be suitable for all three land treatment processes. Further assessment of their suitability requires additional information on the subsurface geology.

8.3.5.2 Soil Borings

Well logs or other information on the soil profile were not available. Consequently, twelve preliminary soil borings were made as shown in Figure 8-2 to confirm the SCS soil map. The results from the boring logs show that groundwater was encountered at the single drill hole (No. 1) and that the depth to bedrock varied from a minimum of 20 ft (6 m) at borings Nos. 4 and 5 to a maximum of 70 to 80 ft (21 to 24 m) at borings Nos. 1 and 8. The underlying geology is a mixture of sands and gravels with clay and fine sand occurring at various depths without hardpan layers. The borings at the lowest elevations have the greatest depth to bedrock, with decreasing soil depth as the elevation increases. The subsoil geology has equal to or greater permeability than the upper soil horizons.

8.3.5.3 Vegetative Cover

The vegetative cover is important as an indicator of the growth conditions for a soil type and as a factor in determining costs of clearing and other site preparation. As shown in Figure 8-3, in the eastern part of the study area, there are open lands and native grasses on the Bomoseen sandy clay loam. In the southwest corner of the study area, there is previously cleared land on Hanover loamy sand and Hunt clay. In the rest of the study area (proceeding northward towards Clyde's Saddle), there is a wooded area of brush and trees, mostly underlain with Hunt clay.

8.4 Process Alternatives

8.4.1 Slow Rate System

The initial determination of the required field area is made using the annual water balance:

$$Pr + L_w = ET + W_p + R \qquad (8\text{-}1)$$

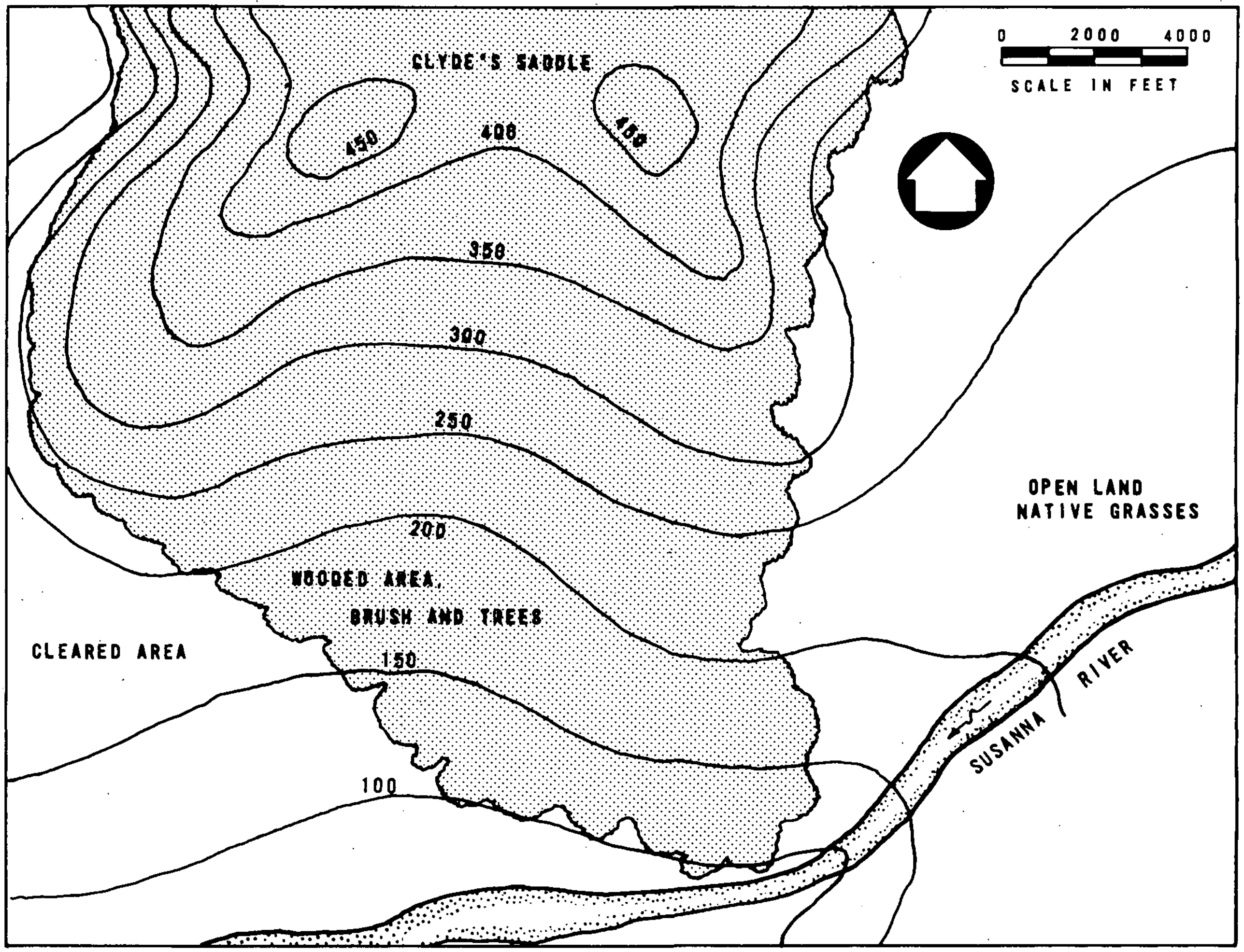
where Pr = precipitation, ft/yr (cm/yr)
L_w = wastewater hydraulic loading, ft/yr (cm/yr)
ET = evapotranspiration, ft/yr (cm/yr)
W_p = percolating water, ft/yr (cm/yr)
R = runoff, ft/yr (cm/yr)

In this case, runoff of applied water will be retained and thus will be considered negligible. The relationship between precipitation and evapotranspiration is given in Table 8-1. Precipitation exceeds evapotranspiration by 2.1 ft/yr (64 cm/yr). Wastewater applications are scheduled for periods when the mean air temperature is above 32°F (0°C), approximately from March 25 to November 3 (Table 8-1). This 32 week application season will avoid extreme temperatures and frozen ground conditions, will ensure some crop response, and necessitate 20 weeks of storage within the design year. The percolating water can be estimated from Figure 3-3 using the permeability value of 0.6 in./h (1.5 cm/h) for the Bomoseen sandy clay loam. A conservative rate of about 3.5 in./wk (8.9 cm/wk) is chosen because crop production is planned. This value is multiplied by the 32 week season to determine the annual loading.

$$(3.5 \text{ in./wk})\ (32 \text{ wk/yr}) \div 12 \text{ in./ft} = 9.3 \text{ ft/yr}$$

FIGURE 8-3

EXISTING VEGETATIVE COVER



The total liquid loading would be reduced by the 2.1 ft/yr (64 cm/yr) of excess precipitation (Table 8-1) for a resultant loading of 7.2 ft/yr (216 cm/yr). The required field area is then calculated to be:

$$F = \frac{3.06\ Q}{L_w} \qquad (8\text{-}2)$$

where F = field area, acres
Q = annual wastewater flow, Mgal/yr
L_w = wastewater loading, ft/yr

$$F = \frac{3.06\ (10)(365)}{7.2}$$

$$F = 1\ 551 \text{ acres}$$

$$\text{say} = 1\ 600 \text{ acres}$$

An examination of the soil classification data and soil boring logs shows that the soils classified as BsN and Hn would be hydraulically suitable for slow rate systems. These soils, as located along the Susanna River and east of Clyde's Saddle (Figure 8-2), comprise 5 910 acres (2 387 ha) (Table 8-3) of suitable land. Thus, it appears that the slow rate process would be potentially feasible for this location and should be investigated further using a nitrogen balance (see Section 8.7.1) to determine if groundwater criteria can be satisfied.

8.4.2 Rapid Infiltration System

The determining factor in hydraulic application is the soil permeability. The Hanover loamy sand has a permeability of at least 3 in./h (8 cm/h), so a wastewater application rate of 25 in./wk (64 cm/wk) is estimated from Figure 3-3. Based on a 52 wk/yr operation, this results in an annual application rate of 110 ft/yr (33.5 m/yr). The wetted field area can be estimated in the same manner as a slow rate system, giving a required wetted field area of 100 acres (45 ha) as follows:

$$F = (3.06)(3\ 650)/110 = 100 \text{ acres}$$

The alternate flooding and drying cycle can be accomplished by having multiple basins, with a set of basins being flooded for 4 days to promote good denitrification followed by an 8 day drying period. This operational schedule results in approximately one-third of the field area (8 or 9 basins) being flooded and two-thirds (16 or 17 basins) being rested at any given time. Approximately 1 350 acres (614 ha) of suitable soil exists, so this alternative should be investigated further to determine if it can satisfy water quality requirements.

8.4.3 Overland Flow System

Overland flow systems require slopes from 2 to 8% on relatively impermeable soils. The almost continuously wet field conditions are not conducive to normal forest or agricultural cover, but usually require special grasses. Nitrogen removal is dependent on complex biochemical responses in addition to crop uptake. These biochemical responses are temperature dependent so there are climatic constraints on overland flow systems.

Since terraces having appropriate slopes and dimensions can be formed and proper soils are available, it should be possible to apply approximately 8 in./wk (20.3 cm/wk) of lagoon effluent during the summer growing season and approximately half that amount, 4 in./wk (10.2 cm/wk), in the spring and fall (Section 5.1.4.1). Since winter storage requires some form of treatment oxidized wastewater will be applied to the slopes. Operational experience will dictate the degree of oxidation required; it may be possible to shut down all of the aerators during the summer. The application schedule and storage requirements are presented in Table 8-4, using the number of days with mean temperature less than 32°F (0°C). The results give a design application of 17.8 ft/yr (5.4 m) and a storage requirement of approximately 142 days.

TABLE 8-4

DETERMINATION OF OVERLAND FLOW APPLICATION SCHEDULE BASED ON CLIMATIC DATA[a]

Time period	Total No. of days in time period	No. of days with mean temperature ≤32°F	Application period, d	Application schedule		Wastewater applied, in.
				No. of wks	in./wk	
Nov 16 - Apr 20	156	133	23	3.3	4	13.2
Apr 21 - Apr 30	10	0	10	1.4	4	5.6
May 1 - Sep 30	153	2	151	21.6	8	172.8
Oct 1 - Nov 15	46	7	39	5.6	4	22.4
Total	365	142	223	31.9		214.0

a. Based on worst year in 10, from Table 8-1.

1 in. = 2.54 cm

The required field area is 627 acres (254 ha), as computed by the same method used for the slow rate system:

$$F = \frac{(3.06)(3\ 650)}{17.8\ \text{ft/yr}} = 627\ \text{acres}$$

An examination of the soils data indicates that the area north of the water treatment plant on the lower slopes of Clyde's Saddle will probably be suitable. Between elevation 100 ft (30 m) and elevation 200 ft (61 m) there is at least 1 230 acres (497 ha) of soils suitable for constructing an overland flow system, hence overland flow should also be considered further.

8.5 Preliminary Performance Estimate

8.5.1 Slow Rate System

The capability of the slow rate system to meet Case 1 groundwater standards was determined by assuming a 10 mg/L design concentration for nitrate-nitrogen. Removal of phosphorus is excellent, with expected removals greater than 99%, even though a phosphorus limit does not exist for drinking water. The concentrations of BOD and suspended solids in the percolate should be less than 1 to 2 mg/L, and pathogenic organism removal by the Bomoseen sandy clay loam should be complete within the upper 2 ft (0.6 m) of the soil.

The limiting design criteria is nitrogen. Based on existing system performance (see Chapter 7), the concentration of nitrate-nitrogen in the slow rate system percolate will be better than the 10 mg/L design value. In addition, the design for 10 mg/L percolate nitrate-nitrogen concentration is conservative, because significant dilution of the percolate nitrate-nitrogen will most likely occur as the percolate water mixes with the underlying native groundwater. Also, design flows are assumed for 1990, so initial applications will be less, and subsequent nitrogen performance better. Seasonal variations in performance should be satisfied by variable monthly applications. The monthly application criteria will be developed if slow rate systems are most cost effective.

8.5.2 Rapid Infiltration System

The treatment performance of a rapid infiltration system should be assessed because design applications are usually determined by hydraulic considerations rather than wastewater constituent applications. For this example, the performance should be evaluated for groundwater discharge, as well as surface discharge. The soil permeability and subsurface geology are both suitable for groundwater discharge.

The total nitrogen applied to the land in a rapid infiltration system can be estimated from Equation 5-3:

$$\begin{aligned} L_n &= 2.7\ C_n\ L_w \\ &= (2.7)(28\ \text{mg/L})(110\ \text{ft/yr}) = 8\ 316\ \text{lb/acre·yr} \\ \text{say} &= 8\ 400\ \text{lb/acre·yr}\ (9\ 410\ \text{kg/ha·yr}) \end{aligned}$$

The nitrogen is rapidly converted from the applied organics and ammonium form to nitrate-nitrogen. The principal removal mechanism is biological denitrification of nitrate-nitrogen, although volatilization and crop uptake (if vegetation is used) can add to the estimated 50% total removal. The estimated percolate nitrogen amounts to 4 200 lb/acre·yr (4 700 kg/ha·yr) and will move with a percolate volume of 112 ft/yr (34 m/yr) [110 ft (33.5 m) applied wastewater and 2 ft (0.6 m) net precipitation]. The average concentration of nitrate-nitrogen would be approximately 4 200/(2.7)(112) = 14 mg/L. This concentration is greater than the assumed design criteria of 10 mg/L total nitrogen for percolate and greater than the 6 mg/L total nitrogen criteria for river discharge. Although the other discharge criteria, i.e., phosphorus, BOD, suspended solids, and pathogens, would be adequately satisfied, rapid infiltration, by itself, will not satisfy the nitrogen design criteria. Further investigation would be necessary to determine the degree of mixing and dispersion that would occur in the groundwater under the site. For this example, rapid infiltration is not discussed further, except in combination with overland flow.

8.5.3 Overland Flow System

The average total nitrogen concentration of an overland flow runoff is expected to be about 3 mg/L (see Table 2-3). Existing overland flow systems have shown that total nitrogen removals (mass basis) have varied from 75 to 90% for systems operating with an application period of 52 wk/yr. The principal nitrogen removal mechanisms are crop uptake and nitrification-denitrification on the soil surface; these mechanisms are adversely affected by low winter temperatures. Therefore, it would be reasonable to expect a 90% nitrogen removal for a system operating with an application period of 32 wk/yr. For the estimated hydraulic application of 17.8 ft/yr (5.4 m), the total applied nitrogen (from Equation 5-3) is (2.7)(28)(17.8) = 1 346 lb/acre·yr (1 509 kg/ha·yr). With a 90% removal, 135 lb/acre·yr (151 kg/ha·yr) is collected in the runoff. The final concentration is dependent upon the water balance, so inputs and outputs are given:

	ft/yr	(m/yr)
Applied wastewater	17.8	(5.4)
Percolate loss[a]	-1.4	(0.4)
Precipitation-evapotranspiration[b]	+1.7	(0.5)
Net runoff	18.1	(5.5)

a. Assume 8% loss for HpG soil.

b. Estimate for application period.

The design runoff nitrogen is estimated to be all in the nitrate form. From a mass of 135 lb/acre·yr (151 kg/ha·yr) and a volume of 18.1 ft/yr

(4.4 m/yr), the concentration is determined as follows: 135/(2.7)(18.1) = 2.8 mg/L. This concentration meets both surface and subsurface nitrogen criteria.

Phosphorus removal, however, is usually 50% since the wastewater contact with the soil is relatively limited (see Section 5.1.4.5). At the design application rate of 17.8 ft/yr (5.4 m/yr), the applied phosphorus is $P = 2.7\ CL_w = (2.7)(8)(17.8) = 385$ lb/acre·yr (431 kg/ha·yr), of which 192 lb/acre·yr (215 kg/ha·yr) can be expected to run off. This would correspond to a runoff concentration of 192/(2.7)(18.1) = 3.9 mg/L. The phosphorus concentration is greater than the river discharge standard of 0.1 mg/L, so overland flow alone would not be allowed for surface discharge. In addition, overland flow alone would not meet discharge criteria for suspended solids.

To make overland flow a feasible alternative for this example, it will be combined in series with rapid infiltration. The combined system would depend on the former for nitrogen and BOD removal and on the latter for suspended solids, microorganisms, and phosphorus removal. The rapid infiltration basins would be designed for the 18.1 ft/yr seasonal net runoff from the overland flow slopes:

Net overland flow runoff = (18.1 ft/yr)(627 acres) = 11 350 acre-ft/season
RI application = 11 350 acre-ft/season ÷ 32 wk/season = 355 acre-ft/wk
For a 100 acre basin area,
weekly application rate = 355 ÷ 100 x 12 = 42.6 in./wk
From Figure 3-3, the maximum
weekly application = 50-70 in./wk; thus, 42.6 in./wk is satisfactory.

The hydraulic capacity of the soil would govern design rather than the loadings of wastewater constituents. Discharge would be to groundwater, and would eventually appear as a seep to the river (nonpoint discharge).

A summary of the preliminary assessments is presented in Table 8-5. The slow rate and the combined overland flow and rapid infiltration processes are capable of providing satisfactory wastewater treatment with the tabulated application rates and land areas. A cost estimate should be determined at this time to decide which option provides the most cost-effective treatment and should be considered for detailed design. In addition, slow rate systems have three distribution options that should be evaluated on a cost-effectiveness basis.

TABLE 8-5

SUMMARY OF DESIGN INFORMATION
FOR TREATMENT ALTERNATIVES

Treatment Alternatives	Design flow, Mgal/d	Annual Wastewater application: Period, wk	Annual Wastewater application: Total, ft	Avg weekly application rate, in.	Application area, acres	Length of storage, d	Storage area, acres[a]	Treatment lagoon, acres[b]	Total area acres[c]
Slow rate (flood, center pivot or solid set)	10	32	7.2	2.7	1 600	140	360	15	2 170
Overland flow followed by rapid infiltration									
Overland flow	10	32	17.8	6.7	627	140	360	15	1 100
Rapid infiltration	10	32	113.5	42.6	100	...	...	..	110

a. Based on 10 Mgal/d flow and 12 ft working depth (see Section 8.7.1.5).

b. Based on 7 days detention at 10 Mgal/d, 15 ft working depth.

c. Includes 10% for roads, buildings, and miscellaneous.

1 Mgal/d = 43.8 L/s
1 ft = 0.305 m
1 in. = 2.54 cm
1 acre = 0.405 ha

8.6 Cost Comparison

The procedures to calculate capital, and operation and maintenance costs have been published [1]. Tabulations of the results are presented in Table 8-6 to show differences due to type of treatment system and distribution system. The cost comparison is made solely to compare land treatment systems. Each system will usually contain a collection system, collection pumping, preapplication treatment, and administrative facilities; these are not included in the comparison since the added capital and operation and maintenance costs should be identical. Additional comparison to a conventional treatment alternative would require inclusion of all costs before comparisons with total treatment system would be made.

The total costs in Table 8-6 include unlined storage, site clearing, site leveling, distribution system, distribution pumping (Alternatives 1-4); tailwater return (Alternative 1); overland flow terrace construction, runoff collection, and open channel transmission from the overland flow to the rapid infiltration site (Alternative 4).

A slow rate system, utilizing center pivot distribution, has the lowest relative cost for this design example. The costs generated are not discussed further since their purpose was only to provide a relative cost effectiveness for the general conditions as described in Table 8-6, and as developed in the text (Sections 8.4 and 8.5). The slow rate, center pivot alternative will be further developed to provide the preliminary system design.

TABLE 8-6

RELATIVE COST COMPARISON, DESIGN EXAMPLE ALTERNATIVES[a]

Alternative	System type	Land area, acres[b]	Total capital cost, $	Amortized capital cost, $/yr	Operation and maintenance cost, $/yr	Total cost, $/yr	Municipal cost, $/yr[c]
1	Slow rate, flood	2 170	8 583 770	756 230	202 750	958 980	391 810
2	Slow rate, center pivot	2 170	8 232 770	725 310	205 720	931 030	387 050
3	Slow rate, solid set	2 170	10 624 120	935 990	186 520	1 122 510	420 520
4	Overland flow and rapid infiltration	1 210	8 495 500	748 450	214 000	962 450	401 110

a. Based on unique or variable land treatment components. Items that are common to and have equal costs in all alternatives are not included.

b. Actual area is determined in the final layout.

c. Computed as 25% capital and 100% operation and maintenance costs.

1 acre = 0.405 ha

8.7 Process Design

In this particular example, the slow rate, center pivot alternative was found to be more cost effective than the treatment system alternative of overland flow followed by rapid infiltration; under other circumstances the reverse may be true. For purposes of illustrating the required design procedures, both treatment system alternatives will be described.

8.7.1 Slow Rate

The development of the slow rate process design includes an assessment of (1) the hydraulic loading criteria, (2) the annual and monthly nitrogen loadings, and (3) phosphorus and trace metal loading criteria. Also included is a discussion of (4) preapplication treatment, (5) storage design criteria, and (6) distribution system criteria.

8.7.1.1 Hydraulic Loading

For slow rate systems, net runoff can be assumed to be negligible. From Table 8-1, the total annual precipitation (Pr) of 50.2 in./yr (128 cm/yr) minus the total annual evapotranspiration (ET) of 25.1 in./yr (64 cm/yr) yields an annual net water excess of 25.1 in./yr (64 cm/yr) or 2.1 ft/yr (0.6 m/yr). Thus Equation 8-1 becomes:

$$W_p = L_w + Pr - ET$$

or

$$W_p = L_w + 2.1 \text{ ft/yr}$$

The amount of percolating water (W_p) resulting from the applied effluent (L_w) has a significant effect on the allowable nitrogen loading (L_n), as is illustrated in the following section.

8.7.1.2 Nitrogen Loading

The annual nitrogen loading can be estimated from procedures in Section 5.1.2.2, as described below:

The annual nitrogen balance, using Equation 5-2, is:

$$L_n = U + D + 2.7\ W_p\ C_p \qquad (5\text{-}2)$$

where L_n = wastewater nitrogen loading, lb/acre·yr (kg/ha·yr)
U = crop N uptake = 325 lb/acre·yr (364 kg/ha·yr) for Reed canary grass (Table 5-2)
D = denitrification = 0.2 L_n (assume denitrification to be 20% of applied nitrogen)
W_p = percolating water = L_w + 2.1 ft/yr
C_p = design percolate N concentration = 10.0 mg/L

Therefore,

$$L_n = 325 + 0.2\ L_n + (2.7)(L_w + 2.1)(10),$$

and the relationship between the nitrogen loading and the hydraulic loading (from Equation 5-3) is:

$$L_n = 2.7\ C_n\ L_w \qquad (5\text{-}3)$$

where C_n = applied nitrogen concentration, mg/L
L_w = wastewater hydraulic loading, ft/yr

Therefore, at C_n = 28 mg/L (from Table 8-2),

$$L_n = 75.6\ L_w$$

or

$$L_w = 0.013\ L_n$$

Now with two equations and two unknowns, the nitrogen balance equation can be solved:

$$\begin{aligned} L_n &= 325 + 0.2\ L_n + (2.7)(L_w + 2.1)(10) \\ L_n &= 325 + 0.2\ L_n + (2.7)[(0.013\ L_n) + 2.1](10) \\ L_n &= 325 + 0.2\ L_n + 0.351\ L_n + 56.7 \\ 0.45\ L_n &= 381.7 \\ L_n &= 848\ \text{lb/acre·yr} \end{aligned}$$

The complete solution for a design percolate nitrogen concentration of 10 mg/L is as follows:

1. Wastewater nitrogen loading = L_n = 848 lb/acre·yr
2. Wastewater hydraulic loading = L_w = 0.013 L_n = 0.013 (848) = 11.0 ft/yr
3. Percolating water = W_p = L_w + 2.1 = 13.1 ft/yr
4. Denitrification = D = 0.2 L_n = 170 lb/acre·yr
5. Percolate nitrogen loading = P_n = 2.7 C_pW_p = 2.7(10)(13.1)

   = 354 lb/acre·yr

6. Required field area = $F = \frac{3.06(365)\ Q}{L_w} = \frac{1\ 118\ (10)}{11.0}$ = 1 015 acres

The slow rate system design is based on maximum nitrogen uptake by the vegetation. For this design, a cool season forage grass, such as Reed canary grass, is chosen since it will provide an estimated nitrogen removal of 325 lb/acre·yr (364 kg/ha·yr) and provide a year-round cover for maximum infiltration, minimal soil erosion after harvest (in contrast to an annual crop), and nitrogen response at the beginning and end of the growing season as a result of an established root system.

The procedure to determine the monthly nitrogen balance accounts for monthly climatic influences. Thus, greater wastewater applications occur when more nitrogen is needed by the vegetation and greater microbial activity occurs.

In order to determine the optimal system design loadings, monthly wastewater applications (values for L_w) were chosen (by trial and error) to the nearest inch, so that the percolate nitrogen concentration (C_p) was less than, or equal to 10.0 mg/L. For the first cut estimate of monthly values for L_w, divide the annual wastewater hydraulic loading (from above) by the number of months in the application season. For this example, the first trial value for L_w for the months of April through October would be 11 ft/yr x 12 in./ft ÷ 7 mo/yr = 19 in./mo. For the cool weather months of March and November (at the beginning and end of the growing season), a wastewater hydraulic loading of 1 in./month was assumed.

The monthly nitrogen loading may be calculated using the following equation:

$$L_n = 0.227\ C_n L_w \tag{8-3}$$

where L_n = wastewater nitrogen loading, lb/acre·month (kg/ha·month)
C_n = applied nitrogen concentration, mg/L
L_w = wastewater hydraulic loading, in./month (cm/month)

The estimated denitrification is calculated as 20% of the total nitrogen applied resulting in an annual loss of 147 lb/acre·yr (165 kg/ha·yr). Crop nitrogen uptake was estimated at 325 lb/acre·yr (364 kg/ha·yr) and distributed monthly by the monthly fraction of the total evapotranspiration occurring during the growing season, which can be estimated to be the months of April through October. This assumes that plants utilize nitrogen and water at similar rates. Whenever possible, estimation of the monthly variation of crop nitrogen uptake should be refined by consulting the local agricultural extension service. Percolate nitrogen (P_n) was computed as the difference between application and denitrification plus crop nitrogen uptake. The percolate nitrogen concentration (C_p) was computed from monthly percolate nitrogen (P_n)(lb/acre) and total percolate volume (W_p). To calculate the monthly percolate nitrogen concentration (C_p), the following equation can be used:

$$P_n = 0.227\ C_p W_p \tag{8-4}$$

where P_n = percolate nitrogen loading, lb/acre·month (kg/ha·month)
C_p = percolate nitrogen concentration, mg/L (10 mg/L limit)
W_p = percolating water, in./month (cm/month)

The result of the monthly nitrogen balance are presented in Table 8-7.

TABLE 8-7

MONTHLY DESIGN NITROGEN BALANCE, SLOW RATE SYSTEM

						Leaching		
Month	(Pr - ET) Net monthly excess water, in.[a]	(L_w) Applied wastewater, in.[b]	(L_n) Wastewater nitrogen loading, lb/acre[c]	(D) Denitrification, lb/acre[d]	(U) Crop N uptake, lb/acre	(W_p) Percolate water, in.[e]	(P_n) Percolate nitrogen, lb/acre[f]	(C_p) Percolate nitrogen concentration, mg/L[g]
Nov	4.0	1[h]	6	1	...	5.0	5	4.4
Dec	4.2	0	...	...	...	4.2	...	...
Jan	4.3	0	...	...	...	4.3	...	...
Feb	3.5	0	...	...	...	3.5	...	...
Mar	4.8	1[h]	6	1	...	5.8	5	3.8
Apr	3.2	9	57	11	19	12.2	27	9.7
May	0.7	15	95	19	43	15.7	33	9.3
Jun	-1.3	20	127	25	62	18.7	40	9.4
Jul	-1.6	24	153	31	73	22.4	49	9.6
Aug	-0.3	20	127	25	58	19.7	44	9.8
Sep	0.9	16	102	20	44	16.9	38	9.9
Oct	2.7	11	70	14	26	13.7	30	9.6
Annual	25.1	117	743	147	325	142.1	271	8.4[i]

a. From Table 8-1.

b. Highest possible volume (to nearest in.) without exceeding 10 mg/L in percolate (found after a series of trials).

c. $L_n = 0.227\ C_n L_w$; C_n = 28 mg/L.

d. $D = 0.2\ L_n$.

e. $W_p = L_w + Pr - ET$.

f. $P_n = L_n - D - U$.

g. $P_n = 0.227\ C_p W_p$; $C_p = P_n/(0.227)(W_p)$

h. Assume 1 in./wk application at beginning and end of growing season.

i. Computed as the average of the monthly values. Conservative since nonapplication season rainwater percolation and groundwater dilution will reduce yearly average total percolate nitrogen.

1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha

The monthly design nitrogen balance results in an annual wastewater application of 117 in./yr or 9.7 ft/yr (3.0 m/yr), which is slightly less than 11.0 ft/yr (3.3 m/yr) in the previous annual assessment. The wetted field area should be adjusted to account for the lesser application, so the required application area is:

$$F = \frac{3.06(365)\ Q}{L_w} = \frac{1\ 118\ (10)}{9.7} = 1\ 150 \text{ acres } (466 \text{ ha})$$

rather than 1 015 acres (410 ha).

The permeability for the soil surface (infiltration rate) and subsoil can be evaluated on the basis of the maximum monthly liquid application to the soil surface. During July, the wastewater application of 24 in. (60 cm) and mean precipitation of 3.8 in. (10 cm) and evapotranspiration of 5.4 in. (14 cm) add up to a monthly infiltration of 22.4 in./mo (57 cm/mo). On a weekly basis, the maximum hydraulic loading would be about 5.6 in./wk (14 cm/wk). For the Bomoseen sandy clay loam with a soil permeability of 0.6 in./h (1.5 cm/h) and a perennial forage cover, the total application could be infiltrated within about 9 hours. This represents less than 6% of the total time in a week, so an application schedule based on equipment capacity can be determined.

8.7.1.3 Other Mass Loadings

The mass application of phosphorus and trace metals to the site can be assessed to determine if they would limit total wastewater applications over the 20 year design life of the project. The phosphorus criterion is based on the total mass application, phosphorus removal in vegetation, and soil retention by adsorption and precipitation. The trace metal criterion is based on mass application over the life of the project for elements retained in the soil, and applied concentrations for elements that are not retained.

At the annual application rate of 9.7 ft/yr (3.0 m/yr) and the total phosphorus concentration of 8 mg/L (as P), the annual application is: (2.7)(8)(9.7) = 210 lb/acre·yr (235 kg/ha·yr). The Reed canary grass will remove 40 lb/acre·yr (45 kg/ha·yr) (Table 8-1) during harvest, so the net application to the soil is 170 lb/acre·yr (190 kg/ha·yr). The sandy clay loam soil will have excellent removal of phosphorus as a result of the clay content. The 3 400 lb/acre (3 811 kg/ha) phosphorus application over 20 years can be completely adsorbed in the top 22 in. (56 cm) of the soil with a 5 day adsorption capacity of 50 mg of phosphorus per 100 g of soil and a bulk density of 1.3 g/cm^3. This is a conservative estimate since it has been estimated that the phosphorus retention (including chemical precipitation) may be at least double that measured by the 5 day adsorption test.

The mass application of trace metals should not pose any further limitations at the proposed site. For the applied concentrations, the mass applications are below the recommended maximum for use on agricultural soils (Table 8-8).

TABLE 8-8

TRACE METALS IN SLOW RATE DESIGN EXAMPLE

Element	Concentration in raw wastewater, mg/L	Concentration applied to land, mg/L	Mass application, lb/acre[a]	Maximum loading criteria, lb/acre[b]	EPA Drinking Water Standard, mg/L[c]
Cadmium	0.01	0.008	4	8	0.01
Chromium	0.03	0.02	10	82	0.05
Copper	0.22	0.10	52	164	1.0
Lead	0.01	0.005	2.6	4 080	0.05
Mercury	0.001	0.001	0.5	.....[d]	0.002
Nickel	0.03	0.02	10	164	No standard
Silver	0.001	0.001	0.5	.....[e]	0.05
Zinc	0.31	0.20	105	1 640	5.0

a. On the basis of 9.7 ft/yr and 20 yr life. Example: Cd = (2.7)(0.008)(9.7)(20) = 4.

b. From Table 5-4.

c. From Table 3-4.

d. No suggested limit since retention is very high and applied concentrations are below drinking water standard.

e. No limit since most applications are too small in comparison with drinking water standard.

1 lb/acre= 1.13 kg/ha

8.7.1.4 Preapplication Treatment

Preapplication treatment is included as a unit process in this design example as a means of odor control for the 20 week winter storage period and for a reduction of suspended solids to minimize clogging in the distribution system. The treatment removals of nitrogen, phosphorus, and other wastewater organic and inorganic constituents are not dependent on a specified level of treatment before application to the land, so partial oxidation of wastewater organics should be adequate. The long-term storage pond should provide for additional wastewater treatment during the retention time of up to 20 weeks, so preapplication treatment by aerated lagoons to reduce BOD down to a concentration of 60 mg/L should be adequate. Other processes exist to oxidize wastewater organics before application to the land, but for the purposes of this example, aerated lagoons are considered the most cost-effective alternative. A further reduction in BOD will occur during the 20 week storage period.

Detailed design procedures for aerated lagoons are covered elsewhere [2] and will not be repeated herein. Experience with lagoons indicates that multiple cells offer operating and maintenance advantages. Since the site is in a northern climate, extra time must be provided to compensate for the slower reaction rates in the winter. A 4 cell system, with parallel units, designed for a total detention of 6 days should provide the desired level of treatment during the winter months at this site.

8.7.1.5 Storage Lagoons

The required volume, area, and depth for the storage lagoon can be calculated in a manner similar to that used for the aerated lagoon. The climatic data were used to calculate a 20 week storage, which resulted in a total storage capacity of 1.4×10^9 gal (5.3×10^9 L). This is equivalent to a volume of 1.87×10^8 ft^3 (5.3×10^6 m^3); so 360 surface acres (164 ha) is required for storage at a depth of 12 ft (3.7 m). The final design should allow an additional 3 ft (0.9 m) for freeboard, for a 15 ft (4.6 m) total depth in storage. Further, the storage lagoon should be divided into multiple cells to reduce wind fetch and wave generation. The final design would consist of 4 basins at 90 acres each or 3 basins at 120 acres each, depending on final topography available for siting and construction.

8.7.1.6 Location of Treatment and Storage Lagoons

The open land to the east of the tree line in Figure 8-3 was identified as potentially feasible for a slow rate system. There is sufficient land for location of the treatment and storage lagoons, as well as the advantage of having all components of the system in proximity. However, the soil characteristics would require lining of the lagoons to control seepage.

Further examination of the topography and soils data indicates significant advantages exist for a location in the general vicinity of borings No. 2 and No. 7, as shown in Figure 8-2. Such a location would permit gravity flow of the raw wastewater to the highest possible elevation shown on the map, and the impermeable surface soils in this area could be stripped and used to line the treatment and storage lagoons. It would require clearing of approximately 375 acres (170 ha) of brush and trees from the site.

8.7.1.7 Slow Rate Distribution System

The design of a mechanical distribution system is usually determined by the equipment available from various manufacturers. However, it is desirable to know the number and size of units so that an estimate of unsprayed areas between wetted circles can be made. The costs (Table 8-6) were estimated on the basis of a maximum sprayed area of 134 acres (61 ha) per sprinkler unit, with the rotating booms typically available as multiples of 100 foot lengths.

To cover the required field area of 1 150 acres (466 ha), 9 units of 134 acres (61 ha) each will be used. Each unit has a rotating boom radius of about 1 300 ft (397 m). The total area required would be 15 to 20% greater, depending on geometric layout of the circles and degree of end area coverage from manufacturer's specifications; 20% should be assumed, so the required area for application is 1 380 acres (560 ha). The application frequency should be as high as possible, again depending on manufacturer's specifications, but at least 2 to 3 rotations per week are desirable to minimize the high instantaneous rates needed to apply all the wastewater to soil.

8.7.1.8 Summary for Slow Rate System Design

A summary of the principal design factors for the most cost-effective alternative, slow rate treatment by center pivot distribution, is presented in Table 8-9.

TABLE 8-9

DESIGN FACTORS, SLOW RATE TREATMENT WITH CENTER PIVOT DISTRIBUTION

Total annual wastewater application, ft	9.7
Length of application season, wk	32
Length of storage, wk	20
Nitrogen balance	
Applied, lb/acre·yr	743
Denitrification, lb/acre·yr	147
Crop uptake, lb/acre·yr	325
Percolate, lb/acre·yr	271
Avg monthly percolate nitrogen concentration, mg/L	8.4
Preapplication treatment detention time, d	
Aerated lagoons	6
Storage lagoons (maximum)	140
Land required, acres	
Wetted area	1 150
Total field area (center pivot only)	1 380
Aerated lagoons	15
Storage lagoons	360
Total (including 10% for miscellaneous)	1 980
Additional application criteria	
Maximum monthly infiltration volume (July), in./mo	22.4
Phosphorus retention (required soil volume), in.	
Conservative	22
Realistic	11
Trace metals	Not restricting

1 ft = 0.305 m
1 lb/acre = 1.12 kg/ha
1 acre = 0.405 ha
1 in. = 2.54 cm

8.7.2 Combined Overland Flow and Rapid Infiltration

The process design for the combined system of overland flow followed by rapid infiltration is presented here. Hydraulic loading rates and cycles and distribution systems are discussed. Preapplication treatment and storage requirements will be the same as for a slow rate system.

8.7.2.1 Hydraulic Loadings and Cycles

The overland flow system will be loaded at 8 in./wk (20 cm/wk) for approximately 22 of the 32 week application period. Applications will be for 6 h/d on a 6 d/wk schedule. This allows 18 h/d of resting plus a full day of resting once a week. This cycle is typical of operating systems (see Section 5.1.4, 7.11, and 7.12).

During the period from October to May, there will be days when the temperature will be below freezing and storage will be provided. When conditions are favorable in this time period, the overland flow system will be loaded at 4 in./wk (10 cm/wk) by operating 6 h/d for approximately 3 d/wk.

The rapid infiltration system will receive the treated runoff from the overland flow slopes. Nitrogen removal is nearly complete in overland flow, so the rapid infiltration system can be managed to maximize hydraulic loading rates (rather than to optimize denitrification, as is the case when rapid infiltration receives a primary or secondary effluent). Thus, application will be for 2 days to a set of basins with a 6 day drying period. Therefore, 42.6 in. (108 cm) of water will be applied over 2 days followed by resting. The water should infiltrate within a day after application ceases. Using the procedure in Appendix C, field testing should be conducted prior to final design to verify adequate infiltration rates. The flooding basin technique, as shown in Figure C-1, should be used for the determination of infiltration rates. It is recommended that several 20 ft^2 basins, located in representative areas of the site, be employed. The resulting infiltration rate data should be analyzed according to the procedure discussed in Appendix C (Section C.3.1).

Soil borings at the proposed rapid infiltration site should also be examined to verify the lack of restrictive layers in the soil profile.

8.7.2.2 Distribution System

For overland flow, the aerated lagoon effluent would be applied using the bubbling orifice (surface application technique used at Pauls Valley, Section 7.11). The application would be at the top of the

150 ft (45 m) long slopes. The slopes would be between 3 and 4%. The runoff would be collected in a series of ditches and conveyed to the rapid infiltration basins. Overland flow effluent would be applied to the rapid infiltration basins on a cycle of 2 days wet and 6 days dry. The 100 acres of basins would be divided into basins ranging in size from 3 to 10 acres each. Four sets of basins (A through D) with each set containing about 25 acres would be established. For 2 days the application would be to set A, followed by sets B, C, and D in rotation. In actual practice some basins will have higher and some lower infiltration rates and the length of flooding and drying can be modified accordingly.

8.8 References

1. Pound, C.E., R.W. Crites, and D.A. Griffes. Costs of Wastewater Treatment By Land Application. Environmental Protection Agency, Office of Water Program Operations. EPA-430/9-75-003. June 1975.

2. Metcalf & Eddy, Inc. Wastewater Engineering. New York, McGraw-Hill Book Co. 1972.

APPENDIX A

NITROGEN

A.1 Introduction

Application of wastewater on land, as compared to the more common practice of discharge to surface waters, has a number of advantages to recommend it. One of these is conservation of valuable resources in the form of contained nutrient elements. Only one of these nutrient elements--nitrogen--is considered here. Where it is feasible to do so, it is much more logical to use nitrogen, the production cost of which is continually increasing, in the production of essential food and fiber rather than to treat it entirely as a waste. Nearly all soils respond to additions of nitrogen by increasing production; however, the requirements of nitrogen for optimum crop production and the need to treat large volumes of nitrogen-containing wastewater may not be in balance. Nitrogen applied to soils in amounts greatly in excess of crop needs and allowed to percolate to the groundwater may result in contamination of the groundwater through leaching of nitrates below the root zone. Nitrogen transformations, removal mechanisms, and overall removals by the land treatment methods are described in this appendix.

A.2 Nitrogen Transformations

A.2.1 Nitrification

In discussing removal of nitrogen from applied wastewater, it is important to understand something about the complex and interrelated series of nitrogen transformations that may occur in soils. The predominant form of nitrogen in wastewater is usually ammonium, although some nitrate is also likely to be present if the preapplication treatment processes have included one or more aerobic stages. A small quantity of organic nitrogen, of which a part is soluble and readily convertible to ammonium through microbial action, is also usually present. Insoluble organic nitrogen associated with the particulate matter is also convertible to ammonium, although somewhat more slowly. When wastewater is applied to soil, a variety of reactions are initiated, some biological and some nonbiological. Of the biological reactions, nitrification and denitrification are very important. Nitrification is important because it converts a form of nitrogen not readily subject to leaching to one that moves readily with percolating water. Denitrification is important because it is the principal process by means of which nitrogen as nitrite or nitrate is lost from the soil system through conversion to gases that may escape to the atmosphere.

A.2.1.1 Nitrifying Bacteria

The conversion of ammonium to nitrate in soil and water systems is due primarily to activities of a few genera of autotrophic bacteria of which Nitrosomonas and Nitrobacter are the most important. These bacteria are normal soil inhabitants and are usually present in sufficient numbers to convert added ammonium to nitrate rapidly and completely, if environmental conditions are suitable. Schloesing and Muntz first discovered the biological nature of the nitrification process by pouring sewage containing ammonium onto columns of soil mixed with limestone and found that, after the elapse of a few days, nitrate appeared in the effluent at the bottom [1].

The nitrifying bacteria are obligate aerobes that derive their energy from the biochemical reactions involved in oxidation of ammonium or nitrite. The principal reactions may be written as follows:

$$NH_4^+ + 3/2\ O_2 \longrightarrow NO_2^- + H_2O + 2\ H^+ \qquad \text{(A-1)}$$

$$NO_2^- + 1/2\ O_2 \longrightarrow NO_3^- \qquad \text{(A-2)}$$

The first reaction is carried out by bacteria of the genera Nitrosomonas, Nitrosococcus, Nitrosocystis, and Nitrospira; the second is accomplished by Nitrobacter and related species. These bacteria require no organic matter as a source of energy. A number of heterotrophic nitrifiers are known to occur, but their activity appears to be slight compared to that of the autotrophic forms [2]. Although nitrifying bacteria are abundant in most soils, populations may be initially low in subsoils or in coarse-textured soils that are prone to be dry much of the time. In such soils, several weeks may be required for nitrifiers to attain maximum numbers after application of wastewater is begun.

A.2.1.2 Rates of Nitrification

Rate constants based on the assumption of steady state conditions and first order kinetics have been published [3]; but these may have little value in relation to field situations where soil properties, population size, and other variables are subject to considerable fluctuation. Rate constants that have been normalized to take into consideration the size of the nitrifying population are more comparable from one soil to another, but are impractical for application to field conditions owing to the difficulty of obtaining reliable counts. Under favorable moisture and temperature conditions, measured values of ammonium converted to nitrate ranging from 5 to 50 ppm nitrogen per day (soil basis) have been reported [4, 5]. For purposes of calculation, if one assumes a depth of only 4 in. (10 cm) of soil implicated in the nitrification process owing to ammonium adsorption near the surface, it can be determined that these rates are equivalent to 6 to 60 lb/acre·d (6.7 to 67 kg/ha·d) of nitrogen. The lower rate would be sufficient to nitrify the ammonium in 1.2 in. (3 cm) of wastewater per day containing 20 mg/L of NH_4^+; and at the upper end of the range, 12 in./d (30 cm/d) could be accommodated. Even higher nitrification rates in soil columns have been reported [6, 7]. These calculations are consistent with observations that complete conversion of input nitrogen to the nitrate form occurs if wastewater application periods are short enough to prevent development of anaerobic conditions [8, 9].

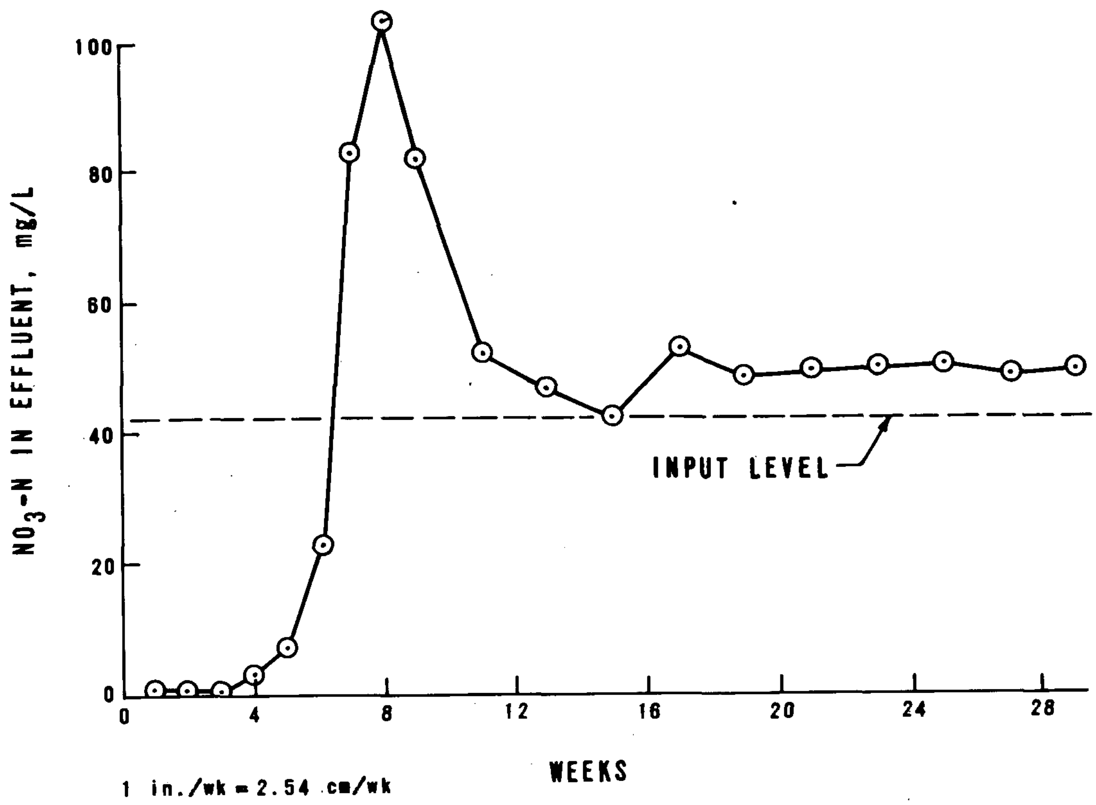
The tendency of soils to adsorb ammonium near the surface may result in temporary buildup of ammonium in a shallow layer, particularly if the nitrifying population has not been increased by previous inputs of ammonium. This situation results subsequently in a wave of nitrate at a high concentration following the increase in the number of nitrifiers to a level that permits rapid oxidation of the adsorbed ammonium. This is illustrated in Figure A-1, where wastewater containing 42 mg/L of NH_4^+-N applied to a soil column at the rate of 3 in./wk (7.5 cm/wk) produced an effluent containing up to 107 mg/L of NO_3^--N [10]. Following the period of population buildup (about 5 weeks in this soil), ammonium was nitrified as rapidly as it was applied, and nitrate concentrations fell to the input level. A recurring nitrate wave phenomenon is readily observed in systems of alternate flooding and drying [11]. Here it is due to the intermittent nature of nitrification, which occurs only during the drying cycle when oxygen is available.

The rate of nitrification is much more likely to be inhibited by lack of oxygen or low temperature than by an inadequate population of nitrifiers. The usual situation in soils is that nitrite rarely accumulates, indicating that the activities of _Nitrobacter_ proceed more rapidly than does the oxidation of ammonium. Nitrite oxidation is inhibited by free ammonia in liquid systems, particularly when the pH is alkaline; but in soils, adsorption of ammonium prevents this inhibition from becoming a practical consideration in most circumstances.

Prolonged application of high ammonia content wastes, such as sludge, may result in loading that exceeds the ammonium adsorption capacity in which case free ammonia may reach concentrations sufficiently high to retard nitrite oxidation.

FIGURE A-1

NITRATE IN EFFLUENT FROM A COLUMN OF SALADO SUBSOIL RECEIVING 3 IN./WK OF WASTEWATER CONTAINING 42 mg/L NH_4^+-N, SHOWING HIGH-NITRATE WAVE [10]



A.2.1.3 Effects of Soil Properties on Nitrification

A.2.1.3.1 Aeration

The theoretical oxygen requirement in nitrification is for about 4.6 mg oxygen per milligram of ammonium-N. Although the nitrifiers are obligate aerobes, they will continue to function at oxygen concentrations well below that of the atmosphere [12, 13].

The rate at which oxygen diffuses to the sites where nitrifying bacteria are located in relation to the rate of oxygen utilization is of critical importance. Studies in wastewater treatment systems indicate that the minimum level of dissolved oxygen that will permit ammonium oxidation is around 0.5 mg/L [14]. In soils, it is impossible to measure the dissolved oxygen in the microsites inhabited by bacteria, and in any event the situation is complicated by the presence of large numbers of heterotrophes which may use a greater proportion of the available oxygen than do the nitrifiers, if oxidizable carbon is available. Thus, anaerobic conditions may readily develop in the smaller pores of unsaturated soils. Lance et al. found that both diffusion and mass flow of oxygen were important as transport mechanisms between periods of intermittent flooding in rapid infiltration [6]. Continuous application of wastewater to soils stops nitrification below the immediate surface by filling soil pores and preventing diffusion of oxygen downward. In overland flow systems, nitrification can proceed as a result of aeration of surface water as it moves over the land via sheet flow [15].

Carbon dioxide is required by nitrifying bacteria as a source of carbon, but since wastewaters usually contain considerably more bicarbonate than ammonium, there is little likelihood that nitrification is ever limited by lack of CO_2 in land application.

A.2.1.3.2 Temperature

Like all biological processes, nitrification is affected by temperature. There is evidence that nitrifiers can adapt to the temperature of their environment to some extent [16], but the optimum usually falls between 75 and 95°F (24 and 35°C). Minimum temperatures as low as 36°F (2°C) have been reported [5, 17]. As a rule of thumb, the activity of nitrifiers increases by a factor of 2 for every 18°F (10°C) rise in temperature. Obviously, nitrification is stopped altogether when soils are frozen.

A.2.1.3.3 pH

The optimum pH for nitrification is in the neutral-to-slightly-alkaline range corresponding closely to the pH of most wastewater. However, when wastewater is applied to soil, the controlling factor is usually the pH of the soil because of the much higher buffer capacity of soils containing any appreciable amount of clay and organic matter. The pH of very coarse textured soils may be altered somewhat by addition of wastewater, particularly with high-rate applications. Nitrification falls off sharply in acid soils, with a limiting value in the neighborhood of pH 4.5 [4].

Nitrification is an acid-forming process, with the liberation of two protons for each ammonium ion oxidized; but the presence of bicarbonate and other buffering substances in wastewater is usually sufficient to neutralize the acid as it is formed [18]. With prolonged application of wastewater, even strong acidic soils may be made neutral or alkaline [19], indicating that acid produced during nitrification does not play a dominant role.

A.2.2 Denitrification

A.2.2.1 Microorganisms

The important bacteria in denitrification are heterotrophes belonging to the genera *Pseudomonas*, *Bacillus*, *Micrococcus*, and *Achromobacter*. One of the autotrophic sulfur-oxidizing bacteria, *Thiobacillus denitrificans*, may also play a significant role in denitrification where reduced forms of sulfur are present. The denitrifiers are facultative anaerobes that preferentially use gaseous oxygen, but can use nitrite and nitrate as electron acceptors in place of oxygen when concentrations of oxygen become very low. Denitrifying bacteria, like the nitrifiers, are common soil organisms of widespread distribution. Focht and Joseph reported very little correlation between denitrification rates and numbers of denitrifying bacteria in soils, indicating that factors other than population size are likely to be rate-limiting [20].

A.2.2.2 Energy Sources

The denitrification reaction may be written

$$C_6H_{12}O_6 + 4\ NO_3^- \longrightarrow 6\ CO_2 + 6\ H_2O + 2\ N_2 \qquad \text{(A-3)}$$

where glucose is used as an example of an organic energy source. In this example, 3.2 g of glucose is required for each gram of nitrogen denitrified. The decomposable organic matter required for denitrification may be present in the soil, may be carried in the wastewater, or may be produced by plants growing on the soil. For municipal wastewaters that are applied after having been stabilized to the degree that most of the BOD has been removed, the organic matter status of the soil to which the water is applied is likely to be more important than that of the wastewater itself for slow rate applications. Cannery wastewater with its high BOD is an exception, as are certain other types of industrial wastewater. The typical distribution of organic matter in soils is such that the high concentrations occur at or near the surface and decline progressively with depth. Moreover, the availability of organic matter near the soil surface as a source of energy for microorganisms is often greater than that at lower depths. Gilmour et al. showed that a flooded surface soil containing 0.91% total carbon denitrified added nitrate readily without organic amendments, but the subsoil containing 0.48% total organic carbon failed to denitrify unless an available organic substrate was supplied [21]. This means that the zone of most active denitrification is likely to be near the soil surface in spite of its proximity to the atmosphere. This has been demonstrated in field experiments by Rolston et al. who observed maximum rates of production of N_2O and N_2 within the top 4 in. (10 cm) [22]. Nitrous oxide is an intermediate in denitrification and may be evolved from soil before it has an opportunity for further reduction to N_2, particularly when it is produced near the surface. McGarity and Myers observed a close correlation between denitrifying activity and total carbon in some soils, whereas in others there was little or no correlation with organic matter parameters [23]. They suggested that this was due to localized accumulation of small quantities of energy-rich available organic matter. With continued input of wastewater, any such accumulations would disappear.

Stanford et al. found a highly significant correlation between total soil carbon and denitrification rate constants for a group of 30 soils of diverse properties. A still better correlation was obtained with extractable glucose carbon [24]. Still not answered, however, is the question of whether such rate constants based on the assumption of first order kinetics would hold up over longer periods than the 10 days used for their determination. Since rate constants are related to available carbon, it is likely that they would decrease over time.

Elemental sulfur or sulfides can also be used as an energy source for denitrification, as has been shown by Mann et al. [25]. Sulfides may play a role in denitrification in marshland, or where anaerobic sludge is disposed to land.

In application of high BOD wastewater, such as cannery wastes, rapid denitrification is very probable. Law et al. reported 83 to 90% removal of total nitrogen from overland flow treatment of cannery wastes [26].

A.2.2.3 Aeration

The threshold oxygen concentration which inhibits denitrification has been shown by Skerman and MacRae to be very low, in the vicinity of 0.2 mg/L [27, 28]. Temporally or spatially restricted anaerobism is a feature of virtually all soils. Temporary saturation may occur during wastewater application, with exclusion of oxygen from the soil pores, or oxygen deficiency may develop in an unsaturated soil if the rate of consumption exceeds the rate of replenishment. The latter circumstance is especially likely in the smaller soil pores. Thus, denitrification may take place in a soil considered to be well aerated. Prolonged exclusion of oxygen from the soil, as in continuous flooding, causes denitrification to cease from lack of nitrate, unless this is present in the input water. Lance et al. reported that, in columns of a loamy sand soil, both mass flow and diffusion were important mechanisms of oxygen transport during intermittent flooding with secondary effluent [6]. They noted that enough oxygen entered the soil during a 5 day drying period to oxidize all the ammonium applied during 6 days of high-rate application of wastewater containing 20 mg/L of NH_4^+-N. Application of ammonium in excess of that which could be oxidized during the drying period resulted in an increase of NH_4^+ in the reclaimed water. Klausner and Kardos reported little effect of secondary sewage effluent on oxygen diffusion rates in silt loam and clay loam soils over the application range of 0 to 2 in./wk (0 to 5 cm/wk) [29].

In overland flow, a sharp gradient in oxygen concentration can develop between the thin layer of water in contact with the atmosphere and the underlying soil where an anaerobic zone may develop just below the soil-water interface. Nitrates formed in the aerated flowing water can diffuse into the reducing zone of soil and undergo denitrification [15]. The development of this reducing zone is favored by the high BOD of wastewaters and the relatively impermeable soils to which the overland flow system of treatment is adapted.

A.2.2.4 Temperature

The optimum temperature for denitrification in soils is very high, 140 to 150°F (60 to 65°C), but Stensel et al. reported little temperature effect in the 68 to 86°F (20 to 30°C) range [30]. Of greater practical importance is the minimum temperature. Bremner and Shaw observed very slow denitrification at 36 and 41°F (2 and 5°C), but the rate increased very rapidly up to 77°F (25°C) [31].

A.2.2.5 pH

Denitrification is very slow in acid soils, increasing rapidly with increasing pH up to the neutral-to-slightly-alkaline range [32, 33]. Denitrification affects soil pH according to the reaction

$$NO_3^- + \text{organic matter} \longrightarrow N_2 + H_2O + CO_2 + OH^- \qquad \text{(A-4)}$$

and has the effect of neutralizing a part of the acid produced in nitrification. The relative balance between nitrification and denitrification will therefore have an influence on changes in soil pH resulting from wastewater application, although other factors are of greater importance in regulating pH, as has been indicated previously.

A.2.2.6 Nitrate Concentration

The denitrification rate is independent of nitrate concentration over a fairly wide range [32, 34]. Recently, Volz et al. reported denitrification to be a zero order reaction, but with the possibility of some dependence on nitrate concentration at very low nitrate levels [35]. Over the range of nitrate concentrations that commonly occur in wastewater, there is little effect on denitrification rates.

A.2.2.7 Effects of Living Plants

The presence of living plants has been shown to stimulate denitrification [36, 37, 38]. Woldendorp attributed this to two effects: (1) low oxygen concentrations in the rhizosphere produced by respiration of roots and microorganisms, and (2) root excretions serving as a source of decomposable organic matter [36]. Similar conclusions were reached by Stefanson, who reported that in the presence of plants, N_2 was evolved preferentially, while in their absence, N_2O accounted for most of the nitrogen loss [37]. Woldendorp also suggests the possibility of stimulation of denitrification by specific amino acids secreted by plant roots [38]. The role of living plants in the denitrification process is particularly important in slow rate and overland flow systems.

A.3 Nitrogen Removal from the Soil System

A.3.1 Crop Uptake

A major advantage of applying wastewater to land is the possibility of recycling part of the plant nutrient content. The important consideration from the standpoint of nitrogen content is the relationship between the crop requirement and the quantity applied in the wastewater. It should be recognized that a crop does not utilize all of the mineralized nitrogen in the root zone. The fraction of total nitrate in the soil that is assimilated by the roots of growing plants varies tremendously, depending on the nature of the plant, depth and distribution of rooting, nitrogen loading rate, rate of moisture flux through the root zone, and other factors; but in general, the efficiency of uptake is not high. Grasses, particularly perennials, tend to be somewhat more efficient than row crops. It is obviously advantageous to have the crop growing actively during all or most of the year in order to maximize nitrogen removal in wastewater application, but climatic restraints make this impossible in many locations. Terman and Brown [39] calculated by means of a regression procedure that average nitrogen recovery at all rates by Bermuda grass in the experiments of Burton and Jackson [40] was 59%.

The most accurate estimates of nitrogen uptake efficiencies are those obtained by use of isotopically labelled input nitrogen, but few of these are available. Apparent uptake values are often computed by dividing the quantity of N found in the crop by the quantity applied. Where the amount of indigenous soil nitrogen is large, the discrepancy between actual and apparent uptake may be enormous. Some comparisons for corn, where actual N uptake was determined by the isotope procedure, and apparent uptake by the conventional procedure, are given in Table A-1 [41].

TABLE A-1

NITROGEN UPTAKE EFFICIENCIES OF CORN IN RELATION TO QUANTITIES OF NITROGEN AND WATER APPLIED [41]
Percent

N applied, lb/acre	Irrigation water applied					
	7.9 in.		23.6 in.		39.4 in.	
	Actual	Apparent	Actual	Apparent	Actual	Apparent
80	57.1	173	55.4	182	55.7	172
160	54.3	122	63.2	139	64.7	123
320	42.3	68.6	43.8	75.5	48.0	78.6

1 lb/acre = 1.12 kg/ha
1 in. = 2.54 cm

Sopper and Kardos in Pennsylvania computed apparent removal efficiency values of 242 and 334% of total applied nitrogen by two varieties of corn silage receiving 1 in./wk (2.5 cm/wk) of wastewater during a single year [42]. At 2 in./wk (5 cm/wk) the nitrogen removal efficiency dropped to 145%. Over a 6 year period, Reed canary grass removed 97.5% of the nitrogen applied in 536 in. (13.6 m) of wastewater. In a hardwood forest, the nitrogen removal efficiency at 2 in./wk (5 cm/wk) was only 39%. It is clear that the apparent removal values in excess of 100% include a great deal of nitrogen resulting from decomposition of soil organic matter and could not be maintained over a long period of time. Much lower values for nitrogen recovery by crop uptake have been reported by McKim et al. [43] and by Karlen et al. [44].

Total quantities of nitrogen removed by harvested crops generally fall in the range 50 to 400 lb/acre·yr (56 to 450 kg/ha·yr), depending on the nature of the crop, fertility of the soil, and a number of management parameters [45]. These amounts may account for a major part of the input nitrogen in slow rate and overland flow systems, and in the former, application rates are primarily limited by plant uptake. However, plant uptake is of relatively little consequence in rapid infiltration systems where input levels as high as 15 tons/acre·yr (33.6 Mg/ha·yr) of nitrogen have been reported [8].

A.3.2 Volatilization of Ammonia

The equilibrium between NH_4^+ and NH_3 is regulated by pH, and the proportion of free NH_3 is small at the pH value of most wastewater.

Application of water through sprinkler systems increases evaporation and with it the quantity of ammonia volatilized. Scott states that the net loss of water during sprinkler irrigation may vary from as low as 5% to as much as 40% of the water applied [46]. Henderson et al. measured ammonia losses as a function of pH of fertilizer solutions applied by sprinkler irrigation and found that, in general, these were less than 10% between pH 7 and 8, but the curve increased sharply above pH 7.8 [47]. Their data would include evaporative losses between the sprinkler head and the soil surface.

With any type of wastewater application, ammonia losses from the soil surface may occur during drying. The magnitude of such losses is highly variable, depending on rate of application, extent of drying, clay content of the soil, pH of the surface soil, temperature, and type of plant cover, if any [48, 49, 50]. The coarse-textured soils favored for wastewater application are prone to ammonia loss because of their low clay content and tendency to dry quickly, although because of their low retention capacity the proportion of total ammonia retained near the surface is unlikely to be large. In a greenhouse study Mills et al. reported that at pH values above 7.2 at least half the nitrogen applied to a fine sandy loam soil was volatilized as ammonia, most of it within 2 days of application [49]. In a laboratory study, Ryan and Keeney measured ammonia volatilized from surface-applied wastewater sludge containing 950 mg/L of NH_4^+-N and obtained values ranging from 11 to 60% of the applied NH_4^+-N, depending on the nature of the soil and loading rate [51]. Losses decreased as clay content of the soil increased, but were directly related to the loading rate. Repeated applications of sludge produced greater percentage losses than a single application.

A.3.3 Denitrification

A.3.3.1 Slow Rate Process

The slow rate process, usually on land which is vegetated at least part of the year, is basically an irrigation procedure. In arid regions the wastewater is used to meet the evapotranspiration requirements of the growing plants, and in humid regions the quantity of wastewater applied is limited to levels which do not greatly exceed plant requirements for water. The soil is thus maintained primarily in an aerobic condition, and nitrification is the dominant process. Nevertheless, in agricultural practice, carefully controlled nitrogen balance experiments usually reveal an unaccounted-for deficit which is attributed to denitrification [52]. The magnitude of this deficit typically falls in the range of 15 to 25% of the applied nitrogen. A balance sheet from a field experiment is presented in Table A-2 [41]. Isotopically labelled nitrogen fertilizer was used which made it possible to distinguish between the applied nitrogen and that present in soil or added from other sources. The losses over a 3 year period were a remarkably constant fraction of the input nitrogen, and consistent in magnitude with other reported values [52].

TABLE A-2

THREE YEAR BALANCE SHEET FOR ISOTOPICALLY LABELLED NITROGEN FERTILIZER APPLIED TO CORN PLOTS ON HANFORD SANDY LOAM [41]

Total N added, lb/acre	Removed in grain, lb/acre	Remaining in soil, lb/acre	Unaccounted for, lb/acre	Loss, %
300	135	118	47	16
600	274	196	129	22
900	312	384	204	23
1 200	317	589	294	25
1 500	321	930	248	17

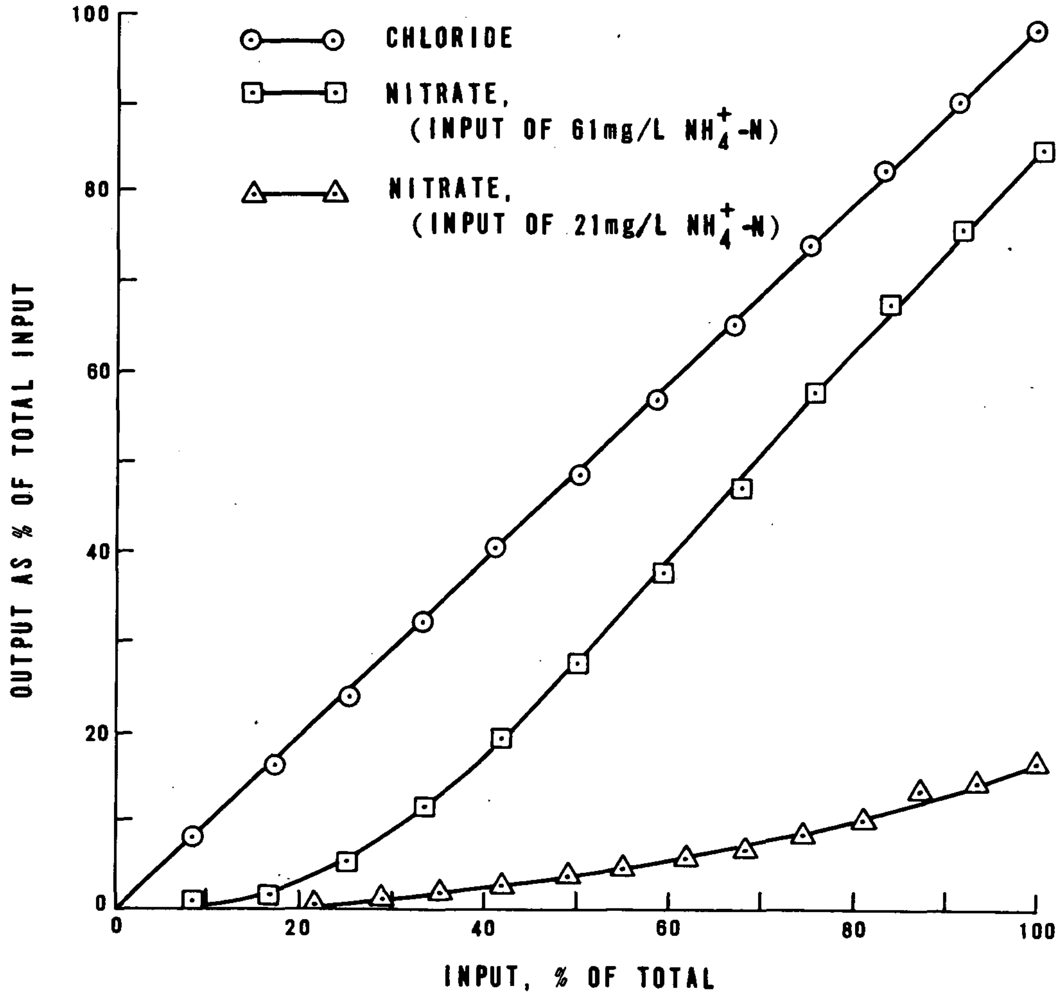
1 lb/acre = 1.12 kg/ha

In wastewater application, the fraction of input nitrogen which is denitrified is strongly dependent on available carbon in the soil. This is illustrated in Figure A-2 which shows data from a column of Panoche sandy loam receiving 3 in./wk (7.5 cm/wk) of wastewater at two different NH_4^+-N levels over a 6 month period. The chloride curve shows the behavior of a nonreactive ion, with no holdup in the soil. At the 21.4 mg/L NH_4^+-N level, there was complete removal of the first 22% of input, followed by several months of nearly complete removal. Over the entire period there was 16% recovery, or 84% removal of input nitrogen. However, at the 61 mg/L NH_4^+-N level, once the supply of available carbon was exhausted, there was very little denitrification. Overall recovery in the latter case was 83%, corresponding to only 17% removal. It should be possible to adjust the loading rate for most soils so as to maximize denitrification in cases where nitrogen removal is the principal consideration.

Agricultural wastes, such as straw residues and manures, are effective in stimulating denitrification, but these pose problems of handling and availability at land treatment sites. Olson et al. applied manure to Plainfield sand at rates varying from 10 to 270 tons/acre (22.4 to 605 Mg/ha) [53]. Under aerobic conditions, nitrate accumulated to 25 to 180 mg/L, but when the soil was maintained in a saturated condition, as might be done by ponding during wastewater application, virtually no nitrates were found. Meek et al. observed that redox potentials in calcareous Holtville clay receiving 180 tons/acre (403 Mg/ha) of manure in each of two successive years did not fall below 400 mV with the normal irrigation schedule, whereas the potential dropped to zero when the number of irrigations was doubled [54]. These authors suggest that it is possible to adjust manure application rates and irrigation schedules for fine-textured soils to achieve maximum denitrification. The principle is applicable to other kinds of wastes as well.

FIGURE A-2

EFFECT ON INPUT NH_4^+-N CONCENTRATION ON N REMOVAL FROM WASTEWATER APPLIED TO PANOCHE SANDY LOAM AT THE RATE OF 3 IN./WK FOR 6 MONTHS [41]



1 in./wk = 2.54 cm/wk

A.3.3.2 Rapid Infiltration

In the intermittent application of wastewater to soil, it can be safely assumed that all of the input ammonium not volatilized will eventually be nitrified if the adsorption capacity of the soil is not exceeded and if the periods of application are interspersed with drying periods of sufficient length and frequency to replenish soil oxygen. Bouwer et al. reported essentially quantitative conversion of ammonium to nitrate in rapid infiltration systems having 2 to 3 days of flooding alternated with 5 days of drying [8]. Robeck et al. obtained about 74% ammonia removal in Ottawa sand with shorter and more frequent applications of wastewater at a rate of 8 in./d (20 cm/d) containing nitrogen equivalent to 50 lb/acre·d (55 kg/ha·d) [9]. This removal was attributed primarily to nitrification. Thus, in rapid infiltration systems, nitrification is the dominant process unless specific steps are taken to promote denitrification.

In rapid infiltration experiments with soil columns, Lance and Whisler found no net removal of nitrogen with 2 days of flooding followed by 5 days of drying, but net removal was 30% with longer cycles involving 9 to 23 days of flooding and 5 days of drying [7]. Lance et al. developed two successful methods for maximizing denitrification in high rate applications which achieved 75 to 80% removal of nitrogen [55]. On the basis of their finding that the percentage of nitrogen removal increased exponentially as the infiltration rate decreased, they reduced infiltration rates by soil compaction to a level that allowed nitrate formed during the dry period to mix with the wastewater subsequently applied in order to provide a favorable ratio of carbon to nitrate. The second method involved recycling water of high nitrate content that had passed through the column as a nitrate peak. This was mixed with two parts of secondary effluent and recycled throughout the remainder of the flooding period. Both methods encounter practical difficulties in field application. Adjusting depth of ponding, compacting the surface of the soil, and altering the solids content of applied wastewater have been suggested as means of changing infiltration rates [55]. Recycling high nitrate water in the field would require interceptor drains below the water table, the effluent from which would be pumped to a holding pond and mixed with wastewater prior to reapplication.

An alternative method of increasing nitrate removal by denitrification is to add an energy source. Methanol has been used for this purpose in reducing the nitrate content of drainage water [56]. The theoretical methanol requirement in this process for wastewater containing 20 mg/L of NO_3^--N would be 45.7 mg/L, or the equivalent of 1.6 gal of methanol per acre-inch (5.9 L/ha·cm) of water, assuming that all the methanol is used by denitrifying bacteria. Experiments with drainage water showed that up to 90% removal of nitrate could be achieved with water initially containing 20 mg/L of NO_3^--N by addition of 70 mg/L of methanol, or about 150% of the theoretical requirement [56]. It is unlikely that methanol added to municipal wastewater and then applied to soil would be used as efficiently, owing to the presence of large numbers of heterotrophic microorganisms in addition to the denitrifiers. An inherent difficulty is that the period of aerobic microbial activity required for nitrification of input ammonium permits rapid depletion of available carbon, leaving little for use of denitrifiers when the soil is again flooded.

A.3.3.3 Overland Flow

In overland flow treatment, a thin film of wastewater passing over the surface of soils of relatively low permeability serves as a barrier to oxygen movement below the soil surface. This permits the development of anaerobic conditions in the soil near the soil-water interface with attendant denitrification. Other aspects of this type of treatment which favor denitrification are the close proximity of an oxidizing zone in the flowing water, and the high BOD of wastewaters to which this method is applicable. This allows nitrification in the water film, followed by movement of nitrate into the reducing zone below the soil surface where energy for denitrifying bacteria is supplied by soluble organic matter from the wastewater. Quantitative data showing the relative importance of denitrification in relation to other nitrogen removal mechanisms such as plant uptake and ammonia volatilization are lacking, but reported high removal efficiencies of 75 to 90% suggest that denitrification is the dominant process [57, 58, 59].

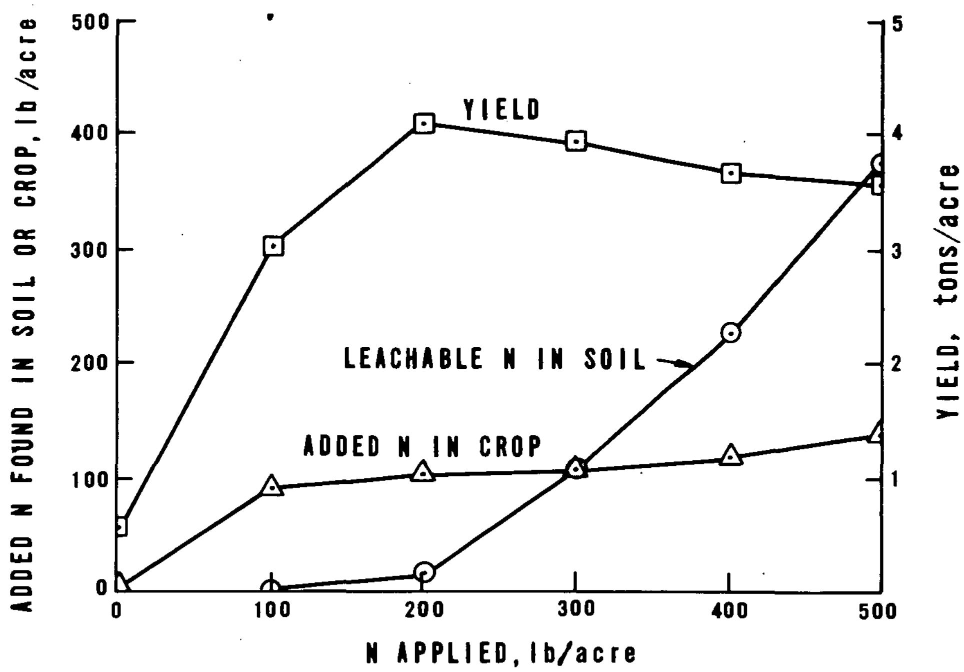
A.3.4 Leaching

Nitrogen applied in excess of crop removal is potentially subject to leaching, but in practice, losses by volatilization of ammonia and by denitrification diminish the actual quantities of nitrogen leached. In arid regions, some leaching is essential to prevent excessive accumulations of salt. In most situations, some movement of nitrate from the root zone to the groundwater is unavoidable.

In land treatment systems, it is desirable to have an estimate of the amount of nitrate leached, but reliable estimates are difficult and expensive to obtain. In considering nitrate as a pollutant, it is important to bear in mind that total mass flow is of greater significance than concentration <u>per se</u>. In applications on cropland at rates not greatly in excess of the consumptive use requirement for water, fairly high concentrations of nitrate in the subsoil would not represent a high pollution hazard because of the low leaching fraction. On the other hand, in high rate application with a large leaching fraction, a much greater mass of nitrogen may move into an aquifer even though the nitrate concentration is relatively low. In crop irrigation systems, the quantity of nitrogen that is potentially leachable (nitrate form) increases sharply above the input level required to achieve maximum crop production, as is illustrated in Figure A-3. The applied nitrogen cannot be balanced by the leachable nitrogen plus crop nitrogen because incorporation of nitrogen into soil organic matter and denitrification amounts in Figure A-3 are unknown.

FIGURE A-3

YIELD, CROP UPTAKE OF N, AND POTENTIALLY LEACHABLE NITRATE IN RELATION TO FERTILIZER APPLICATION RATE ON CORN GROWN ON HANFORD SANDY LOAM [41]



1 lb/acre=1.12 kg/ha
1 ton/acre = 2.24 Mg/ha

Monitoring nitrate flux in a field situation is not a simple matter. Porous ceramic probes, sometimes referred to as suction lysimeters, are often used to obtain samples of soil solution at various depths and locations without disturbing the soil after the initial installation. A rather dense network of such probes is required to obtain reliable estimates of soil nitrate concentrations. Even in soils considered to be uniform, these concentrations are subject to wide variations both in time and in space. This is illustrated by the data of Table A-3, obtained from probes located in a corn field on Yolo fine sandy loam. It will be noted that individual samples vary by an order of magnitude or more from replicate samples in several instances, and standard deviations from the mean ranged from 32 to 114% of the mean.

TABLE A-3

NITRATE-N CONCENTRATIONS IN SOIL SOLUTION SAMPLES OBTAINED BY MEANS OF SUCTION PROBES AT FOUR DEPTHS ON TWO DATES IN A CORN FIELD ON YOLO FINE SANDY LOAM [41]

	No. of samples and depth			
	4 at 4 ft	4 at 6 ft	8 at 8 ft	8 at 10 ft
Jul 28, 1975				
Mean NO_3^--N, mg/L	31.9	25.8	30.3	32.0
Range, mg/L	15.2-60.0	18.9-35.6	6.8-58.9	7.8-55.4
Standard deviation, % of mean	61	32	58	45
Aug 28, 1975				
Mean NO_3^--N, mg/L	19.5	12.3	26.3	32.4
Range, mg/L	1.8-50.4	2.3-22.4	1.8-47.2	5.1-58.9
Standard deviation, % of mean	114	67	66	54

1 ft ≈ 30 cm

It is clear that estimations of nitrogen removal based on a few suction lysimeter samples may be in serious error. It should be further realized that measurements of moisture flux in unsaturated soils are subject to the same kind of variation, making calculations of mass balance even more hazardous. This variability is inherent in sampling natural bodies for virtually any parameter. The conclusion is that it is not generally practical to attempt to estimate nitrate removal from wastewater in slow rate applications by monitoring composition of the soil solution. In rapid infiltration applications, where the amount of water applied is much greater than consumptive use and where applied nitrogen greatly exceeds any soil contribution, measurements made on samples from the zone of saturated flow obtained by means of suction cups, wells, or tile lines are somewhat more reliable.

A.3.5 Storage of Nitrogen in Soil

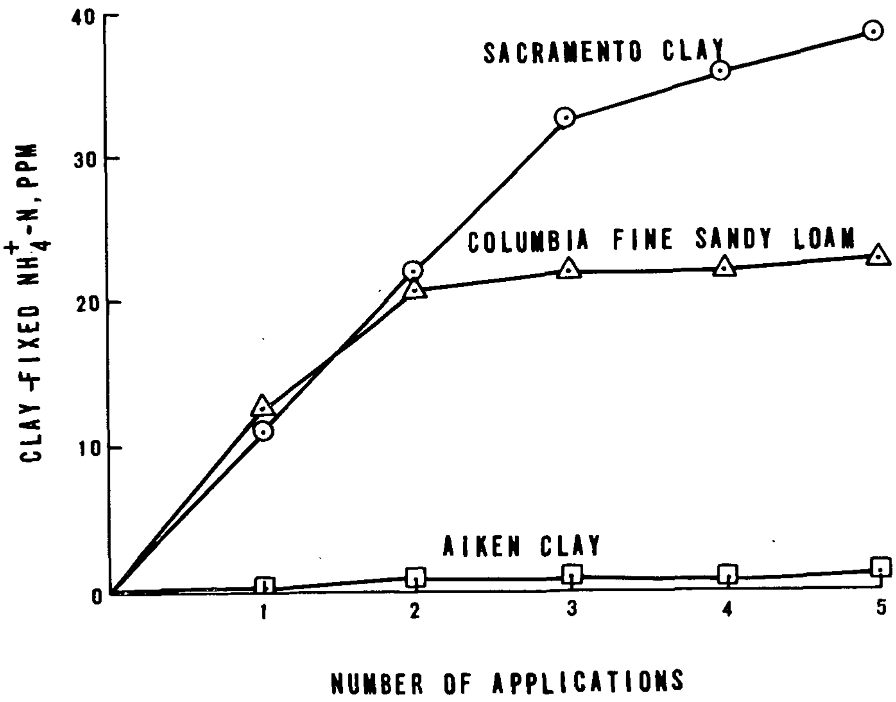
In a theoretical equilibrium situation over the long term, where additions and removals of nitrogen are in balance, the storage capacity of the soil is of little consequence from the standpoint of management practice, even though the residence time in the soil may be quite long. In actual wastewater application practice, particularly with slow rate systems, the storage of nitrogen is very important because equilibrium is not quickly attained. The principal storage mechanisms are fixation of ammonium by clay minerals and organic matter, retention of ammonium as an exchangeable cation, and incorporation into soil organic matter.

A.3.5.1 Ammonium Fixation

Certain clays that commonly occur in soils, particularly those of the vermiculite group, have the ability to trap ammonium ions within the crystal lattice. Ammonium ions thus fixed do not exchange readily with other cations and are not accessible to nitrifying bacteria [60]. Fixation of NH_4^+ by clays is enhanced by wetting and drying cycles but may occur without drying. The quantities so fixed depend on the kinds and amounts of clay present. Quantities of NH_4^+ fixed by three different soils receiving five consecutive applications of a solution containing 100 mg/L of NH_4^+-N without intervening drying periods are shown in Figure A-4. The Aiken clay, containing predominantly kaolinite, fixed no NH_4^+. The Columbia fine sandy loam, typical of coarse textured soils that might be used for wastewater disposal, fixed 22 ppm NH_4^+ (soil basis), equivalent to about 275 lb/acre (308 kg/ha) of nitrogen in the top 3 ft (1 m) of soil. This soil and the Sacramento clay contain vermiculite and montmorillonite capable of NH_4^+ fixation.

FIGURE A-4

CLAY-FIXED NH_4^+ IN THREE SOILS RESULTING FROM FIVE APPLICATIONS OF A SOLUTION CONTAINING 100 mg/L NH_4^+-N, WITHOUT INTERVENING DRYING [41]



Another mechanism of NH_4^+ fixation involves reaction with soil organic matter to form stable complexes. The amounts fixed depend strongly on pH and quantity of organic matter present [61]. It is unlikely that this mechanism is of much importance at the low NH_4^+ concentrations and near-neutral pH of wastewaters normally applied to soils of low organic matter content, but it may assume considerable importance in sludge applications where NH_4^+ concentrations are at least an order of magnitude higher and where organic matter is supplied by the sludge.

A.3.5.2 Exchangeable Ammonium

Like other cations in wastewater, NH_4^+ can be adsorbed by the negatively charged clay and organic colloids in soil. Lance discussed a method of estimating the quantity of NH_4^+ that might be adsorbed from a particular wastewater based on the ammonium adsorption ratio calculated from the concentrations of NH_4^+, Ca^{++}, and Mg^{++} in the water [62]. In slow rate systems, the ammonium adsorption capacity of soils is usually sufficient to retain the applied ammonium near the surface. Continuous flooding in rapid infiltration systems will in time saturate the ammonium adsorption capacity and permit downward movement of ammonium. Retention of ammonium in the exchangeable form is temporary in any case, since the adsorbed ammonium is nitrified when oxygen becomes available; but exchangeable NH_4^+ plays a very important role in the nitrification-denitrification sequence by holding nitrogen near the soil surface until the environment becomes aerobic during drying.

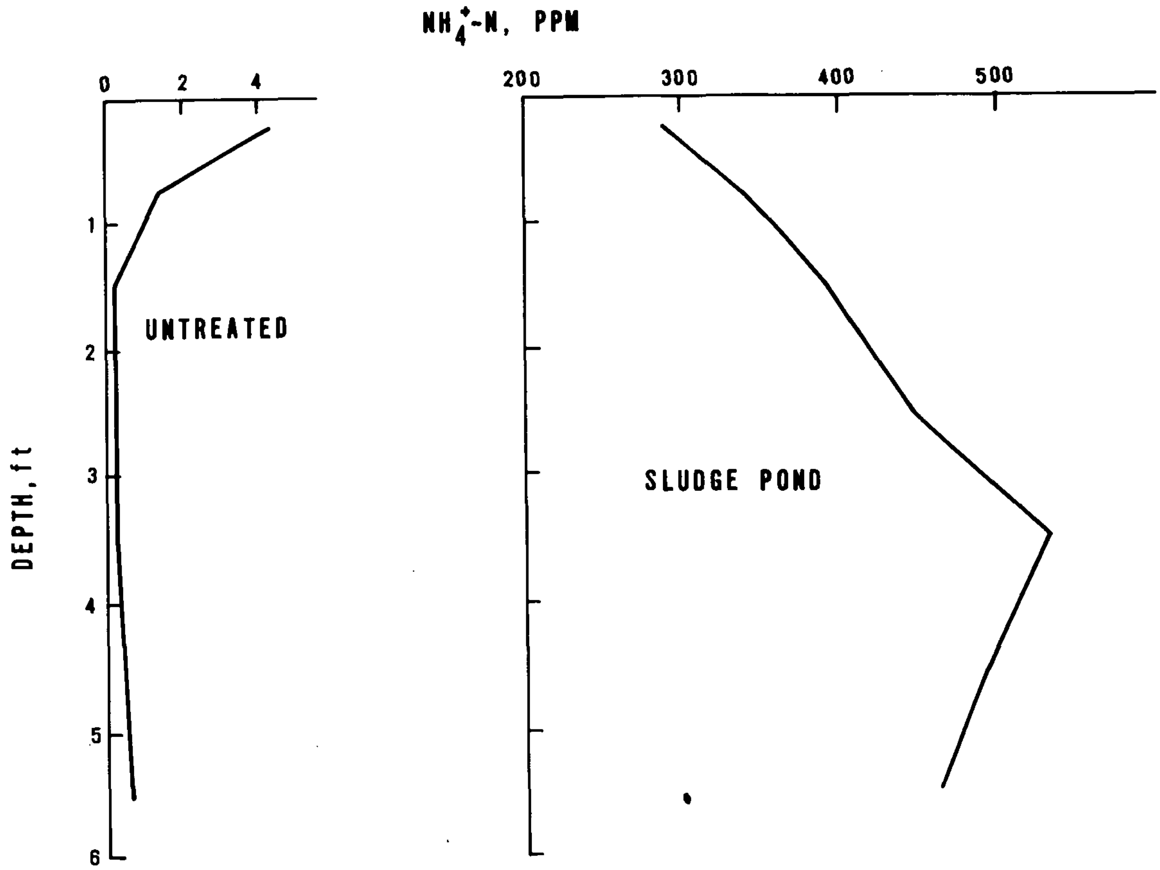
Even in sandy soils of low cation exchange capacity the quantity of exchangeable ammonium is of consequence. A profile of exchangeable NH_4^+ beneath a sludge drying pond as compared to untreated soil is shown in Figure A-5. This represents a situation where high NH_4^+ concentrations combined with a low infiltration rate have resulted in dominance of the cation exchange complex by NH_4^+. The total quantity of exchangeable NH_4^+ in this soil to a depth of 6 ft (1.8 m) is 10 530 lb/acre (11 800 kg/ha). The same profile also contained 1 250 lb/acre (1 400 kg/ha) of NO_3^--N. In a somewhat different situation, Lance cites calculated values of exchangeable NH_4^+ equivalent to 1 554 lb/acre (1 740 kg/ha) for a wastewater applied to a soil with an exchange capacity of only 5 meq/100 g [62].

A.3.5.3 Incorporation Into Organic Matter

Ammonium may be incorporated into organic matter by the fixation mechanism previously discussed, through assimilation by microorganisms, and by plant uptake. Net immobilization by microorganisms requires the presence of decomposable organic matter having a nitrogen content less than about 1.2%. Except for cannery wastes and certain types of industrial wastes, these conditions are not met for land treatment systems. The presence of mature crop residues on land receiving wastewater may result in immobilization of a small amount of nitrogen, though probably not more than 40 to 60 lb/acre (45 to 65 kg/ha).

FIGURE A-5

EXCHANGEABLE NH_4^+ IN THE PROFILE OF A SANDY SOIL BENEATH A SLUDGE POND AS COMPARED TO AN UNTREATED AREA [41]



1 ft = 0.305 m

The most important mechanism of storage is through plant uptake and subsequent conversion of root and other residues into soil humus. Large quantities of input nitrogen can be stored in soil for long periods of time in this way, particularly in soils of initially low organic nitrogen content. This is illustrated by the profiles of organic nitrogen shown in Figure A-6 for a cropped area near Bakersfield, California, where wastewater had been used to irrigate crops for a periods of 36 years at the time of sampling, compared to an adjacent area of untreated soil that had never been cropped or irrigated. Total nitrogen down to a depth of 5 ft (1.5 m) increased by 7 400 lb/acre (8 290 kg/ha) as a result of wastewater application, representing an average annual increment of 205 lb/acre (230 kg/ha) over the 36 year period.

Lesser quantities of nitrogen would be stored in the organic form in soils of initially higher organic nitrogen content; and in some instances, such as those reported by Sopper and Kardos where apparent crop removals of nitrogen greatly exceeded the quantity applied with the wastewater, net mineralization of soil organic nitrogen will actually decrease the quantity stored [42]. Net immobilization is common on soils of arid regions where there has been little previous input of organic matter. Net mineralization is more likely in soils of more humid regions where the native level of organic matter is usually higher because of more abundant vegetation. Soils of arid regions which have been irrigated for many years would be unlikely to accumulate much additional N during wastewater application.

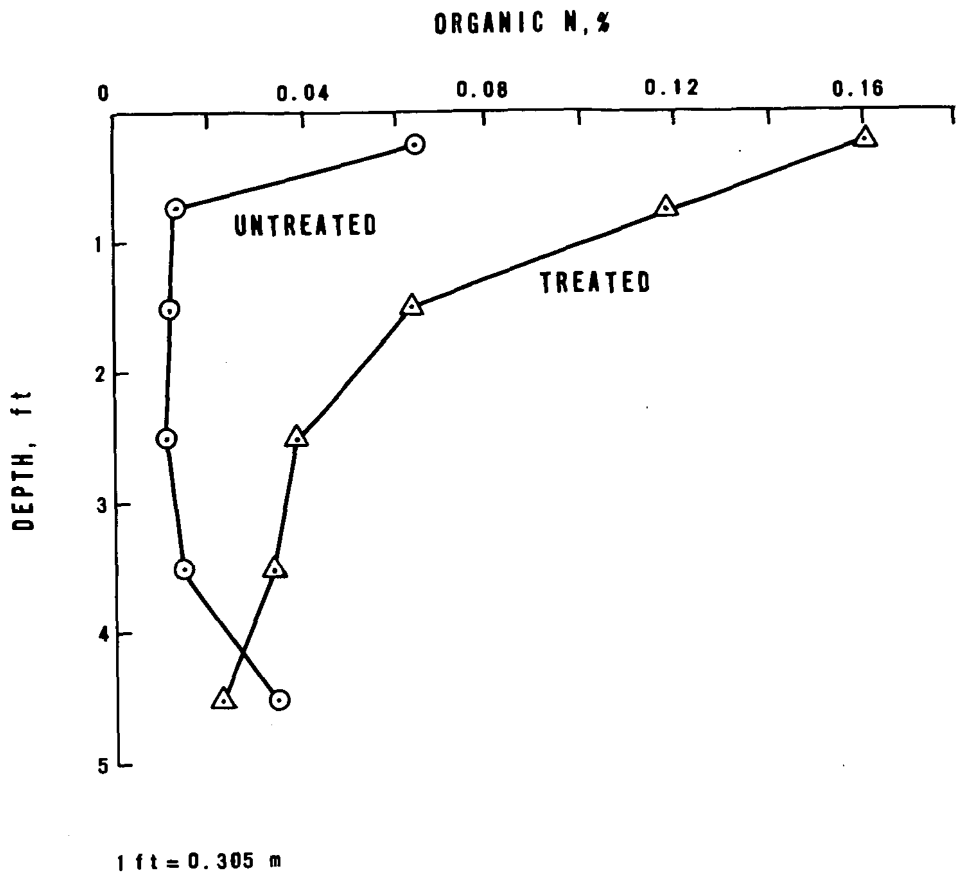
A.4 Nitrogen Removal with Various Application Systems

A.4.1 Slow Rate Systems for Irrigation of Crops

Wastewater used for crop irrigation is commonly applied by sprinklers or ridge-and-furrow distribution systems, with the rate of application geared to the needs of the crop for water and nutrients. Nearly all data on efficiency of nitrogen removal have been obtained at experimental sites. In an EPA survey of facilities using land application of wastewater, nitrate concentrations in groundwater were reported at only 10 of 155 locations using municipal wastewater and at only 2 of 56 locations where industrial wastes were applied [63]. Of the 12 locations with groundwater nitrate data, only 3 reported total nitrogen inputs. In slow rate systems, estimates of system performance based on comparisons of input nitrogen concentrations with nitrate concentrations in groundwater may be very misleading. Salts in the soil solution are concentrated by evaporation and transpiration, particularly in arid regions, or diluted by irrigation and rainfall. Estimates of the leaching fraction may be made by measurement of chloride concentrations in influent and effluent water provided that plant uptake of chloride is insignificant.

FIGURE A-6

EFFECT OF 36 YEARS OF WASTEWATER APPLICATION ON ORGANIC N IN A SOIL AT BAKERSFIELD, CALIFORNIA [41]



Mineralization of organic nitrogen in soils may also contribute appreciably to nitrate that eventually reaches groundwater. This is illustrated by the data of Table A-4 which give total and tagged nitrogen in the effluent from soil columns treated with wastewater in which the input water, applied at 3 in./wk (7.5 cm/wk) contained NH_4^+-N labelled with the ^{15}N isotope. This made it possible to identify the nitrogen in the effluent which was derived from the applied wastewater. The difference in percent recovery of total and tagged nitrogen is due to the contribution of soil nitrogen, most of which was converted from organic forms to nitrate during the period of treatment. Thus, net removal from the Salado fine sandy loam after application of 137 in. (348 cm) of wastewater would be calculated at only 6%, whereas the true removal was 48%. Total N added to this soil in wastewater was equivalent to 1 315 lb/acre (1 473 kg/ha), and the effluent contained

1 236 lb/acre (1 384 kg/ha) of nitrogen; however, only 684 lb (766 kg) was derived from wastewater, the other 552 lb (618 kg/ha) of nitrogen in the effluent being produced by decomposition of soil organic matter. In soils of low organic content, such as the Salado subsoil, this factor is of minor importance, as shown by the close correspondence in the figures for total and tagged nitrogen. At low application rates, nitrogen removal can be completely masked by mineralization of organic nitrogen in the soil, as is illustrated by the Panoche sandy loam. A comparison of nitrogen input versus output shows a net gain, or no removal, whereas in fact 97% of the input nitrogen did not appear in the effluent.

TABLE A-4

RECOVERY OF TOTAL AND TAGGED N IN EFFLUENT FROM THREE SOILS RECEIVING ^{15}N-LABELLED WASTEWATER AT THE RATE OF 3 IN./WK [41]

	Wastewater applied		N recovered in effluent, %	
Soil	in.	lb N/acre	Total	Tagged
Salado fine sandy loam	48	459	24	1.3
	137	1 315	94	52
Salado subsoil	48	459	20	17
	137	1 315	78	75
Panoche sandy loam	87	429	141	2.7

1 in. = 2.54 cm
1 lb/acre = 1.12 kg/ha

If total nitrogen input does not greatly exceed crop requirements for nitrogen, removals of 35 to 60% can be expected as a result of crop uptake. Depending on soil properties and irrigation schedules, denitrification may account for 15 to 70% of the input nitrogen, or even more at low loading rates. In agricultural practice where attempts are made to minimize denitrification, losses of 15 to 30% are common [64, 65]. Denitrification losses with sprinkler irrigation are likely to be lower than with furrow application because the soil is less likely to reach the saturated condition, but this may be balanced out by higher ammonia loss in sprinkler application. Ammonia loss from the soil surface during periods of drying may be a more important consideration than is commonly realized [50].

Normally, in wastewater application to crops, it is desirable to rely primarily on crop uptake as a means of nitrogen removal, and a number of years of field experience indicates that the procedure is effective in both forest and cropland when rates of application are adjusted to soil and crop capacity [42, 66]. The capacity of soil to receive nitrogen may be greatly enhanced by long-term storage in those soils where substantial buildup of organic nitrogen may occur. The Werribee farm in Australia is a case in point [67]. Soil nitrogen increased from 1 200 to 2 620 ppm after 12 years of irrigation with wastewater. Even if it is conservatively assumed that the increase was restricted to the surface 6 in. (15 cm), the additional nitrogen stored is equivalent to about 2 500 lb/acre (2 800 kg/ha) averaging a little over 200 lb/acre·yr (225 kg/ha·yr), which is of the same order-of-magnitude as crop removal. This value is almost identical to the previously cited value maintained over a 36 year period at Bakersfield, California. In the latter instance, however, a much greater depth of soil was implicated in the storage. After 26 years of wastewater application, the Werribee farm showed a surprising drop in nitrogen content, the reason for which is not apparent. Adriano et al. estimated total nitrogen immobilization during 20 years of cannery waste application to sand and loamy sand soils to be as much as 2 700 lb/acre (3 000 kg/ha), accounting for approximately one-third of the total nitrogen applied [68]. The quantity of nitrogen immobilized in a given situation depends somewhat on wastewater composition, being greater with wastewater of high BOD and low nitrogen content, but it is also affected by climatic variables and nature of the soil. With constant management, an equilibrium level of organic nitrogen will eventually be attained, but this may require many years.

Slow rate land treatment provides sufficient nitrogen removal in several reported instances to produce a soil percolate below 10 mg/L of NO_3^--N [42]. Karlen et al. reported reduction of nitrogen content from 15 to 7 mg/L with an annual application rate of 79 in. (200 cm) of wastewater containing 268 lb/acre (300 kg/ha) of nitrogen on corn growing on a loam soil with tile drainage [44]. The maximum weekly rate was 5.3 in. (13.4 cm). McKim et al. in New Hampshire reported total removals of nitrogen ranging from 73 to 91% with primary or secondary effluents applied to grass at 2 and 4 in./wk (5 and 10 cm/wk) [43]. Total nitrogen applications varied from 212 to 426 lb/acre (238 to 478 kg/ha), and the average concentration of nitrogen in the wastewater of about 35 mg/L was reduced to 3 to 10 mg/L in the percolate. In the well-known long-term experiments at Pennsylvania State University, soil solution samples at the 4 ft (1.2 m) depth in Reed canary grass plots receiving 2 in./wk of wastewater consistently showed less than 4 mg/L of NO_3^--N. Application of 2 in./wk to red pine and hardwood plots resulted in soil nitrate concentrations at the 4 ft (1.2 m) depth substantially in excess of 10 mg/L of nitrogen, although with 1 in./wk (2.5 cm/wk) they remained below this value. Kardos and Sopper conclude that, with appropriate management of nitrogen loading rates to maximize crop uptake and with hydraulic loadings adjusted to maximize denitrification, it should be possible to recharge water that meets drinking quality standards for nitrogen into the aquifer below a land treatment site [19].

A.4.2 Rapid Infiltration Systems

Rapid infiltration systems use application rates as high as 360 ft/yr (110 m/yr) and annual nitrogen loading up to 36 000 lb/acre (40 300 kg/ha) on highly permeable soils. Although grass is sometimes grown on the receiving areas, the quantity of nitrogen removed by the crop is only a small fraction of the total applied and exerts little influence on the quality of the percolating water. Much of the quantitative data on rapid infiltration systems is derived from the Flushing Meadows project at Phoenix, Arizona. Lance and Whisler concluded that the only feasible mechanism for removing the large quantities of nitrogen in high-rate applications is denitrification [7]. Bouwer et al. reported that overall nitrogen removal during sequences of long flooding and drying periods was about 30% [8]. Reducing the infiltration rate 50% had the effect of increasing nitrate removal to 80%. Lance published a table showing calculated percentages of nitrogen removal ranging between 75 and 80% using different management systems [62]. The systems involved reduction of the infiltration rate or recycling high-nitrate percolate and mixing it with secondary effluent prior to reapplication. These techniques for achieving high nitrogen removal, although promising, require testing on a field scale before widespread adoption.

In the Santee, California, project, municipal effluent applied to the alluvium of a shallow stream channel undergoes about 10 ft (3 m) of vertical percolation followed by considerable lateral movement underground [69]. Total nitrogen in the renovated water was reduced to 1.5 mg/L, compared to about 25 mg/L in the spreading basins. At Detroit Lakes in Minnesota where about 98 ft/yr (30 m/yr) of effluent was applied by sprinkling on a schedule of 20 hours on and 4 hours off, input nitrogen was converted to nitrate, but little denitrification occurred and nitrate appeared in the groundwater at concentrations equal to the influent [70]. In another system with a loading rate of 45 ft/yr (14 m/yr) where 2 weeks of wetting was followed by 2 weeks of drying, 70% removal of total nitrogen was achieved [71]. At Fort Devens, Massachusetts, where rapid infiltration of primary effluent has been used since 1942, recent data show that where 91 ft/yr (28 m/yr) of wastewater was applied on a schedule of 2 days of flooding followed by 14 days of drying, nitrate-nitrogen concentrations in the groundwater were 20 to 40% of the average total nitrogen input level of 47 mg/L [72].

A.4.3 Overland Flow Systems

Land application of wastewater on fine-textured soils of low permeability has been made possible by development of the overland flow treatment method. The relatively high clay content of such soils is advantageous in nitrogen removal because of their increased capacity for adsorption of ammonium and slow diffusion rates of gases through them, thereby permitting development of an anaerobic zone near the surface. Hoeppel et al. have shown that concentrations of NH_4^+ and NO_3^- in surface runoff are linearly correlated with flowrate, indicating that efficiency of nitrogen removal depends on time of contact between water and the soil surface [57].

In this mode of treatment a ground cover is required, usually a species of grass that is tolerant of wet conditions, such as Reed canary grass. The rates of application in some overland flow systems exceed plant uptake by a substantial margin, but plant uptake undoubtedly plays an important role in nitrogen removal. Carlson et al. reported a pronounced gradient in the growth of grass between the lower and upper ends of the slope in their model, with nitrogen deficiency evident at the lower end, which shows that much of the inorganic nitrogen present was assimilated by the grass, lost to denitrification, or both [58].

In addition to crop uptake, the important processes involved in nitrogen removal during overland flow may include ammonia volatilization, adsorption of ammonium by clays and organic matter, immobilization, and denitrification. Insufficient data are available to evaluate the relative importance of these processes under a particular set of circumstances. Law et al. reported the maximum pH of cannery waste at the Paris, Texas, site was 9.3, while the value in the runoff was 8.1 [26]. At these values, ammonia volatilization could be appreciable. Ammonium adsorption is probably involved in development of a slope gradient in nitrogen available to the grass.

The overland flow system is ideally adapted to the nitrification-denitrification sequence, which requires aerobic and anaerobic zones in close proximity. Applied wastewater is aerated as it contacts the atmosphere as a thin film flowing over the surface, thereby permitting nitrification to occur. Nitrate thus formed diffuses into the soil, encountering reducing conditions in which denitrification can proceed. The presence of living plants provides a mat of organic debris and root excretions which can be used as a substrate by denitrifying bacteria. Conditions are even more favorable for denitrification with wastewater of high BOD, such as cannery effluents. Thomas states that denitrification is the major mechanism of nitrogen removal in overland flow systems [59]. Another aspect of the role of plants is their influence on the loss of nitrogen through the nitrification-denitrification processes in the root zone. Plants capable of surviving in wet environments have a mechanism for translocating oxygen from the tops to the roots, and may even excrete oxygen from the roots. For example, healthy rice roots grown in flooded soil often have a reddish coating due to hydrous oxides of ferric iron, clearly denoting an oxidizing micro-environment even though negative, or strongly reducing oxidation-reduction potentials exist in the soil proper. In the immediate root zone, or rhizosphere, nitrification may occur, after which the nitrate so formed will diffuse away from the site of formation and be denitrified. In support of this view is the observation that nitrogen losses occur in rice soils even when ammonia sources are placed directly in the reducing zone [73].

The overland flow systems for which data are available show high nitrogen removal efficiencies. Law et al. reported 83 to 90% removal of total nitrogen from cannery wastes applied on grassland at the rate of 515 lb/acre·yr (578 kg/ha·yr) of nitrogen [26]. In this case where most of the input nitrogen was organic and the wastewater had a high BOD, it is possible that much of the applied nitrogen was incorporated into the soil organic fraction. Johnson et al. cite 60% removal of total nitrogen from raw sewage at Melbourne, Australia [67]. Hoeppel et al. reported nearly complete removal of NH_4^+ or NO_3^- by a model overland flow system using municipal wastewater on a kaolinitic clay soil [57].

A.4.4. Wetlands

Very little information is available on the use of marshes and wetlands for wastewater treatment, but a consideration of the foregoing discussion on the factors that favor the denitrification process will make it evident that such areas have the requisite characteristics of a nitrogen sink. In marshes and swamps, the rate of plant growth is greater than the rate of decomposition of plant residues as a result of exclusion of oxygen from the surface soil by excess water, since in an anaerobic environment decomposition of plant residues is neither rapid nor extensive. Hence, soils formed under these conditions typically have high organic matter levels, some falling in the peat and muck categories. Abundant organic matter and an aerobic-anaerobic zone at the mud-water interface provide excellent conditions for denitrification. The potential for nitrogen removal is illustrated by consideration of the area of peat and muck soils in the Sacramento-San Joaquin delta area of California. When these soils are drained, aerobic decomposition of the soil above the water table is so rapid that subsidence up to 3 in./yr (7.5 cm/yr) is observed. The organic nitrogen in the soil is mineralized and converted to nitrate, which appears in the drained soil at high concentrations. A subsidence of 2 in./yr (5 cm/yr) represents the release of nitrogen in the inorganic form of about 4 500 lb/acre (5 050 kg/ha). Notwithstanding this enormous input, the drainage waters and groundwater in the area maintain low concentrations of nitrate as a result of denitrification in the saturated zone.

Raveh and Avnimelech have reported substantial enhancement of nitrate removal by sprinkling or flooding soils in the Hula valley in Israel, an area previously covered by a lake and marshes [74]. When the water level in a field was raised to the surface by flooding, the redox potential dropped to about -100 mV throughout the profile, and the nitrate concentration in the top layer dropped rapidly from 1 250 to 250 ppm (soil basis). The quantity of nitrate reduced was 1 650 lb/acre (1 850 kg/ha) of nitrogen, or about 70% of the amount initially present in the top 3 ft (1 m). In another experiment, the soil was wetted by sprinkling for about 20 hours at a rate lower than the infiltration rate so that the surface soil remained unsaturated. In this case, nitrate disappearance from the top 3 ft (1 m) was about 980 lb/acre (1 100 kg/ha). These authors emphasized the importance of surface drying in releasing readily available organic matter, which stimulates oxygen consumption and provides a substrate for denitrifying bacteria. In layers that remained permanently wet, even though the redox potentials were very low, nitrate reduction was negligible.

Engler and Patrick investigated nitrate removal from floodwater in relatively undisturbed cores of a fresh water swamp soil and a saltwater marsh soil in Louisiana [75]. The latter was more effective in nitrogen removal, with an average rate of 8.2 lb/acre·d (9.2 kg/ha·d), while the fresh water swamp soil removed 2.9 lb/acre·d (3.3 kg/ha·d). Addition of organic matter to a rice soil was shown in other experiments to have the effect of decreasing the depth of soil through which nitrate had to diffuse before being reduced, and this drastically increased the rate of nitrate removal.

A.5 Summary

The important processes involved in nitrogen removal from wastewater applied to land are ammonia volatilization, crop removal, soil adsorption of ammonium, incorporation into the soil organic fraction, and denitrification. The relative contribution of individual processes to overall nitrogen removal is dependent on a large number of soil, climatic, and management parameters. While it is not yet possible to predict nitrogen removal in a particular situation with a high degree of confidence, enough is known about the influence of management factors, such as loading rates, flooding and drying periods, and type of plant cover, to design systems that will remove the major part of input nitrogen for a wide variety of disposal requirements and local circumstances.

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APPENDIX B

PHOSPHORUS

B.1 Introduction

Phosphorus (P), in the element form, is a highly reactive material and is thus usually found in nature in an oxidized state in combination with oxygen and a number of mineral elements. It is also found in many organic compounds in naturally occurring materials. Because phosphorus is essential for all forms of life, it must be present in available forms in all soils and waters if these are to be biologically productive. The production of large amounts of biological materials on land surfaces is usually considered desirable because these can be used for food, fiber, fuel, and building materials. In waters, however, the production of large amounts of biological materials usually causes undesirable effects. Thus, on agricultural land, materials containing phosphorus are added to increase biological production; whereas in most waters, attempts are made to keep the phosphorus concentration within low limits to avoid undesirable production of organic materials that cause problems in the use of water for municipal, industrial, agricultural, and recreational purposes.

Concentrations of phosphorus in municipal wastewaters usually range from about 1.0 to 40 mg/L, depending on the phosphorus concentration of the input water and removal during treatment [1-3]. Thomas used 10 mg/L as a typical phosphorus concentration [4]. Most concentrations are usually less than 20 mg/L.

On the other hand, the concentration of phosphorus in the soil solution in most soils is usually between 3 and 0.03 mg/L [5], but typical concentrations are a few tenths mg/L [6]. Because of these differences in ranges of phosphorus concentrations between wastewaters and soil solutions, a reduction in phosphorus concentration as the wastewater enters the soil is to be expected. As a result of various adsorption and precipitation reactions, the concentration of phosphorus will decrease as the wastewater enters the soil, depending on the intensity of these reactions, the capacity of the soil materials to maintain them, and the time allowed for them to proceed. Harvested crops also serve as a sink for the added phosphorus and a certain amount returns to the soil annually in plant materials (roots, stems, and leaves) that are not harvested.

The objectives of this appendix are to discuss the reactions of phosphorus with soils, to show their applications to the removal of phosphorus from wastewaters applied to soils, and to assess the present status of our abilities to predict the capacities of soils to remove phosphorus from such waters. The chemistry of phosphorus in soils, plants, and waters is complex, and the literature is voluminous. Consequently, no attempt has been made to provide a complete literature review. Reports and textbooks that do review the literature are available, including Russell [5], Tisdale and Nelson [7], Larson [6], Holt et al. [8], and Ryden et al. [9].

B.2 Removal Mechanisms

The phosphorus that enters a soil in fertilizers, wastes, or wastewaters is (1) removed in harvested crops; (2) accumulated in the solid phase of the soil as organic compounds, adsorbed ions, or precipitated inorganic compounds; (3) removed by soil erosion as soluble phosphorus or phosphorus adsorbed or precipitated on soil particles; or (4) leached from the root zone in percolating water. The chemical reactions between added soluble phosphorus and soil materials or sediments influence the availability of phosphorus to crops and the desorption or solubility of phosphorus when the soil materials become sediments in streams and lakes, and control the leaching of phosphorus through soil profiles.

The amount of phosphorus in soils is usually between 0.01 and 0.2%, but heavily fertilized surface soils can contain greater amounts. Usually much less than 0.1% of the total phosphorus in soils is soluble in water. Solid phase phosphorus consists of (1) organic phosphorus, the quantities of which are highly dependent on the amount of organic matter in the soil; (2) inorganic compounds; and (3) phosphorus adsorbed on various types of surfaces in the soil. The orthophosphate form, in which one phosphorus atom is combined with four atoms of oxygen, is the most stable configuration in the soil environment.

In discussing the reactions of phosphorus with soils and sediments, it is assumed that the phosphorus is in the orthophosphate form and that other forms convert to this form in the soil system [10-13]. The main soil constituents that react with phosphorus at concentrations usually found in wastewaters are (1) iron and aluminum as soluble ions, oxides and hydroxides, and silicates; and (2) calcium as a soluble ion and as carbonate.

Soluble inorganic phosphorus introduced into a soil is chemically adsorbed on surfaces and can also be precipitated. In the adsorption processs, the reaction is with iron, aluminum, or calcium ions exposed on solid surfaces. Reactive iron and aluminum surfaces can occur at the broken edges of crystalline clay minerals, as surface coatings of oxides or hydroxides on crystalline clays, and at the surfaces of particles of oxides and hydroxides and of amorphous silicates. Aluminum in the form of positively charged hydroxide polymers and as an exchangeable ion in acid soils can also adsorb phosphorus. Reactive calcium surfaces are mainly found on solid calcium carbonates and calcium-magnesium carbonates. Precipitation reactions occur with soluble iron, aluminum, and calcium. Particles of phosphate compounds can also form by separation of adsorbed phosphorus along with iron, aluminum, or calcium from solid surfaces.

The reactions of phosphorus with soils are complex, and the soil system is complex. Consequently, the uncertainty whether phosphorus is being adsorbed or precipitated leads to the use of the term "sorption," which covers both processes and means only that the phosphorus has been removed from solution.

A given soil material does not have a fixed capacity to sorb the phosphorus added in wastewaters. Sorption is dependent not only on the concentration of phosphorus in solution, but also on a number of factors, including soil pH, temperature, time, the total amount of phosphorus added, and the concentrations of various constituents in the wastewater that directly react with phosphorus or that influence such soil properties as pH and oxidation-reduction cycles. Another basic factor is that the downward movement of phosphorus in a soil profile is diffuse. Because the capacity for sorption of phosphorus is concentration-dependent, there is a large transition zone between highly enriched and nonenriched soil, which is described as a diffuse rather than as an abrupt boundary as illustrated in Figure B-1. That is, a given depth interval gradually accumulates sorbed and soluble phosphorus, and the breakthrough curve at the bottom of a soil column extends over considerable time and/or volume of effluent.

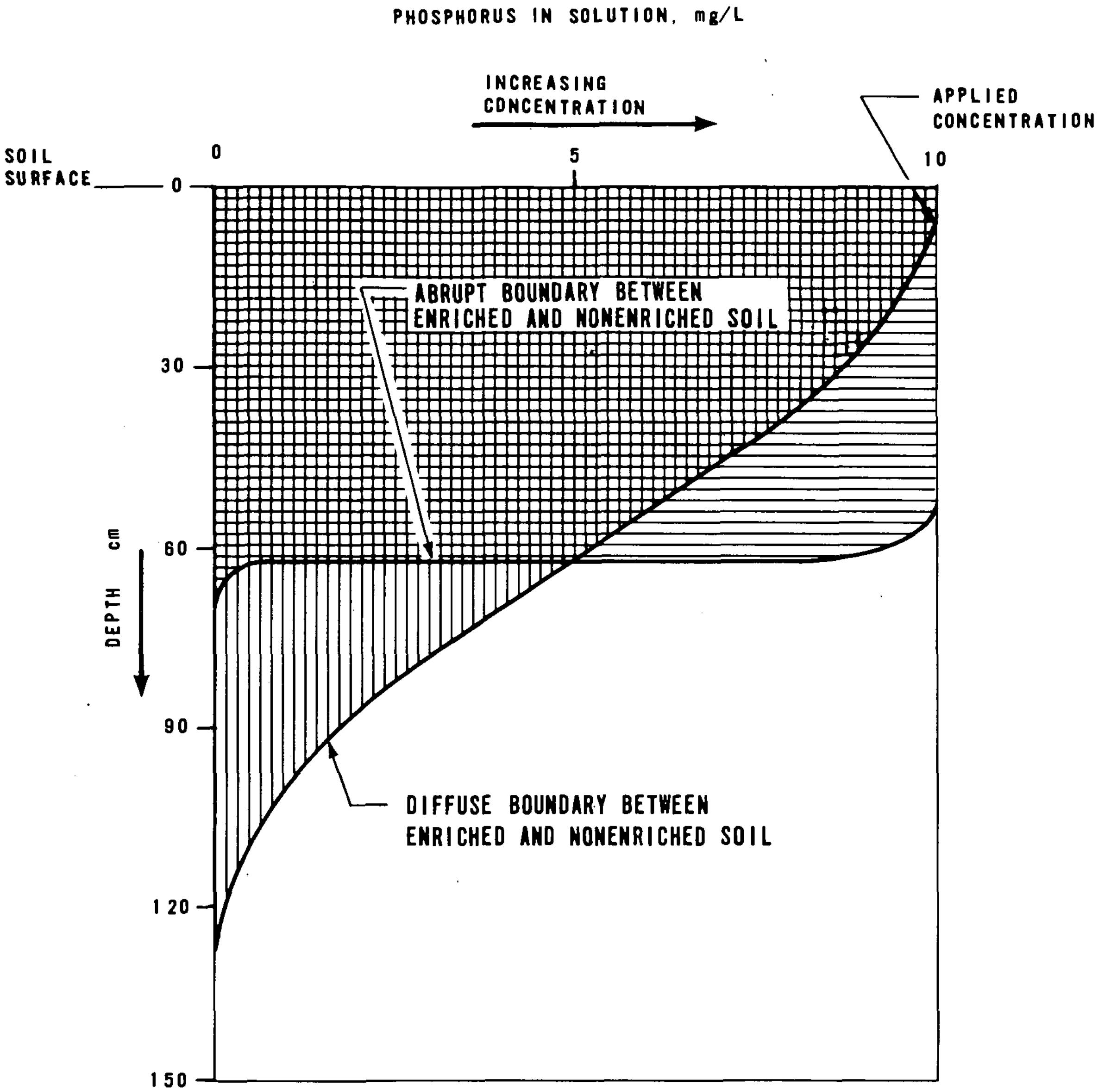
B.2.1 Crop Removal

In most cropped soils, the application of phosphorus increases growth of plants. However, as more phosphorus is accumulated (i.e., excesses are added), negative effects are sometimes found. These decreased yields that result from excess available phosphorus in the soil are indirect effects of phosphorus on the availability of copper, iron, and zinc and are referred to as nutrient imbalances [14-17]. Corrections of these imbalances can be made by soil or foliar applications of the needed elements.

The removal of phosphorus in harvested crops depends on the yield and the phosphorus concentration in the harvested material, which in turn are dependent on the crop, soil, climate, and management factors, including the amount of phosphorus added to the soil. Typically, the harvested portions of annual crops contain only 10% or less of the fertilizer phosphorus added during the season in which the crop is grown, but recoveries as high as 50 to 60% are possible [5]. However, recovery is low not only because the soil reacts with the added phosphorus to make it less available, but also because plants absorb considerable amounts of phosphorus from soil supplies, including the residues from applications in previous years. Thus, the total removal per year as a fraction of the total added per year is more important than the recovery of that added during the year the crop is grown.

FIGURE B-1

RELATIONSHIPS BETWEEN PHOSPHORUS CONCENTRATION AND SOIL DEPTH FOR ABRUPT AND DIFFUSE BOUNDARIES BETWEEN ENRICHED AND NONENRICHED SOIL



Data for amounts of phosphorus removed by the usual harvested portions of selected agronomic, vegetable, and fruit crops are presented in Table B-1. Variations range from as low as 10 lb/acre·yr (11 kg/ha·yr) to as high as 85 lb/acre·yr (95 kg/ha·yr).

TABLE B-1

REMOVAL OF PHOSPHORUS BY THE USUAL HARVESTED PORTION OF SELECTED CROPS

Crop	Annual crop yield, per acre	Phosphorus uptake, lb/acre·yr
Corn [18]	180 bu	31
Cotton [18]		
Lint and seed	3 700 lb	17
Wheat [18]	80 bu	20
Rice [18]	7 000 lb	20
Soybeans [18]	60 bu	22
Grapes [18]	12 tons	10
Tomatoes [18]	40 tons	30
Cabbage [18]	35 tons	16
Oranges [18]	600 boxes (90 lb/box)	10
Small grain, corn-hay rotation [19]	....	29
Reed canary grass [19]	....	40
Corn silage [19]	....	27-36
Poplar trees [20]	....	23-62
Barley-sudan grass rotation for forages[a]	....	75-85
Johnson grass [18]	12 tons	84
Guinea grass [18]	11.5 tons	45
Tall fescue [18]	3.5 tons	29

a. Unpublished data for barley in the winter followed by sudan grass in the summer. P.F. Pratt and S. Davis, University of California, and USDA-ARS, Riverside, California.

1 lb = 2.2 kg
1 acre = 0.405 ha

Amounts of phosphorus removed by crops can range by an order of magnitude and are higher with forage crops than with most other crops. Removal in harvested materials, as a fraction of that added, decreases as the amount of phosphorus added increases. Where double cropping during a long season is possible, removal can be nearly double that where only one crop can be grown.

B.2.2 Adsorption

When solutions containing soluble phosphorus at concentrations usually found in reclaimed waters (20 mg/L or less) are added to soils, the initial (and more rapid) reaction can be described by the Freundlich or Langmuir equations. Slow reactions, such as precipitation, are not modeled by these equations so that their use will yield conservative results. Olsen and Watanabe found that, up to an equilibrating concentration of nearly 20 mg/L, using a reaction time of 24 hours, the reaction was described by the Langmuir equation [21]. That is, if a water containing any specific concentration of phosphorus within the range of a few to 20 mg/L is added to a soil and allowed to equilibrate for 24 hours, phosphorus will be sorbed by the soil and the concentration in solution will decrease. If a number of waters containing various phosphorus concentrations are added to samples of the same soil, a relationship between phosphorus sorption and final phosphorus concentration can be described by the Langmuir equation. From this equation the decrease in phosphorus concentration and the sorption of phosphorus can be predicted for any other initial concentration using the same soil and reaction time. But at equilibrating concentrations greater than 20 mg/L, the reactions in most soils are not described by this equation.

Larson concluded that, in dilute solutions, adsorption generally follows the Langmuir equation where a plot of C/V against C (where C is concentration and V is phosphorus adsorbed per unit weight of soil) gives a straight line [6]. Larson reported that a study of 120 soils showed straight line relationships of C/V to C up to concentrations of about 19 mg/L of phosphorus. Above this concentration the C/V-C line curved, indicating that the adsorption equation was no longer valid. At higher concentrations, the concentration was assumed to be limited by the formation of sparingly soluble compounds, and the value V increased as more of these compounds were formed.

Ellis [13] and Ellis and Erickson [22] used the Langmuir equation to calculate relative capacities of soil profiles to retain phosphorus. The retention of phosphorus at a solution concentration of 10 mg/L ranged from 71 to 95% of the adsorption maximum calculated from the Langmuir equation for a number of soil materials. Amounts retained from a solution concentration of 10 mg/L ranged from 77 to 1 898 lb/acre (86 to 2 126 kg/ha) for 12 in. (30 cm) depth intervals, respectively, for a dune sand to a clay loam. The reaction time in these studies was 24 hours.

Even though the original or initial capacity to retain phosphorus can be described by adsorption equations, the retention increases as a function of time so that the initial retention is only useful if the ratio of the slow reaction to the initial reaction is known for each soil. The slow reaction involves the formation of precipitates of limited solubilities and the regeneration of adsorptive surfaces. Crystallization of precipitates also reduces their solubilities. Thus, there is small probability that Langmuir adsorption equations will be generally useful in predicting quantities of phosphorus that will react with soils over periods of months or years.

B.2.3 Precipitation

The dominant precipitation reactions in soils are with calcium, iron, and aluminum ions. Reactions of phosphates with iron and aluminum are not completely identical, but they are sufficiently similar that for some parts of this discussion they are considered together.

Qualitative and quantitative determinations of definite compounds or minerals of phosphorus in soils are difficult. Empirical extraction techniques, such as that of Chang and Jackson [23], have been used to semiquantitatively differentiate among calcium, aluminum, iron, and organic phosphates. More definite determinations of specific compounds have been made [24-28], but these are qualitative determinations of reaction products formed from high concentrations of soluble phosphorus usually near simulated fertilizer bands. Another approach is to study the formation and stability of phosphorus compounds in solutions and then assume that the same compounds form in soils under similar chemical conditions, or to study phosphorus reactions with relatively pure solids and assume that the same reactions occur in soils. These various approaches lead to the same generalities concerning phosphorus compounds in soils.

The dominant factor that determines whether calcium phosphates or iron and aluminum phosphates form in soils is the pH. The phase diagrams presented by Lindsay and Moreno for a number of phosphorus compounds suggest that calcium compounds predominate above pH 6 to 7, and iron and aluminum compounds predominate below pH 6 to 7 [29]. The exact pH cannot be specified without knowing the calcium ion activity and the calcium phosphate species that is controlling the solubility of phosphorus. Larson stated that, as the pH decreases, a level of acidity is found at which calcium phosphates can no longer control phosphorus solubility, and he suggested that this lower limit might be pH 5 [6]. This limit might be found if fluoroapatite is the calcium phosphate controlling the phosphorus concentration in solution, whereas when hydroxyapatite or octocalcium phosphate is the controlling compound, the pH limit would be near 6. When dicalcium phosphate dihydrate is the controlling calcium compound, the pH limit would be near 7. For these limits, it is assumed that iron and aluminum are present in the soil and compete with calcium for control of the phosphorus solubility. The partial pressure of carbon dioxide in the soil can influence the activity of calcium ions and thus exert an effect on a calcareous system.

The usual concentrations of iron and aluminum ions in solution, in the acid pH range where iron and aluminum phosphates may form, are so low that the direct precipitation of iron and aluminum phosphates is unlikely. Concentrations of iron and aluminum in the soil solution of moderately acid to slightly alkaline soils, pH 5.5 to 8.0, are in the range of a few µg/L or in the parts per billion range. An exception to this statement can be found in highly acid soils containing exchangeable aluminum in sufficient quantities that direct precipitation of aluminum phosphate might occur. Exchangeable aluminum, measured by extraction with a potassium chloride solution, in excess of about 20 ppm in the soil can be expected to cause a direct precipitation of phosphorus as amorphous aluminum phosphate. A more general pathway for formation of iron and aluminum compounds, when dilute solutions of phosphorus are added, is that the phosphorus first reacts by adsorption on surfaces containing reactive iron and aluminum, followed by a breaking away from the surface to form amorphous forms of strengite (iron phosphate) or variscite (alluminum phosphate), which then slowly crystallize into more ordered and less soluble forms of these compounds.

There is thermodynamic evidence that the phosphorus adsorbed on iron surfaces is stable in the well-aerated soils, whereas surface films of phosphate on aluminum surfaces are not [5]. This suggestion is that aluminum phosphates break away from surfaces, exposing a new surface to continue the adsorption process, whereas iron phosphates do not follow this pattern. Thus, well-aerated soils containing dominantly aluminum materials would have much higher capacities to retain phosphorus than soils containing dominantly iron materials. Taylor et al. found that iron materials were much less important than aluminum materials in the initial reactions of ammonium phosphate with soils [30, 31]. If soils undergo alternate cycles of oxidation and reduction, surface iron phosphates are more unstable than those of aluminum because of cycles of reduction of iron to the ferrous form and oxidation to the ferric form.

In contrast to the situation with iron and aluminum, for which concentrations in the soil solution are usually in the µg/L range, calcium concentrations are in the mg/L range. Concentrations of 10 to 200 mg/L are common. In neutral and alkaline soils irrigated with wastewaters, calcium concentrations are likely to be sufficiently large that calcium phosphates will precipitate directly. Under alkaline conditions and calcium concentrations of 20 to 200 mg/L in the soil solution, dicalcium phosphate dehydrate can precipitate directly from solution, depending on the pH and the phosphorus concentration. This compound then redissolves and the less soluble octocalcium phosphate forms. With more time the octocalcium phosphate is converted to the less soluble hydroxyapatite.

Another significant difference between the iron and aluminum phosphate system and the calcium phosphate system, relative to the application of wastewaters to soils, is that these waters may have only traces of iron and aluminum, but they usually have substantial amounts of calcium. The reactions with iron and aluminum are thus limited to the supplies of these in the soil or sediment, whereas the water may supply its own calcium, setting up a system that can precipitate calcium phosphates indefinitely. In calcareous soils the calcium from calcium carbonate can also be a highly significant source for precipitation of phosphorus over an extended period of time.

The reactions of phosphorus with organic soil are qualitatively the same as in mineral soils, but the capacities for sorption are usually much smaller. Many organic soils have small quantities of iron and/or aluminum and calcium and thus are not highly suitable for removal of phosphorus from wastewaters. There are organic soils that have accumulated iron and aluminum materials that have relatively large capacities to sorb phosphorus and there are calcareous organic soils that will retain phosphorus. However, as a general rule, mineral soils will be more suitable for removal of phosphorus from wastewaters.

B.2.4 Reaction Rates

The initial adsorption of phosphorus from dilute solutions is rapid. An apparent equilibrium is attained in a few days [32-35]. But, following this apparent equilibrium, there are slow reactions that continue for months or years [10, 36-47]. Ellis and Erickson found that most soils recovered their sorptive capacities in about 3 months [22]. Kao and Blanchar found that the adsorptive capacity of the Mexico soil of the Sanborn field at Columbia, Missouri, had changed little after 82 years of phosphate fertilization [48]. Barrow and Shaw found that the rate of the slow reaction decreased dramatically as the temperature decreased [35].

This slow reaction is perhaps mostly the result of the precipitation and crystallization of highly insoluble compounds, such as the conversion of dicalcium phosphate dehydrate to octocalcium phosphate or hydroxyapatite, and the exposure of fresh adsorptive surfaces where previously adsorbed phosphates slough off from surfaces of soil particles. The relationship between adsorption and precipitation can be illustrated by the equilibrium reaction [6]:

$$\text{P adsorbed} \rightleftarrows \text{P in solution} \rightleftarrows \text{P precipitated} \qquad \text{(B-1)}$$

If soluble phosphorus is added or removed, the immediate reaction is with the phosphorus adsorbed, but at equilibrium; the precipitated forms control the phosphorus in solution. Equilibrium is attained very slowly, however, and under conditions of irrigation with wastewaters where phosphorus is added periodically if not continuously, the reaction will be to the right, and the system will be continuously in a state of disequilibrium. DeHaan reported that the adsorptive capacity of a soil was too small to account for the large amount retained and suggested that adsorption occurred during the application stage and that precipitation took place during the resting stage with a regeneration of the adsorptive capacity [49].

B.2.5 Leaching

The leaching of phosphorus from the root zone of a cropped land area or from the surface soil material in any wastewater treatment project depends on the amount of water that moves across the boundary being considered and the phosphorus concentration in that volume. The amount leached may be calculated as follows:

$$\text{Amount leached} = 0.225\ WC_p \quad \text{(U.S. customary units)} \qquad \text{(B-2)}$$
$$\text{Amount leached} = 0.1\ WC_p \quad \text{(SI units)} \qquad \text{(B-2a)}$$

where amount leached is in lb/acre·yr (kg/ha·yr)
W = water that moves across the boundary, in./yr (cm/yr)
C_p = concentration of phosphorus, mg/L

Because volumes of percolating water are small and concentrations are low, the downward movement of phosphorus in croplands is usually a very slow process. If 12 in. (30 cm) of water percolates past a given boundary in the soil profile and the concentration of phosphorus in this water is 0.2 mg/L, as might be the case in well-fertilized fertile soils, the amount of phosphorus leached is 0.54 lb/acre (0.6 kg/ha). The amounts of phosphorus absorbed by plant roots from soil depths beneath the zone of incorporation of added phosphorus more than balance this amount. Thus, under usual fertilizer practices in agricultural lands, the net leaching of phosphorus is usually very small.

However, in rapid infiltration systems where large volumes of water move through the soil per year, the quantities of phosphorus that leach can be orders of magnitude higher than is usual for croplands. Under these conditions of high rates, the limiting factor is the solubility of phosphorus, which is controlled by the capacity of the soil to retain phosphorus (i.e., adsorption, precipitation, and reaction rates).

B.3 Phosphorus Removal by Land Treatment Systems

Land application has been used for centuries, and there are hundreds of systems in use in the United States today. Although soil phosphorus, the reactions of fertilizer phosphorus with soils, and the movement of phosphorus with surface flows and with leaching waters have been the subject of many reports during the past few decades, studies of the behavior of phosphorus in land application systems have been initiated only in the past few years. Thomas reported in 1973 that historically the effects of land treatment approaches on plant life, soils, and groundwater had not received much attention [50]. Thus, because technical questions dealing with the behavior of phosphorus during wastewater applications to lands have been asked only recently, there are very few reports on phosphorus retention by soils in land application systems.

B.3.1 Slow Rate Systems

Slow rate systems, as defined in Chapter 2, are those in which total wastewater applications range from 2 to 20 ft/yr (0.6 to 6.1 m/yr) at weekly rates of 0.5 to 4 in. (1.2 to 10 cm). A vegetative cover is an integral component of the system and can utilize phosphorus for crop growth in accordance with typical values given in Table B-1. Because the application rates are similar to those studied in agricultural systems, much of the information gained from agricultural study is applicable to slow rate treatment systems. Even though phosphorus is removed from solution rapidly in slow rate systems by adsorption and precipitation, it is useful to quantify these numbers for engineering design purposes. Wastewater applications are usually limited by nitrogen or hydraulic considerations on a short-term basis, but phosphorus application may be a limiting factor over the life of the system.

For the purposes of this design manual, it is useful to know the net phosphorus application to the soil, i.e., the quantity of phosphorus applied in the wastewater after the removal by crops is considered. This value is useful in estimating the life of system in accordance with the empirical model presented in Section B.4.4.

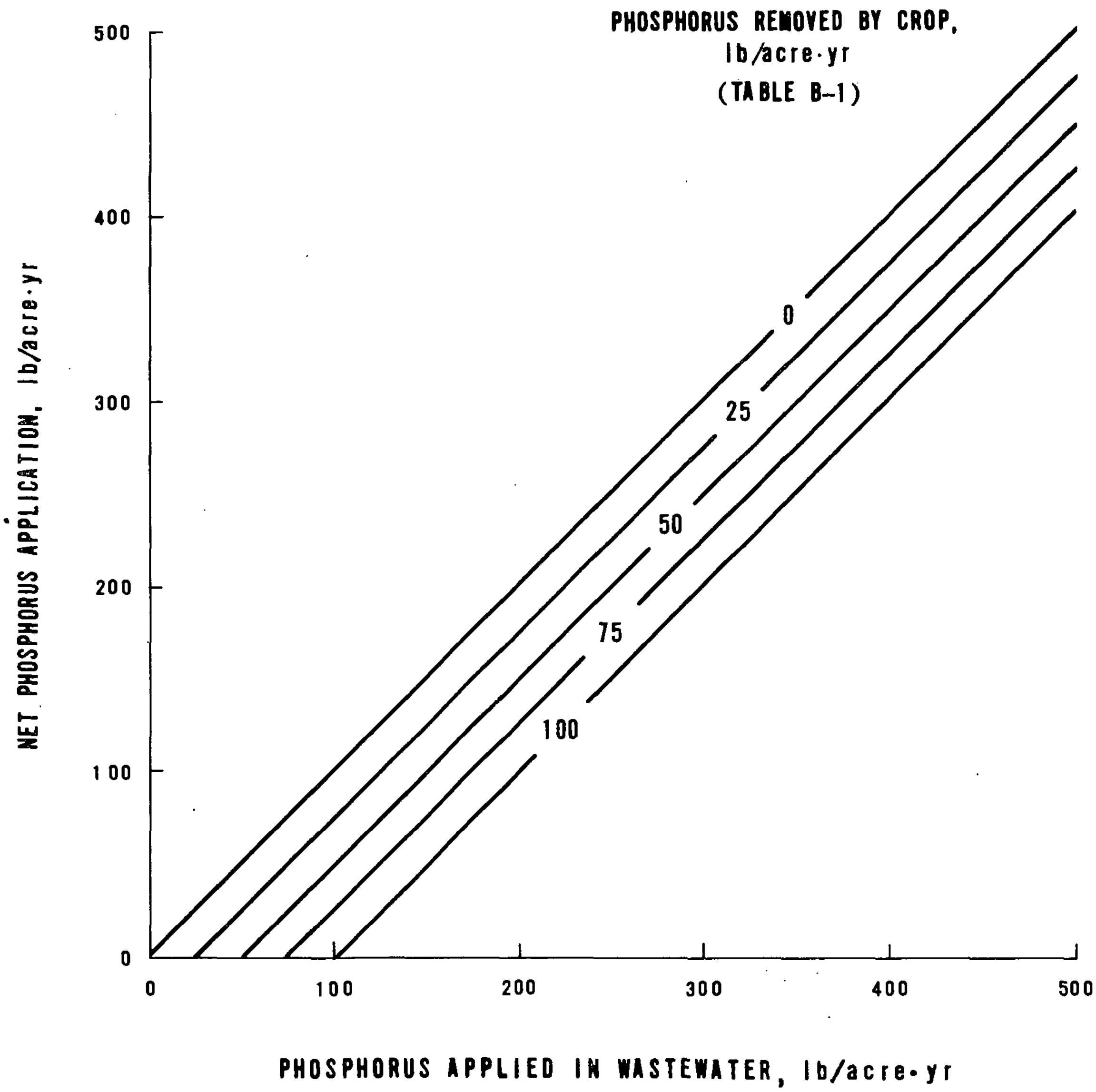
Crop removal as a factor in predicting a net application of phosphorus on land is illustrated graphically in Figure B-2, which shows the relationships among crop removals in pounds per acre per year, total phosphorus applications, and the net application to soil.

The net application to the soil is important in estimating phosphorus sorption as a prediction of the life of the system to retain phosphorus. An empirical model uses Figure B-2 as input into computing an estimate of long-term phosphorus retention.

Because of the similarity of slow rate systems to usual practices on croplands, much of the information obtained on the behavior of phosphorus in crop production is applicable. A number of studies have shown that the retention of phosphorus near the place of its incorporation into soils is high, i.e., the movement is slow, except in acid sandy soils and in acid organic soils containing only small amounts of iron and aluminum [5, 19, 51-56]. In addition, the transfer of phosphorus from land areas to streams, for lands protected from excessive soil

FIGURE B-2

NET PHOSPHORUS APPLICATION
TO THE SOIL



1 lb/acre = 1.12 kg/ha

erosion, is usually less than 1.0 lb/acre·yr (1.1 kg/ha·yr) [8, 9, 57-61]. These small amounts are insignificant in terms of the efficiency of use of phosphorus by plants and they are small when expressed as a percent of the phosphorus sorbed by the soil. In terms of the quality of the drainage water, however, these small amounts of phosphorus can be signficant, as illustrated in Figure B-3. If phosphorus concentrations greater than 0.030 mg/L are conducive to algal blooms in lakes and streams, some streams that contain mainly drainage, including both surface runoff and subsurface drainage, should have sufficient phosphorus to support algal blooms [8]. Of course, in many streams, runoff from forested areas containing very low concentrations of phosphorus dilutes the drainage water from croplands [8], and in some cases, sediments eroded from stream banks and nonfertilized soils act as phosphorus sorbing agents to reduce the soluble inorganic phosphorus in the stream [61, 62].

The optimal plans for a slow rate system should involve (1) a forage crop that removes large amounts of phosphorus, (2) erosion prevention to eliminate surface runoff, and (3) a long pathway consisting of sorptive materials between the surface soil and the point of discharge of the water so that concentrations of phosphorus are reduced to low levels, depending on the intended use of the water. A sufficient pathway length might be 6 ft (2 m) in clayey soils, but greater lengths should be required for sandy or silty soils.

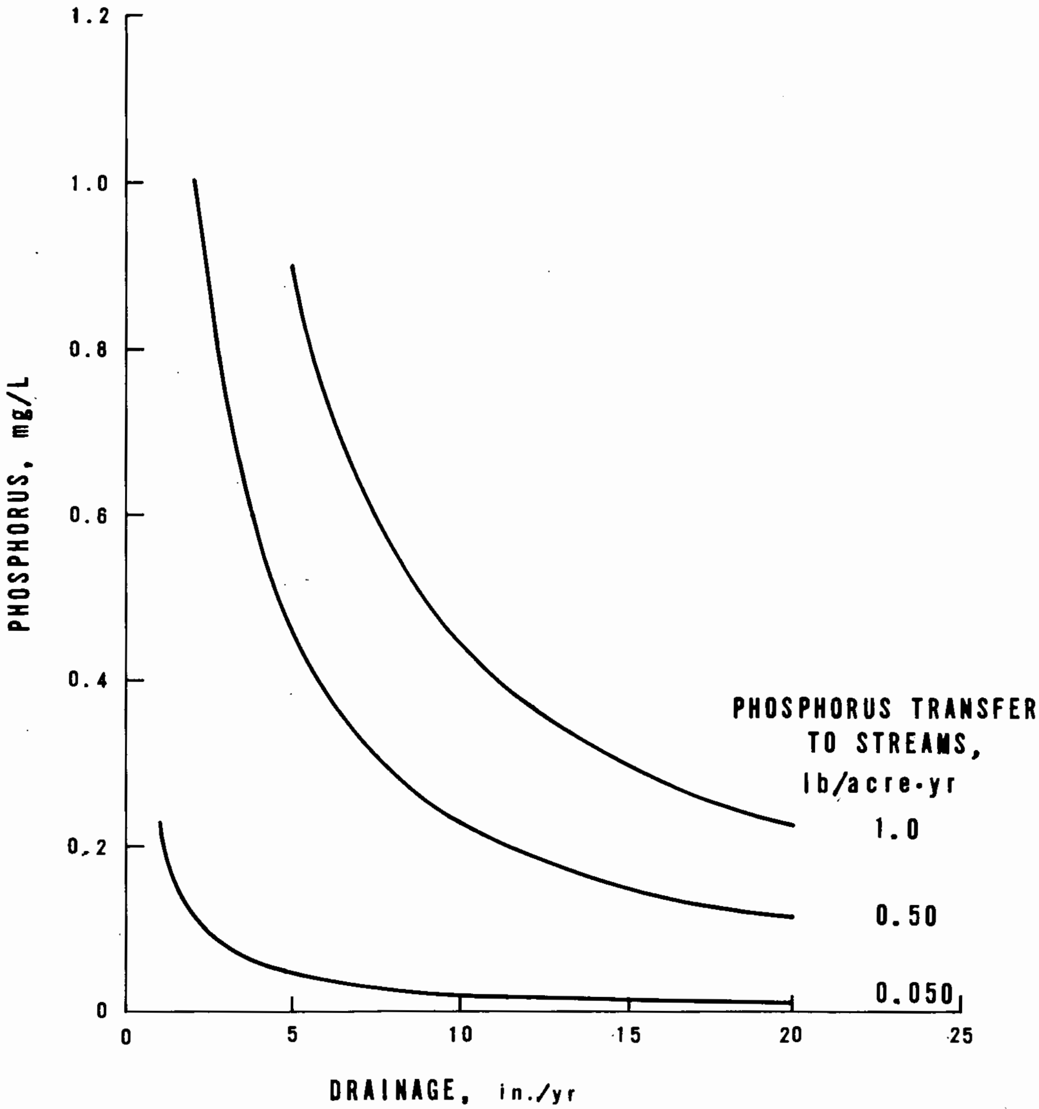
B.3.2 Rapid Infiltration Systems

Rapid infiltration systems, as described in Section 2.3, are those in which the wastewater is applied at annual rates of 20 to 560 ft (6 to 170 m), and weekly rates of 4 to 120 in. (10 to 300 cm). Vegetation may be grown on the surface of the basins, but since the typical applications range from 550 to 15 000 lb/acre (616 to 16 800 kg/ha) of phosphorus, at a concentration of 10 mg/L, the crop uptake is not a significant part of the phosphorus budget. The removal mechanisms of interest are based on the sorption capacities of the soil. The chemical composition of the wastewater is also important because the compounds of iron, calcium, and aluminum, and pH are important in precipitating the phosphorus from soil solution.

Greenberg and Thomas reported phosphorus retention by a Hanford sandy loam during a rapid infiltration system in which water application was about 0.5 ft/d (0.6 m/d) [63]. The phosphorus concentration was about 2 to 3 mg/L in the final effluent added to the infiltration basins. During about 2 years of operation, all of the phosphorus added was retained in the surface foot of soil. The calcium and bicarbonate concentrations in the water were sufficiently high that the soil would have become alkaline, and phosphorus sorbed could have been largely converted to calcium phosphates.

FIGURE B-3

RELATIONSHIP AMONG PHOSPHORUS CONCENTRATION, DRAINAGE FLOW, AND THE AMOUNT OF PHOSPHORUS TRANSFER FROM LAND TO STREAMS



1 lb/acre·yr = 1.12 kg/ha yr

1 in./yr = 2.54 cm/yr

Perhaps the most definitive report on phosphorus in rapid infiltration systems has been done in the Flushing Meadows project [1, 64]. The phosphorus concentration in the wastewater averaged 15 mg/L in 1969 but decreased to about 10 mg/L for the 1970 to 1972 period. Phosphorus removal was increased with an increase in travel distance which was related to time. A travel distance of 30 ft (9 m) removed about 70% of the phosphorus in 1969. The removal was reduced to about 30% in 1970 because of a substantial increase in flowrate, and it was increased to 50% in 1972. With a flow distance of 330 ft (100 m), the phosphorus removal was about 90%. The effluent at this distance had a phosphorus concentration less than 1 to 3 mg/L in the 1971 to 1972 period, and the reduction was greater with an even longer travel distance. After 5 years of operation of this system and phosphorus additions of nearly 43 000 lb/acre (48 000 kg/ha), the removal efficiency was rather stable. The phosphorus removal mechanism in this coarse gravelly soil was precipitation of calcium phosphates.

The phosphorus removal in the Flushing Meadows project is entirely satisfactory for reuse of water for irrigating crops even at short travel distances. The removal to the level (less than 0.03 mg/L) where phosphorus would limit biological production in lakes is not definite in this rapid infiltration system at the high rates with this soil material. Perhaps, with time for attainment of an equilibration with hydroxyapatite at high calcium concentrations and alkaline pH values, this concentration would be obtained, but the required time is not known.

Rapid infiltration systems naturally require coarse gravelly soils that can sustain high infiltration rates at the surface and also high transmissivity from the point of infiltration to the point of discharge into surface waters or into wells. This means that no layers with high sorptive capacity for phosphorus are likely to be encountered. What little capacity there is will soon be saturated, and the retention will then depend on precipitation reactions. The most logical precipitant is the calcium supply in the wastewater. If that supply is insufficient, application of a calcium supply may be considered if removal of phosphorus is deemed to be necessary for future uses of the waters.

Rapid infiltration systems naturally use a cycle of flooding and drying to maintain the infiltration capacity of the soil material, and in some cases, to control insect pests in surface applied waters. Therefore, these rapid infiltration systems can be considered to use flooded soils. These cycles of reduction and oxidation can increase the phosphorus retention capacity of the soil if considerable iron is present in the soil or in the wastewater. Reduction during flooding and oxidation during drying increase the reactivity of the sesquioxide fraction of the soil, increase the phosphorus sorbed, and decrease the phosphorus solubility [65]. Of course, most rapid infiltration systems will use coarse gravelly soils for which the calcium phosphate chemistry will be the critical factor; iron and aluminum will have minor effects because of the very low surface area of the soil material and the limited number of sorption sites on iron and aluminum surfaces will soon become saturated.

B.3.3 Overland Flow Systems

Overland flow systems are used where the soil is slightly permeable and treatment occurs by biological, chemical, and physical reactions on the soil surface (Section 2.4). A large portion of the applied wastewater is collected as treated runoff at the toe of the slope. Since the wastewater flow is predominately on the soil surface, the soil contact is less than for slow rate and rapid infiltration systems. As such, the phosphorus removal may be less for overland flow systems, although combinations can be used to achieve treatment alternatives (Section 3.3). The usual reductions in phosphorus concentrations in the wastewater have been 35 to 60% with overland flow systems [67-69]. Thomas et al. found that the phosphorus concentration was reduced from 10 to about 5 mg/L of total phosphorus in an overland flow system [69]. However, applications of aluminum sulfate at a concentration of 20 mg/L to the wastewater reduced the phosphorus concentration of the treated water to about 1 mg/L for a 90% removal of the total phosphorus input.

Data from Carlson et al. show that the phosphorus concentration decreased at a fairly uniform rate as wastewater ran over overland flow plots [70]. The phosphorus concentration in the effluent water was 40 to 60% of that in the applied water. However, water that percolated through the soil had only traces of phosphorus for nearly complete removal. The harvested grasses removed less than 10% of the applied phosphorus. The recommendation for more effective removal of phosphorus in the overland flow effluent was to obtain more contact between the water and soil surfaces by increased soil roughness or by increased flow path.

B.3.4 Wetland Systems

Although wetland systems have only been studied recently as a means of wastewater treatment, the principles behind phosphorus behavior under such conditions are known. Sufficient research has been completed on flooded rice culture that the behavior of phosphorus in flooded soils is fairly well understood. Also, recent reports have added considerable information on the chemistry of phosphorus in lake sediments.

When soils are flooded with a few feet of water, biological activities in the soil deplete the available oxygen, and the soil becomes anaerobic or, more specifically, anoxic (lack of oxygen). The water usually remains aerobic, and a transition zone between aerobic and anaerobic conditions develops in the immediate surface of the soil. The surface of this transition zone is aerobic and oxidized, whereas the bottom is anaerobic and reduced. The thickness of the oxidized part of this transition layer can vary from about 0.1 to 1 in. (0.25 to 2.5 cm) or more, depending on the rate of supply of oxygen to the surface of the soil and the rate of consumption of oxygen in the lower soil depths [65]. In wetland situations where there is seasonal flooding followed by drying, such as in rice production, the soil goes through seasonal or yearly cycles of reduction and oxidation that result from cycles of anoxic and oxic conditions. Even in wetlands that are not seasonally flooded but are wet because of cycles of inputs of water, alternate periods of reduction and oxidation occur to some degree. Thus, one feature associated with all types of wetlands is the occurrence of reduced conditions or cycles of reduction and oxidation.

When soils and sediments become anoxic, most show an increase in soluble phosphorus [65, 71-73]. This increase in soluble phosphorus was found to be greatest with alkaline soils and with soils that have low iron contents, and it was found to be lowest with acid soils with high iron contents [71]. Some acid soils with high iron contents show no increase or decrease in soluble phosphorus as reducing conditions develop. Patrick and Mahapatra stated that the possible mechanisms for release of soluble phosphorus principally involved the reduction of iron from the ferric to the ferrous state with a release of phosphorus from ferric phosphates and the hydrolysis of iron and aluminum phosphates [65]. In soils with large amounts of iron oxide and iron hydroxide surfaces or aluminum oxides, the net result of reduction is a decrease in phosphorus solubility because of secondary precipitation of the dissolved phosphorus on surfaces that become more reactive when the soil is reduced.

When phosphorus is added to soils and sediments, the effect of reduction is to increase phosphorus sorption as compared to the oxidized state [65, 73]. Khalid et al. found a significant correlation between phosphorus sorbed under reduced conditions and the iron extracted by oxalate, also under reduced conditions. They postulated that poorly crystallized and amorphous oxides and hydroxides of iron play a primary role in phosphorus retention in flooded soils and sediments [73]. Bortelson and Lee concluded from studies of lake sediments that iron, manganese, and phosphorus are closely related in their deposition patterns and that iron content appeared to be the dominant factor in phosphorus retention [74]. Williams et al. [75] and Shukla et al. [76] found that noncalcareous sediments sorbed more phosphorus than calcareous sediments. Shukla et al. reported that the oxalate treatment of lake sediments to remove iron and aluminum almost completely eliminated the ability of sediments to retain phosphorus [76]. The amounts of iron removed were much greater than the amounts of aluminum removed by oxalate. They suggested that a gel complex of hydrated iron containing small amounts of aluminum oxide, silicon hydroxide, and organic matter was the major phosphorus-sorbing component in sediments under reduced conditions. Norvell found that sediments, maintained under reducing conditions, sorbed phosphorus at temperatures of 39 to 41°F (4 to 5°C) and that calcium, iron, and manganese were lost from both exchangeable and soluble forms during phosphorus sorption [77].

The retention of phosphorus by sediments in aqueous suspension has been demonstrated adequately. Thus, the problem of getting phosphorus retention by soils and sediments is no greater than in aerated soils. But the problem of getting contact between the sediments and soils and the wastewater represents a serious limitation. Pomeroy et al. found evidence of significant exchange of phosphorus between sediments and water when the sediments were suspended in the water, but when they were separated the exchange was trivial [78]. Where the sediments are not suspended, only a thin layer at the boundary between the water and the sediments is active in phosphorus retention. When wastewaters are added to wetlands, the sediments can play a significant part in removal of phosphorus from the water only if the water moves in and out of the sediments, or if wind or wave action keeps the sediments suspended. Running wastewater slowly over flooded soils in which plants are growing might be expected to remove phosphorus in a similar manner, and to about the same extent as found in overland flow systems, but in both cases the capacity of soil and sediments to reduce phosphorus concentrations is not fully used.

Spangler et al., after a 4 year study of natural and artificial marshes, concluded that these had potential for wastewater treatment [79]. In relation to phosphorus in a natural marsh, they found that (1) the marsh removed phosphorus during the summer and released it during other seasons, thus acting as a buffer; (2) harvesting of marsh vegetation was not a potential for removing a large portion of the phosphorus input; and (3) passage of wastewater through 6 232 ft (1 900 m) of the marsh reduced the orthophosphate and total phosphorus by 13% or less. Some of this reduction was probably a result of dilution with other water. A mass balance, using estimated water flows and concentrations, showed the same order of magnitude of phosphorus leaving the marsh as entering it. In other words, the marsh acted as a buffer for phosphorus concentration but was not effective in reducing the output.

However, Spangler et al. found that, in contrast to the natural marsh, artificial marshes removed 84% of the phosphorus input into greenhouse installations and 64% of the input into marshes constructed in the field [79]. They predicted that the removal in the field would be 80% under optimum conditions. Recommendations were for a system in which water would flow through, rather than over, the soil in the artificial marsh, which would be highly significant in removal of phosphorus as demonstrated by the work of Pomeroy et al. [78].

B.4 Models

B.4.1 Background

In a model that would be adequate for predicting the life of a wastewater treatment system, based on phosphorus retention in soils and sediments, many factors should be considered, including:

1. The rate of application of phosphorus

2. The amounts of calcium, iron, and aluminum in the wastewater and the influence of these constituents on the sorption of phosphorus in the soil

3. The removal of phosphorus by plant roots if the model deals with time intervals of days or weeks, or annual removal of phosphorus in harvested crops if the time intervals are years or decades

4. The travel distance and transit time of water flow

5. The transit time for water to move through the system relative to the kinetics of phosphorus sorption in soils and sediments

6. The rate of phosphorus application to the land relative to the kinetics of phosphorus reactions with soils (rapid infiltration systems might move phosphorus through before the slow reactions have an effect on phosphorus concentration in the flowing water)

7. Capacities and kinetics of sorption of phosphorus in soils and sediments from land surface to the point of discharge into ground or surface waters

Such a model would obviously be a three-dimensional model that would require information on water flow and phosphorus reactions that is usually not available and not easily obtained. Most water flow and proposed models for phosphorus retention deal only with flow in one direction, although some work, such as that reported by Jury [80, 81] deals with two-dimensional water flow. Thus, the discussion here will deal with flow downward through soils and to a depth that can be sampled and studied at reasonable cost. This depth is perhaps 6 to 10 ft (1.8 to 3 m) in most cases but might be much deeper in cases of deep alluvial materials.

All models are based on a materials balance, i.e., the phosphorus that goes into a volume of soil must be sorbed into the solid phase, must be removed by plants, or must move through the soil volume in percolating water. This means that all models have a water flow component and a phosphorus reaction component, and, of course, in systems involving crops, plant removal is a third component for both water and phosphorus. Models can consist of simple bookkeeping for water and phosphorus balances or of mathematical equations of various degrees of sophistication.

B.4.2 Limitations of Models

Although progress has been made during the past few years in the development of models of phosphorus movement in soils, a number of problems need to be solved before mathematical models or any other predictive models can be used with any degree of accuracy [10, 39, 40, 46, 82-84]. Large spatial and temporal variability in the hydraulic conductivity of soils in the field is tremendous and brings up the questions of how many and what kinds of samples or measurements are needed to characterize the water flow over an area for a given time. After the data are obtained, there are some problems of averaging and interpreting such large variations [85, 86].

There have been no studies of the numbers of samples needed to characterize the phosphorus sorption properties of a field to a given depth. Most sampling studies have dealt with problems of estimating the level of nutrients in the plow layer of soils. Recent studies of soil sampling for estimating the concentrations of soluble salts and nitrate in the unsaturated zone (to depths of 15 to 20 ft or 4.5 to 6 m) suggest that large numbers of samples are required and that adequate sampling of a field cannot be planned until some knowledge of the variability is obtained [87, 88]. Similar information on phosphorus sorption is needed before models can be accurately applied to fields, even if other limitations to the models are removed.

The composition of the wastewater (i.e., concentration of iron, aluminum, and calcium) will have an influence on the phosphorus reactions in the soil and the reactions of wastewaters that acidify or alkalinize the soil; for example, the influence of bicarbonate on neutralizing soil acidity will have effects on phosphorus reactions. Until these effects become inputs into a reliable model, the proper procedure would be to test each possible soil with the wastewater, or a reasonable simulation of the wastewater, being considered.

Perhaps the most serious limitation to all models is that the reaction of phosphorus with the soil cannot be predicted from measurements of simple soil properties that can be mapped in the field or measured quickly in the laboratory [46, 89]. Methods of characterizing soil that might correlate with phosphorus retention are likely to be more time-consuming than direct measurements of phosphorus sorption.

To allow near maximum application rates, rapid infiltration systems will require coarse gravelly or sandy soils having generally low sorptive capacities. These may soon be saturated, and the retention by the soil system will depend mostly on the iron, aluminum, and calcium in the wastewater and not on the original soil material. Exceptions to this general statement might include the use of calcareous sands and gravels for rapid infiltration systems. Plant removal is usually too small to be significant in rapid infiltration systems. Thus, the need is for a model that considers the constituents in the wastewater and how these will react during flow through the soil and sediments as a function of distance of flow and rate of flow.

B.4.3 Models of Kinetics of Phosphorus Reactions

The mathematical model for one-dimensional phosphorus movement in soils has been expressed in a number of ways. Enfield et al. [46] expressed it as

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial X^2} - V'\frac{\partial C}{\partial X} - \frac{\rho}{\theta}\frac{\partial S}{\partial t} \qquad \text{(B-3)}$$

where C = concentration of phosphorus in solution, mg/L
D = dispersion coefficient at velocity V', cm/h
t = time, h
V' = average pore-water velocity, cm/h
X = distance from beginning of flow path, cm
ρ = bulk density of soil, g/cm^3
θ = volumetric water content in the soil
S = sorbed phosphorus in solid phase, $\mu g/g$

The first two expressions in the equation deal with water flow, and the third deals with the retention of phosphorus by the soil (i.e., the kinetics of phosphorus reactions). Before this equation becomes a useful model, the kinetics of phosphorus reactions must be known.

The kinetics of phosphorus reactions in soils has been studied by a number of researchers [10, 33-36, 40, 47, 82, 90, 91]. Perhaps the most definitive study was that of Enfield et al. who measured the reactions of phosphorus in 25 soils for a period of 2 to 18 weeks, depending on the soil, and then used the data to test five kinetic models [46]. All kinetic models agreed adequately with the experimental models. Correlation coefficients between predicted values and experimental values averaged 0.81 to 0.88, but these were averages of values for individual soils. That is, coefficients for each kinetic model were calculated for and unique to individual soils, so that to use the models, the phosphorus reactions must be measured to supply the coefficients for the model for any individal soil material.

However, Enfield and Shew [40], using two of the models tested by Enfield et al. [45], found good agreement between the predicted movement of phosphorus and values experimentally determined in small laboratory columns which were fed a solution containing 10 mg/L of phosphorus. The first model was

$$\frac{\partial S}{\partial t} = \alpha \, (KC - S) \qquad \text{(B-4)}$$

where C = concentration of phosphorus, mg/L
S = concentration of sorbed phosphorus, μg P/g of soil
t = time, h
and, α, K = constants that depend on the soil

The second model was

$$\frac{\partial S}{\partial t} = a \, C^b S^d \qquad \text{(B-5)}$$

where the symbols have the same meaning as before and a, b, and d are constants that depend on the soil. Solutions to these equations were provided, and constants were calculated for two soils. Combinations of these with water flow data predicted phosphorus breakthrough curves for the two soils studied over a period of several days. Breakthrough curves indicated that the boundary between saturated and unsaturated soil (enriched versus nonenriched) was diffuse as illustrated in Figure B-1, so that the concentration of phosphorus in the effluent increased very slowly as the effluent volume increased.

Novak et al. developed a theoretical model for movement of phosphorus in soils in which the phosphorus sorption factor of Equation B-3 was calculated from existing adsorption-desorption models developed for chromatography and ion-exchange processes [82]. This model predicts an abrupt boundary for breakthrough of soluble phosphorus into any given layer of soil.

Harter and Foster developed an empirical model which describes the movement of phosphorus in soils [83]. In this approach, a soil sample is repeatedly treated with a solution of known phosphorus concentration, and the sorbed phosphorus is determined. The relationship between phosphorus sorbed and the volume of solution that has contacted the soil is then expressed as a polynomial adsorption equation

$$Y = A + BX + CX^2 + DX^3 \qquad \text{(B-6)}$$

where Y is the phosphorus adsorbed, X is the amount of phosphorus added, and A, B, C, D are constants that depend on the soil. From this relationship, the phosphorus breakthrough curves, or the phosphorus leaching front, can be plotted against depth or volume of wastewater added. The model is simple and might be adequate for most purposes, but there are no data available showing the effectiveness of the approach in predicting field data.

Shah et al. developed a materials balance mathematical model which agreed well with field data obtained from the barriered landscape water renovation system used to treat liquid swine manure [84]. The kinetic equation in this model was based on the Langmuir adsorption equation.

B.4.4 Empirical Model for a Slow Rate System

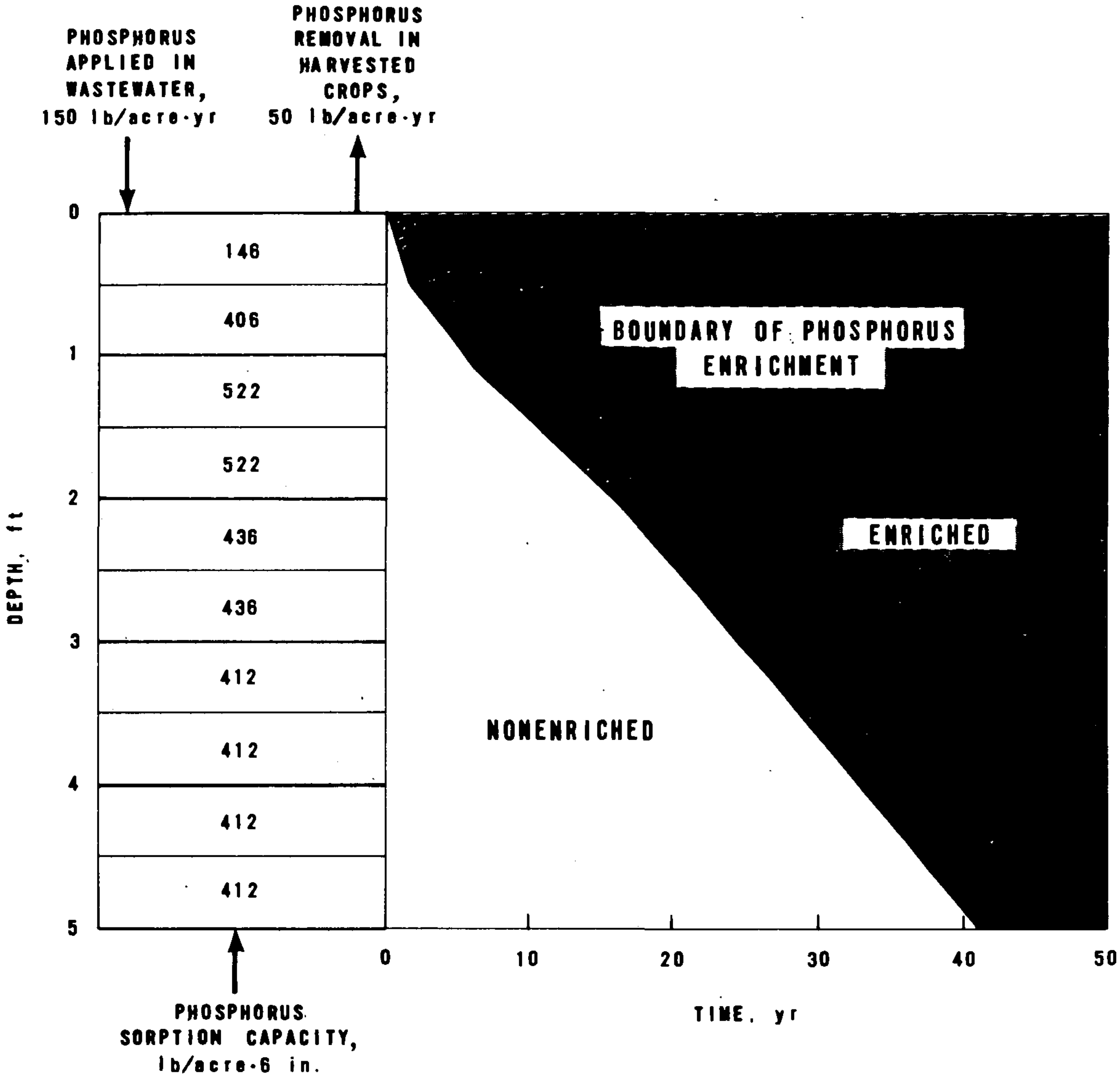
There are no models that adequately describe all factors in water and phosphorus movement in field soils receiving municipal wastewaters. Also, there is not sufficient knowledge of phosphorus reaction kinetics to predict the sorption of phosphorus in a field over periods of decades. Thus, the model presented here as Figure B-4 provides only an empirical assessment of relative phosphorus retentions by soil profiles.

In this simple model, the phosphorus added minus the phosphorus removed in harvested crops (Figure B-1) is assumed to react progressively with successive depth increments in the soil. The first depth increment becomes "saturated" before phosphorus moves to the next depth increment, and the boundary between the phosphorus-enriched soil and the non-enriched soil is assumed to be rather abrupt, as in the theoretical model of Novak et al. [82]. The term "saturated" is defined for the purposes of this model as the soil in which enrichment with phosphorus has been sufficient that movement with percolating water is significantly above background for the original soil material. Also, in this model, (1) water movement is considered to be so unimportant relative to phosphorus reactions that it can be disregarded, (2) there is sufficient time for slow phosphorus reactions to have a large impact, and (3) the phosphorus sorption capacities for the depth increments include the slow reactions.

The sorption capacities for the soil horizons used in Figure B-4 were taken from a study by Enfield and Bledsoe [10] in which 10 grams of soil were treated with 100 mL of solution containing a phosphorus concentration of 10 mg/L. The reaction time in this study was only

FIGURE B-4

ILLUSTRATION OF A SIMPLE PHOSPHORUS BALANCE-
PHOSPHORUS REACTION MODEL FOR A SLOW RATE SYSTEM



NOTE: THE PHOSPHORUS SORPTION CAPACITIES USED WERE DOUBLE THE SORPTION MEASURED IN A 125 DAY REACTION PERIOD IN THE LABORATORY.

1 lb/acre·yr = 1.12 kg/ha·yr

125 days; consequently, the sorption capacities in the Enfield and Bledsoe study were doubled to adjust for slow reactions that continue for indefinite periods in most soils.

This model, expressed mathematically, is

$$T = \frac{S_p}{I_p - H_p} \tag{B-7}$$

where T = time for the phosphorus front to reach a given depth in the soil, yr

S_p = the sorption capacity of the volume of soil above that depth, lb/acre (kg/ha)

I_p = the input phosphorus, lb/acre·yr (kg/ha·yr)

H_p = the phosphorus removed in harvested crops, lb/acre·yr (kg/ha·yr)

The uncertainties in the model are in the measurements of the phosphorus sorptive capacity and the assumed abrupt boundary between enriched and nonenriched soil. There is reason to believe that the phosphorus retention characteristics of a soil cannot be adequately characterized in the laboratory in a fixed period of time. Also, there is ample evidence from fields that have received large amounts of phosphorus as fertilizers or as wastes that the soluble phosphorus gradually decreases as a function of depth with a diffuse rather than an abrupt boundary between the highly enriched and the nonenriched soil horizons [5, 19, 49, 51, 52, 93]. But, perhaps even considering these uncertainties, the model can be useful as a preliminary estimate of the phosphorus retention characteristics of various soils and sediments. This type of model is implied in the rate classes proposed by Schneider and Erickson for phosphorus sorption measurements in soils as a limitation for the use of the soil for treating municipal wastewaters [94]. Their limitation classes ranged from very high to very low, respectively, as the phosphorus sorption increased from less than 1 000 to more than 2 000 lb/acre (1 120 to 2 240 kg/ha) in 3 ft (0.9 m) of soil.

If a model such as presented in Figure B-4 is used to classify the desirability of various potential areas for a given wastewater, monitoring of the phosphorus movement in the site selected can be used to revise estimates of the longevity of the site as it is being used.

B.4.5 Model for Rapid Infiltration Systems

At the present time, there is no accepted model that can predict the movement of phosphorus through soil profiles. But, one promising approach which considers the rate of reaction of phosphorus with soils is that of Enfield and Bledsoe [10]; Enfield and Shew [40], and Enfield [39, 95]. In this approach, solutions containing phosphorus at various concentrations were reacted with soils for up to 125 days, using batch techniques. The sorption data thus obtained were compared with phosphorus breakthrough curves using small 1.9 in. (5 cm) long soil columns.

As might be expected from discussions of reaction kinetics of phosphorus in soils, the sorption of phosphorus obtained from a 10 hour reaction period seriously underestimated the amount of phosphorus sorbed by the columns in 55 days. However, sorption curves had been obtained at time intervals from 1 to 3 000 hours, approximating a geometric progression, so that sorption surfaces as a function of (1) equilibrium solution concentration, (2) amount of phosphorus sorbed by the soil, and (3) time, were produced for each soil. When these sorption surfaces were used to adjust the ratio of phosphorus sorbed to equilibrium solution concentration for time corresponding to time intervals in the breakthrough curves, there was a marked increase in the agreement between the batch and column techniques.

The sorption surfaces were used to calculate the sink term $\partial S/\partial t$ in the equation

$$\frac{\partial C}{\partial t} = -\overline{V}\,\frac{\partial C}{\partial X} - \frac{e}{\theta}\,\frac{\partial S}{\partial t} \qquad \text{(B-8)}$$

where C = solution phase concentration of phosphorus, mg/L
$\overline{V}$ = average pore-water velocity, in./h (cm/h)
X = distance from the beginning of the flow path, in. (cm)
e = bulk density of the soil, g/cm^3
θ = volumetric water content of the soil
S = solid phase concentration of sorbed phosphorus, µg/g
t = time, h

and the sink term was calculated from

$$\frac{\partial S}{\partial t} = a\ C^b S^d \qquad \text{(B-9)}$$

where a, b, and d are constants and C, S and t are as defined for Equation B-8. Using these equations, the breakthrough curves agreed well with predicted curves in three of four soils.

Enfield recognized that this approach was not satisfactory for all soils and that it did not predict the effects of rest periods and desorption of phosphorus during rains and the resultant leaching with rainwater [95]. Nevertheless, this approach gives an adequate first approximation to the transport of phosphorus through soils and can provide a basis for design of wastewater treatment systems. Enfield recommended that an average phosphorus concentration or application rate adjusted for rest periods, plant removals, and rainwater, be used [95].

There are some serious questions about this approach that must be added as words of caution. The spatial variability in phosphorus reactions and in water flow can be much smaller in the laboratory columns than in the field. Temperature effects on the kinetics of phosphorus reactions are not built into the laboratory studies but will be encountered in the field. In the field, a given volume of soil will react with increasing concentrations of phosphorus as a function of time of treatment with wastewater; whereas, in the sorption measurements presented by Enfield and Bledsoe [10], the soil reacted with decreasing concentrations of phosphorus. Solutions of phosphorus at 10, 40, and 100 mg/L were added to soils and the decrease in concentration was measured as a function of time which is an approach to equilibrium from the opposite direction as in the field. The sink term, $\partial S/\partial t$, of Equation B-8 might be substantially different under these two conditions in some soils.

Another aspect concerning this approach is more pragmatic. Considering the status of such models, it may be no more expensive to set up columns of soils in the laboratory and treat them with the wastewater for a period of 4 to 6 months and directly measure the movement of phosphorus.

B.5 References

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APPENDIX C

HYDRAULIC CAPACITY

C.1 Introduction

The hydraulic capacity of the soil to accept and transmit water is crucial to the design of rapid infiltration systems and important in the design of most slow rate systems. The important hydraulic parameters are infiltration, vertical permeability (percolation), and horizontal permeability. In this appendix, the basic hydraulic properties are defined and techniques for measurement and estimation of the more important parameters are presented. Both vertical and horizontal flow of groundwater are discussed, and an analysis of groundwater mounding is presented. The relationship between predicted hydraulic capacity and actual operating rates is also discussed.

C.2 Hydraulic Properties

For purposes of this manual, hydraulic properties of soil are considered to be those properties whose measurement involves the flow or retention of water within the soil profile. These properties include soil-water characteristic curve, percent moisture at saturation, permeability, infiltration rate, specific yield, specific retention, and transmissivity. In addition, the terms of field capacity, permanent wilting point, and drainability are commonly used in irrigation practice. However, these terms describe qualitative relationships--not unique, measurable properties. The concepts of field capacity and permanent wilting point are discussed in conjunction with soil-water characteristic curves.

Soil permeability and infiltration rate are especially important to system design. They should be determined by field testing; however, they may be estimated from other physical properties (mentioned in Section F.3.3.1). An in-depth discussion of the more common methods of estimating or measuring both soil and aquifer properties is presented in this appendix. Field testing procedures for determining soil infiltration rates and permeability are outlined in Section C.3, along with methods of analyzing and interpreting test results. Methods of measuring or estimating properties of groundwater aquifers are outlined in Section C.4.

C.2.1 Soil-Water Characteristic Curve

Water in the soil below the saturation level is held in the soil against the force of gravity primarily by forces that result from the surface tension of water, the cohesion of water molecules, the adhesion of water molecules to soil surfaces, and other electrical attractive forces at the molecular level. The energy required to remove water from unsaturated soil when expressed on a per unit mass of water basis is termed the soil-water pressure potential or matric potential. Soil-water pressure potential is expressed as J/kg or erg/g. The energy is sometimes expressed on a unit volume basis in which case it is termed soil-water pressure. The resulting units (erg/cm^3) convert to those of pressure, $dyne/cm^2$, or more commonly, bar (10^6 $dyne/cm^2$). The most common method of expressing the energy is on a unit weight basis in which case it is termed soil-water pressure head or simply head. The resulting units, erg/dyne, convert to centimetres. Soil tension and suction are terms that also have been used to describe the energy of soil-water retention, but these terms make no distinction among units. They are also considered positive quantities, while the above terms are negative quantities.

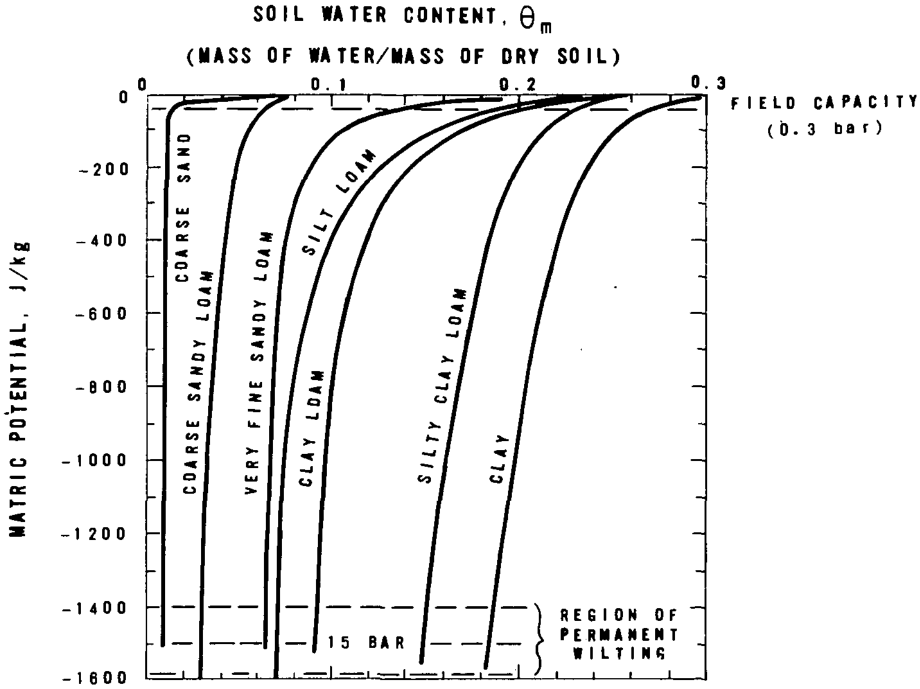
The force by which water is held in the soil is approximately inversely proportional to the pore diameter. Thus, the larger the pore the less energy is required to remove water. As soil dries or drains, water is removed from the larger pores first. The water remains in the smaller pores because it is held more tightly. Thus, as soil-water content decreases, soil matric potential increases. The graphical relationship between soil-water content and matric potential is the soil-water characteristic curve. Examples of such curves for several different types of soil are shown in Figure C-1. It should be mentioned that different curves will be obtained depending on whether the soil-water content is changed by drying or by wetting. This hysteresis phenomenon is due primarily to soil pore configurations. In most cases we are interested in the soil-water characteristic curve resulting from drying or drainage.

It is apparent from Figure C-1 and the previous discussion that the shape of the soil-water characteristic curve is strongly dependent on soil texture and soil structure. For example, sandy soils have mostly large pores of nearly equal size. Consequently, nearly all water is removed from sands at a very small matric potential. On the other hand, medium-textured, loamy soils have a greater porosity than sands and a wide pore size distribution. Thus, more water is held at saturation in soils than in sands and it is removed much more gradually as matric potential becomes larger.

Some important aspects of soil-water plant relations may be explained by the shape of the soil-water characteristic curve. In irrigation practice, it has been common to describe the maximum amount of water in soil that is available for plant uptake as the difference in water content at field capacity (the upper limit) and that at permanent wilting point (the lower limit). A soil is said to be at field capacity when the rate of water removal from the soil, due to drainage following an irrigation or heavy rain, begins to be reduced. As such, field capacity is not a unique value, but represents a general region of water percentages as illustrated in Figure C-2. Using the soil-water characteristic curve, field capacity then may be expressed as a range of soil-water pressure potentials. The range of potentials in the region of field capacity varies with soil texture. For sands, the range of field capacity is about 10 to 15 J/kg or 0.1 to 0.15 bar. For medium- to fine-textured soils, the range is about 0.3 to 0.5 bar. A value of 0.3 bar is commonly used as a rough approximation in these soils.

FIGURE C-1

SOIL-WATER CHARACTERISTIC CURVES FOR SEVERAL SOILS [1]

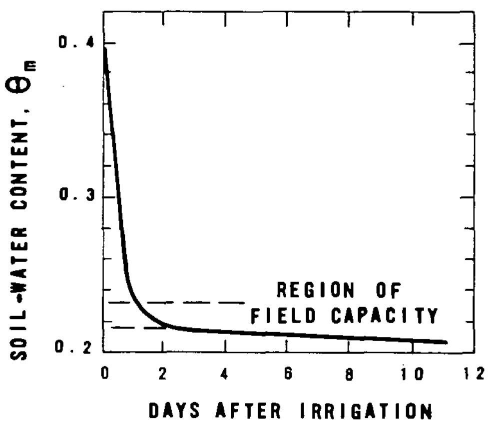


The time required for a thoroughly wetted soil to drain to the field capacity region is also dependent on texture and structure. In the absence of significant evaporation or transpiration, field capacity in pure sands may be reached in a few hours; for coarse soils about 2 to 3 days; for medium- to fine-textured soils, a week or more; and for poorly structured clays, much longer.

There are a few misconceptions associated with the concept of field capacity that should be pointed out. The first is that field capacity is a unique property of a soil. It is apparent from the previous discussion that field capacity expresses only a crude qualitative relation. The second misconception is that no drainage occurs in soils at or below field capacity. In fact, drainage does not cease at field capacity but continues at a reduced rate for a long time, as illustrated in Figure C-2. The third misconception is that field capacity represents the upper limit of water that is available to plants and any water applied in excess of field capacity will be lost from the soil profile as deep percolation. However, water in excess of field capacity is available to plants while it remains in contact with plant roots.

FIGURE C-2

FIELD CAPACITY RELATIONSHIP



The lower limit of water availability, the permanent wilting point, like field capacity, is not a unique value but a range of water percentages over which the rate of water taken up by the plant is not sufficient to prevent wilting. The ability of the plant to take up water is directly related to the matric potential of the soil-water rather than the actual water content. It has been found that most plants exhibit permanent wilting when the soil-water matric potential is in the range of 1 500 J/kg (15 bars).

If it is assumed that the so called available reservoir of water is the water content between field capacity and permanent wilting point, some general observations can be made regarding the effect of texture on irrigation scheduling. From the shapes of the various soil-water characteristic curves shown in Figure C-1, it is apparent that sandy soils have a relatively small difference in water content between the regions of field capacity and permanent wilting. Medium- to fine-textured soils, on the other hand, exhibit a rather large difference in water content between field capacity and permanent wilting point.

Another generalization that can be made is that coarse soils approach the permanent wilting point very rapidly with small changes in water content. Thus, plants grown in such soils would be expected to exhibit wilting symptoms quite suddenly. Medium- to fine-textured soils approach permanently wilting point more gradually and plants grown on these soils will likely show very gradual signs of wilting.

Soil-water characteristic curves generally must be determined in the laboratory using techniques described in Taylor and Ashcroft [1]. Published soil moisture versus matric potential data are available for selected typical soils. The USDA Agricultural Research Service Publication 41-144 [2] provides such data for 200 typical soils in 23 states. In addition, bulk density, total porosity, and saturated vertical permeability data are presented in this compendium.

C.2.2 Percent Moisture at Saturation

The percent moisture at saturation or saturation percentage is defined as the number of grams of water required to saturate 100 grams of air-dry soil. It is a convenient parameter to measure since a saturated paste is normally prepared for other analyses. Saturation is reached when the soil surface glistens but no free moisture is present. A saturated paste will not flow from a container unless shaken. Due to the subjective nature of the test, large variations in test results from different sources are common. Thus, saturation percentage data should be used with caution.

Saturation percentage is a useful parameter because it provides a quick, rough estimate of the available water-holding capacity of the soil. The field capacity is approximately one-half the saturation percentage, and about one-half the field capacity of the soil can be considered available to plants. Of course, a better estimate of available water-holding capacity can be obtained from soil-water characteristic curves as previously described.

The value of saturation percentage is related to soil texture. Typical ranges for various soil textures are presented in Table C-1.

C.2.3 Permeability

Soil permeability is a term that has been used rather loosely to describe the ease with which liquids and gases pass through soil. In this manual, the term permeability will be synonymous with hydraulic conductivity. Hydraulic conductivity is the more descriptive and the preferred term, but, for the sake of consistency with much of the literature, permeability will be used in this manual. These terms are most easily defined if a few basic concepts of water flow in soils are introduced first.

TABLE C-1

RELATION OF SATURATION PERCENTAGE
TO SOIL TEXTURE

Soil texture	Saturation % range
Sand or loamy sand	Below 20
Sandy loam	20-35
Loam or silt loam	35-50
Clay loam	50-65
Clay	65-135
Peat or muck	Above 135

In general, water moves through soils or porous media in accordance with Darcy's law:

$$q = K\ dH/dl \qquad \text{(C-1)}$$

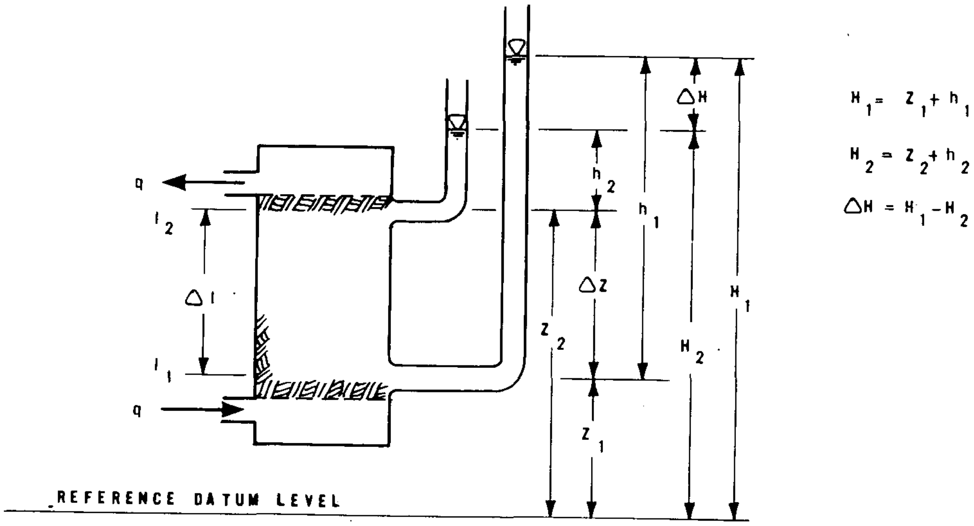
where q = flux (flow) of water per unit cross sectional areas, in/h (cm/h)
K = permeability (or hydraulic conductivity), in./h (cm/h)
dH/dl = total head (hydraulic) gradient, ft/ft (m/m)

The total head (H) is the sum of the soil-water pressure head (h), and the head due to gravity (Z), or $H = h + Z$. The hydraulic gradient is the change in total head (dH) over the path length (dl). These relationships are illustrated schematically for saturated and unsaturated conditions in Figure C-3.

The permeability is defined as the proportionality constant, K. Permeability (K) is not a true constant but a rapidly changing function of water content. Even under conditions of constant water content, such as saturation, K may vary over time due to increased swelling of clay particles, change in pore size distribution due to classification of particles, and change in the chemical nature of the soil-water. However, for most purposes saturated permeability (K_s) values can be considered constant for a given soil. In general, the K value for flow in the vertical direction will not be equal to K in the horizontal direction. This condition is known as anisotropic.

FIGURE C-3

SCHEMATIC SHOWING RELATIONSHIP OF TOTAL HEAD (H), PRESSURE HEAD (h), AND GRAVITATION HEAD (Z) FOR SATURATION FLOW



SATURATED FLOW

The permeability of soils at saturation is an important parameter because it is used in Darcy's equation to estimate groundwater flow patterns (see Section C.4) and is useful in estimating soil infiltration rates. Permeability can be estimated from other physical properties but much experience is required and results are not sufficiently accurate for design purposes.

As suggested by the inclusion of textural classification in Tables 3-7 and 3-9, soil permeability is determined to a large extent by soil texture with coarse materials generally having higher conductivities. However, in some cases the soil structure may be equally as important. A well structured clay with good stability can have a greater permeability than a much coarser soil.

Permeability of soils is also affected by the ionic nature of the soil water. A simplified explanation of this effect is given. Clay particles in the soil are negatively charged due to substitution of lower valence atoms for higher valence atoms (e.g., Al^{3+} for Si^{4+}) in the crystal structure. Because of the charge, the clay particles repel each other and remain dispersed in the soil unless the charge is neutralized by positively charged cations in the soil-water. Thus, waters high in salts will contain sufficient cations to neutralize the clay particles and allow them to come close enough together so that short-range molecular attractive forces will unite, or flocculate the particles. Flocculation of particles will result in larger soil pores and increased permeability. Thus, waters low in salts may result in low permeability problems.

The type of ion in the soil-water also affects permeability. Water with larger sodium percentages can cause reduced permeability. This occurs because the sodium ion in its hydrated state is much larger than other ions, calcium and magnesium in particular. Thus, the layer of sodium ions necessary to neutralize clay particles is thicker and the clay particles are restricted from coming together and remain dispersed. As a result, the permeability of the soil is low. If sufficient salt is present in the soil-water, the layer of sodium ions will be suppressed to the point that clay particles will flocculate and permeability will be adequate. However, the salt concentration may be so high that the growth of plants may be restricted (see Section 5.6.2.3).

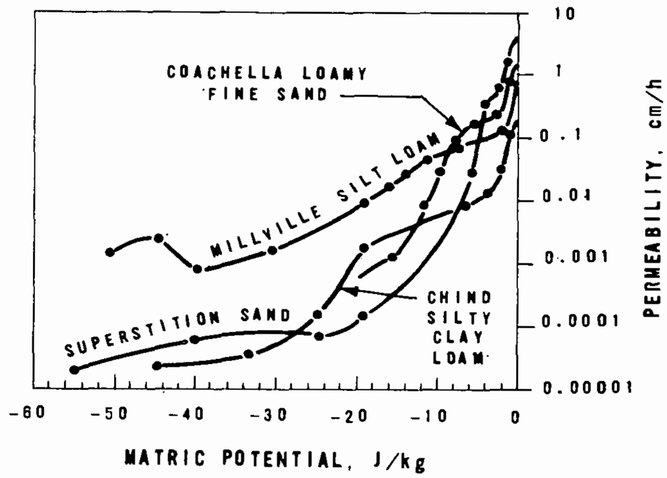
The type of vegetation will also affect the permeability of soil within the root zone. The effects of ions and vegetation on permeability are not of concern for groundwater aquifers because they are below the root zones and contain little, if any, clay.

As previously discussed, the permeability of soil varies dramatically as water content is reduced below saturation. Since matric potential also varies as a function of water content in accordance with the soil-water characteristic curve, permeability may be described as a function of matric potential. The inverse relationship between permeability and matric potential is illustrated for soils of several different textures in Figure C-4. The significant relationship to note is that the permeability of sandy soils, although much higher at saturation (matric potential = 0) than loamy soils, decreases more rapidly as the matric potential becomes more negative. In most cases, the permeabilities of sandy soils eventually become lower than the medium soils. This relationship explains why a wetting front moves more slowly in sandy soils than medium or fine soils after irrigation has stopped and why there is little horizontal spreading of moisture in sandy soils after irrigation.

Estimating water movement under unsaturated conditions using Darcy's equation and unsaturated K values involves relatively complex mathematical techniques. A discussion of such techniques is not within the scope of this manual. The user is referred to Kirkham and Powers [3] or Klute [4] for further details on the subject of unsaturated flow.

FIGURE C-4

PERMEABILITY AS A FUNCTION OF THE
MATRIC POTENTIAL FOR SEVERAL SOILS [1]



C.2.4 Infiltration Rate

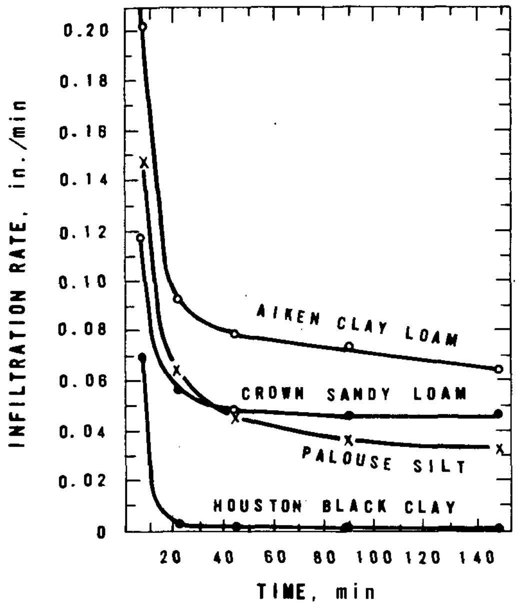
The infiltration rate of a soil is defined as the rate at which water enters the soil from the surface. When the soil profile is saturated, the infiltration rate is equal to the effective saturated permeability of the soil profile. This occurs because at saturation the matric potential (h) is zero at all depths and the total head gradient (d[h + z]/dz) is equal to unity. Thus, the flux q is equal to k according to Darcy's law (Equation C-1).

When the soil profile is relatively dry, the infiltration rate is higher because water is entering large pores and cracks. With time, these large pores fill and clay particles swell reducing the infiltration rate rather rapidly until a near steady-state value is approached. This change in infiltration rate with time is shown in Figure C-5 for several different soils. The effect of both texture and structure on infiltration rate is illustrated by the curves in Figure C-5. The Aiken clay loam has good structural stability and actually has a higher final infiltration rate than the sandy soil. The Houston black clay, however, has very poor structure and infiltration drops to near zero.

As with permeability, infiltration rates are affected by the ionic composition of the soil-water and the type of vegetation. Of course, any tillage of the soil surface will affect infiltration rates. Factors which have a tendency to reduce infiltration rates include clogging by organic solids in wastewater, classification of fine soil particles, clogging due to biological growths, and gases produced by soil microbes.

FIGURE C-5

INFILTRATION RATE AS A FUNCTION
OF TIME FOR SEVERAL SOILS [1]



The steady-state infiltration rate is extremely important and generally serves as the basis for selection of the design hydraulic loading rate for slow rate and rapid infiltration systems. A more detailed discussion of soil infiltration, including field measurement techniques, is presented in Section C.3.

C.2.5 Drainability

Drainability is a qualitative term that is commonly used in soil surveys and elsewhere to describe the relative rapidity and extent of the removal of water from the root zone by flowing through the soil to subsoils or aquifers. A soil is considered well-drained if, upon saturation, water is removed readily, but not rapidly. A poorly-drained soil is one in which the root zone remains waterlogged for long periods of time following saturation and insufficient oxygen supply to roots becomes growth limiting to most plants. An excessively-drained soil is one from which the water is removed so completely that most crop plants suffer from lack of water.

In general, loamy soils are well-drained and provide the best balance between drainage and water holding capacity for crop production. Poorly structured, fine, or moderately fine textured soils normally are poorly-drained and are best suited to overland flow systems. Sandy soils are often excessively-drained and best suited to rapid infiltration systems.

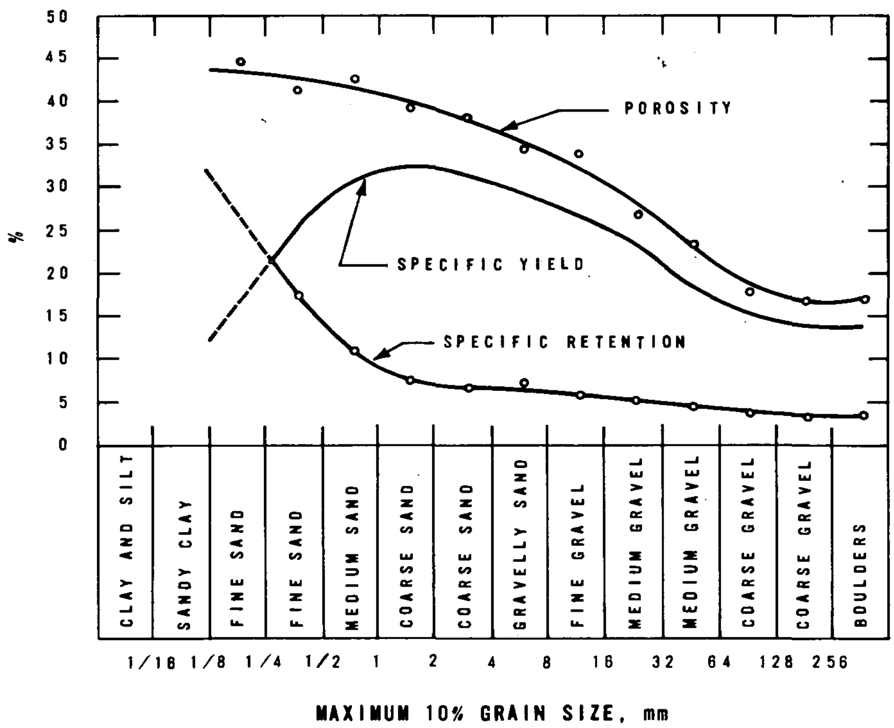
It should be recognized that the drainability of a soil profile will be determined by the most restrictive layer in the profile. Thus, a shallow sandy surface layer underlain by a poorly-drained clay layer will be poorly-drained.

C.2.6 Specific Yield and Specific Retention

Specific yield and specific retention are related properties that are a measure of the amount of groundwater an aquifer will yield upon pumping. Specific yield is the amount of water that will drain by gravity from a saturated aquifer divided by the bulk volume of the aquifer. This value is typically 10 to 20% for unconfined aquifers [5]. Specific retention is equal to the porosity (subsurface void space) minus the specific yield under saturated conditions. The relationship among specific yield, specific retention, and porosity is shown in Figure C-6.

FIGURE C-6

POROSITY, SPECIFIC RETENTION, AND
SPECIFIC YIELD VARIATIONS WITH GRAINS SIZE,
SOUTH COASTAL BASIN, CALIFORNIA [5]



C.2.7 Transmissivity

Transmissivity is a parameter that is sometimes measured in the field as part of a method to determine horizontal permeability of aquifers [6]. Transmissivity T is the rate at which water is transmitted through a unit width of the aquifer under a unit hydraulic gradient. It is equal to the permeability K multiplied by the aquifer thickness.

C.3 Soil Infiltration Rate and Permeability Measurements

Field measurements of soil infiltration rates and permeability are an essential part of the design of rapid infiltration systems and most slow rate systems. These hydraulic parameters serve as the basis for the designer's selection of an application rate that will be within the hydraulic capacity of the soil at the proposed site. In this section, the principal methods for measuring infiltration rates and vertical permeability are reviewed along with procedures for using the field test results to obtain infiltration equations that are useful in the design of irrigation systems. The relation between infiltration and vertical permeability is discussed. Measurement of soil moisture profiles is also addressed.

C.3.1 Infiltration

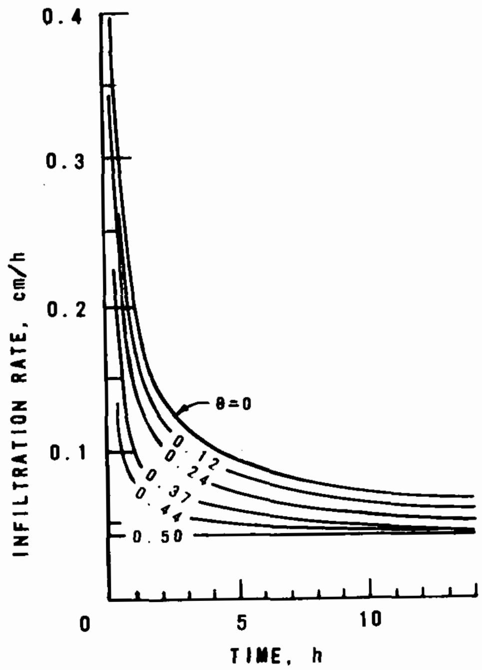
Infiltration refers to the entry of the water into the soil. Hydraulic or liquid loading is infiltration over a long term (a year, for example) and includes resting or drying periods. The factors that affect infiltration have been thoroughly discussed in the literature [7] and must be firmly kept in mind when planning and making field measurements. Otherwise, the measurements, which are relatively easy to make, may be meaningless for the intended purposes.

C.3.1.1 Interpretation and Use of Infiltration Data

As previously mentioned, (Section C.2.4), when water is applied to a soil that is below field capacity, the rate of infiltration generally decreases with time, approaching a nearly constant or steady state value after several minutes or hours of application. However, initial infiltration rates may vary considerably, depending on the initial soil moisture level. Dry soil has a higher initial rate than wet soil because there is more empty pore space for water to enter. The effect of soil moisture level on initial infiltration rates and the change in infiltration rate as a function of time is illustrated in Figure C-7. The short-term decrease in infiltration rate is primarily due to the change in soil structure and the filling of large pores as clay particles absorb water and swell.

FIGURE C-7

INFILTRATION RATE CURVES SHOWING THE INFLUENCE OF INITIAL WATER CONTENT, θ (FRACTION BY VOLUME), ON INFILTRATION RATE COMPUTED FOR YOLO LIGHT CLAY [1]



Long-term decreases in the steady state infiltration rates that take place over several months of application are also observed. These decreases are the result of several factors, including (1) the migration and concentration of fine soil particles in the soil profile, (2) the buildup of organic and biological solids in the soil pores, and (3) the blockage of soil pores by gases produced by soil microbes. The steady state infiltration rate may be improved or maintained by soil tilling and other management practices.

The short-term change in infiltration rate as a function of time is of interest in the design and operation of irrigation systems. A knowledge of how cumulative water intake varies with time is necessary to determine the time of application necessary to infiltrate the quantity of water required to irrigate a crop. The design application rate of sprinkler systems is selected on the basis of the infiltration rate expected at the end of the application period. The short-term change in infiltration rate can be closely approximated by the simple equation:

$$I = At^n \qquad \text{(C-2)}$$

where I = infiltration rate, in./min
A = a constant, representing the instantaneous intake rate at time = 1 (usually minutes)
n = an exponent which for most soils is negative with values between 0 and -1

Integration of the rate equation yields an equation for the cumulative intake Y at any time t. The equation has the following form:

$$Y = \frac{A}{n+1} t^{n+1} \qquad \text{(C-3)}$$

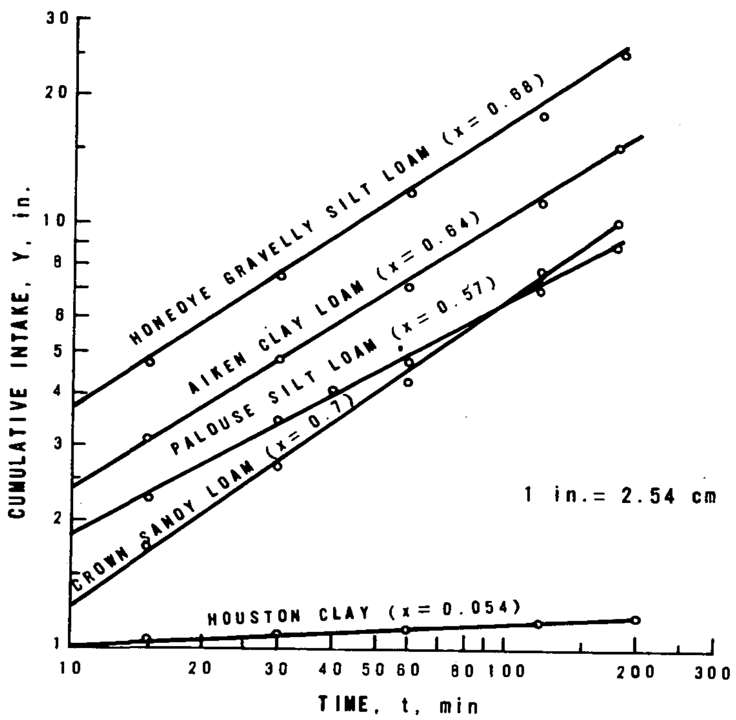
Data from infiltrometer studies can be plotted to yield cumulative intake curves from which the coefficients for Equations C-2 and C-3 may be obtained. Alternatively, the cumulative intake may be computed as a function of time using the Green-Ampt infiltration model. Knowledge of the vertical permeability profile and the initial soil moisture profile is needed to apply this technique. The K profile may be determined by the methods described in Section C.3.3. The soil moisture may be determined by using a neutron moisture probe or gravimetric sampling [1]. The calculation procedures are described by Bouwer for various K and moisture profiles [8, 9]. The advantage of the calculation method over infiltrometer measurements in generating Y versus t data is that the uncertainty associated with lateral seepage under infiltrometers is avoided, and the Y versus t relationship can be computed easily for any soil moisture profile. Data obtained from the Green-Ampt model can be plotted in the same manner as infiltrometer data to yield infiltration rate versus time curves.

The most direct method to determine coefficients for Equation C-3 is to plot the data points on log-log paper with time on the abscissa and cumulative intake on the ordinate, and to fit the best straight line through the points. An example of such a plot for several different soils is shown in Figure C-8 [1]. The intercept of the curve at $t = 1$ is equal to $A/n + 1$ (not shown in Figure C-8), and the slope of the line is equal to $n + 1$.

The most important application for the cumulative intake curves is in the design and evaluation of border irrigation systems. The curves may be compared with a set of intake family curves developed by SCS for border irrigation design, and the appropriate intake family can then be selected. Cumulative intake curves may also be developed for furrow irrigation system design and evaluation. However, infiltrometer data are not directly applicable to furrow irrigation because only part of the land is in contact with the water, and lateral seepage represents a large part of the total intake. Consequently, actual field trials using furrows are required to develop infiltration rate versus time curves. Infiltration rate curves may be obtained by applying the constants A and n to Equation C-2. Infiltration rate curves may be useful in selecting sprinkler application rates.

FIGURE C-8

CUMULATIVE INTAKE CURVES SHOWING THE INFILTRATION OF WATER INTO SOIL FROM SINGLE RING INFILTROMETERS [1] ($\alpha = n + 1$)

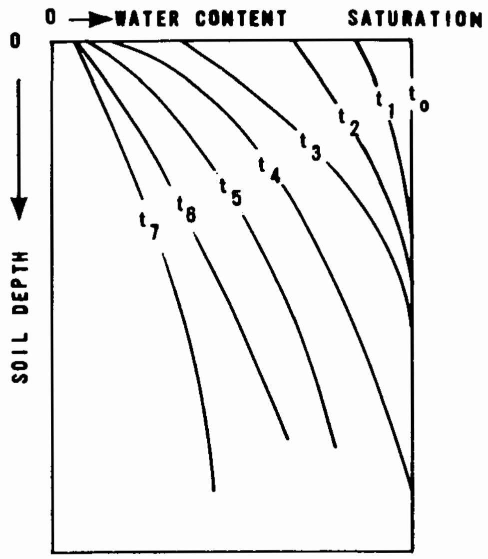


C.3.1.2 Soil Profile Drainage Studies

For slow rate systems that are operated at application rates considerably in excess of crop irrigation requirements, it is often desirable to know how rapidly the soil profile will drain and/or dry after application has stopped. This knowledge, together with knowledge of the limiting infiltration rate of the soil and the groundwater movement and buildup, allows the designer to make a reasonable estimate of the maximum volume of water that can be applied to a site and still produce adequate crops. A typical moisture profile and its change with time following an irrigation is illustrated in Figure C-9 for an initially saturated profile.

FIGURE C-9

TYPICAL PATTERN OF THE CHANGING MOISTURE PROFILE DURING DRYING AND DRAINAGE



Moisture profile changes may be determined in the field by measuring the soil-water tension at various times and at various depths in the profile with tensiometers. Soil tension data can then be converted to moisture content values by use of the soil-water characteristic curve of the soil at each measured depth. Soil-water characteristic curves are determined by laboratory methods. A discussion of these methods and the use of tensiometers is presented in Taylor and Ashcroft [1].

C.3.1.3 Estimates of Infiltration From Soil Properties

Estimation of infiltration and percolation rates without benefit of actual onsite testing is an undesirable practice, but the general relationships that have been established between hydraulic capacity and soil properties through experience are certainly reliable enough to permit preliminary screening of several available sites. Soil scientists generally agree that when a large number of widely different soils are considered, no single factor can serve as an index for determining the infiltration rate or permeability of an individual soil profile.

In a comprehensive study by O'Neal, it was concluded that structure is probably the most important single soil characteristic in evaluating hydraulic characteristics, but it was impossible to estimate these on the basis of structural factors alone [10]. The approach suggested in the basis of structural factors alone [10]. The approach suggested in the study resulted in only fair precision using four principal factors: (1) relative dimension (both horizontal and vertical) of structural aggregates, (2) amount and direction of overlap of aggregates (3) number of visible pores, and (4) texture. None of these factors, when considered singly, was a reliable indicator of permeability, and each one had to be considered with reference to all the others. The seven soil permeability classes used by the SCS are as follows:

Class	Soil permeability, in./h
Very slow	<0.06
Slow	0.06-0.2
Moderately slow	0.2-0.6
Moderate	0.6-2.0
Moderately rapid	2.0-6.0
Rapid	6.0-20.0
Very rapid	>20.0

1 in./h = 2.54 cm/h

Using these classes, the soils experts who participated in O'Neal's work were able to estimate the correct permeability class for 68% of the 271 horizons examined, and they were within one class ranking for an additional 24%. Note particularly, however, that these results were achieved by trained persons. Moveover, it must be remembered that even within a particular class there is room for an error of up to 400%.

The difficulty is that even if permeability could be accurately determined from a particular property (such as particle size distribution), it would still be influenced by factors which that particular measurement could not account for. These might include grain orientation, colloid migration or swelling, bulk density changes by compaction, and chemical or biological effects. Thus, it would seem reasonable for reviewing agencies to insist on at least some field measurements, of the type recommended in Chapter 4, for all land treatment systems where the intended application rates are well in excess of the known evapotranspiration rates.

C.3.1.4 Infiltration Measurement Techniques

The value that is required in land treatment is the long-term acceptance rate of the entire soil surface on the proposed site for the actual wastewater effluent to be applied. The value that can be measured is only a short-term equilibrium acceptance rate for a number of particular areas within the overall site. It is strongly recommended that hydraulic tests of any type be conducted with the actual wastewater whenever possible. Such practice will provide valuable information relative to possible soil-wastewater interactions which might create future operating problems. If suitable wastewater is not available at the site, the ionic composition of the water used should be adjusted to correspond to that of the wastewater. Even this simple step may provide useful data on the swelling of expansive clay minerals due to sodium exchange.

The theory of infiltration, in which great strides have been made in the past decade, has simply not yet found practical application in the land treatment of wastewater. Modeling the physics of unsaturated flow, while important, has not answered the present need for simple and economical assessment of soil hydraulic capacity. Measurements have been made by numerous different techniques without follow-up studies to relate operating results to the original measurements. Research is needed in this area to improve existing design techniques.

There are many potential techniques for measuring infiltration including basin flooding, sprinkler infiltrometers, cylinder infiltrometers, and lysimeters. The technique selected should reflect the actual method of application being considered. The area of land and the volume of wastewater used should be as large as practical. The two main categories of measurement techniques are those involving flooding (ponding over the soil surface) and rainfall simulators (sprinkling infiltrometer). The flooding type of infiltrometer supplies water to the soil without impact, whereas the sprinkler infiltrometer provides impact similar to that of natural rain. Flooding infiltrometers are easier to operate than spinkling infiltrometers, but they almost always give higher equilibrium infiltration rates. In some cases, the difference is very significant, as shown in Table C-2. Nevertheless, the flooding measurement techniques are generally preferred because of their simplicity.

TABLE C-2

COMPARISON OF INFILTRATION MEASUREMENT USING FLOODING AND SPRINKLING TECHNIQUES [11]

	Equilibrium infiltration rate, in./h	
Measurement technique	Overgrazed pasture	Pasture, grazed but having good cover
Double-cylinder infiltrometer (flooding)	1.11	2.35
Type F rainfall simulator (sprinkling)	0.14	1.13

in./h = 2.54 cm/h

Before discussing these four techniques, it should be pointed out that the standard U.S. Public Health Service (USPHS) percolation test used for establishing the size of septic tank drain fields [12] is not recommended except for very small subsurface disposal fields or beds. Comparative field studies have shown that the percolation rate from the test hole is always significantly higher than the infiltration rate as determined from the double-cylinder (also called double ring) infiltrometer test. The difference between the two techniques is of course related to the much higher percentage of lateral flow experienced with the standard percolation test. The final rates measured at four locations on a 30 acre (12 ha) site using the two techniques are compared in Table C-3. The lower coefficient of variation (defined as the standard deviation divided by the mean value, $C_v = \sigma/M$) for the double-cylinder technique is especially significant. A plausible interpretation is that the measurement technique involved is inherently more precise than the standard percolation test.

TABLE C-3

COMPARISON OF INFILTRATION MEASUREMENT USING STANDARD USPHS PERCOLATION TEST AND DOUBLE-CYLINDER INFILTROMETER[a]

Location	Equilibrium infiltration rate, in./h Standard USPHS percolation test	 Double-cylinder infiltrometer
1	48.0	9.0
2	84.0	10.8
3	60.0	14.4
4	138.0	12.0
Mean	82.5	11.6
Standard deviation	40.0	2.3
Coefficient of variation	0.48	0.20

a. Using sandy soil free of clay.

1 in/h = 2.54 cm/h

C.3.1.4.1 Flooding Basin Techniques

Where pilot basins have been used for determination of infiltration, the plots have generally ranged from 10 ft^2 (0.9 m^2) to 0.25 acre (0.1 ha). Larger plots are provided with a border arrangement for application of the water. If the plots are filled by hose, a canvas or burlap sack over the end of the hose will minimize disturbance of the soil [7]. Although basin tests are desirable, and should be used whenever possible, there probably will not arise many opportunitites to do so because of the large volumes of water needed for measurements. A sample basin is shown in Figure C-10. In at least one known instance, pilot basins of large scale (5 to 8 acres or 2 to 3.2 ha) were used to demonstrate feasibility and then were incorporated into the larger full-scale system [13].

FIGURE C-10

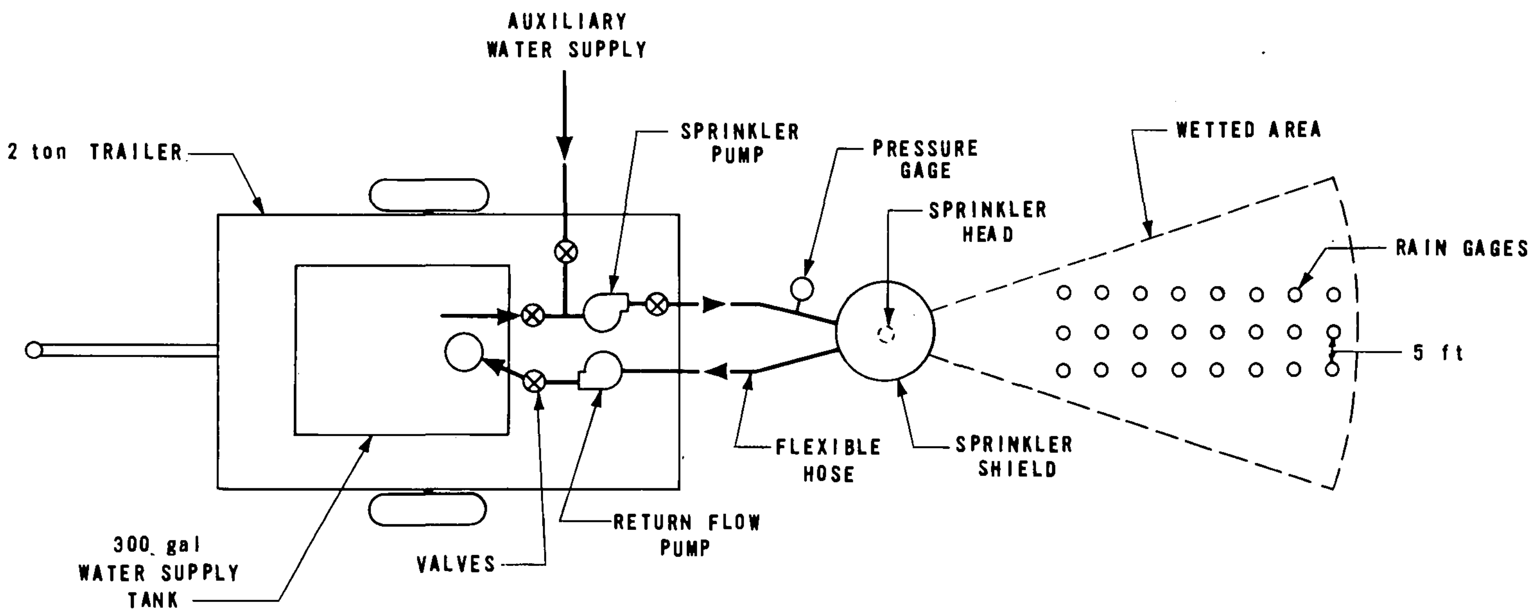
FLOODING BASIN USED FOR MEASURING INFILTRATION

C.3.1.4.2 Sprinkler Infiltrometers

Sprinkler infiltrometers are used primarily to determine the limiting application rate for systems using sprinklers. To measure the soil intake rate for sprinkler application, the method presented by Tovey and Pair can be used [14]. The equipment needed includes a trailer-mounted water recirculating unit, a sprinkler head operating inside a circular shield with a small side opening, and approximately 50 rain gages. A schematic diagram of a typical sprinkler infiltrometer is presented in Figure C-11. A 2 ton (1 814 kg) capacity trailer houses a 300 gallon (1 135 L) water supply tank and 2 self-priming centrifugal pumps. The sprinkler pump should have sufficient capacity to deliver at least 100 gal/min (6.3 L/s) at 50 lb/in.^{2} (34.5 N/cm^2) to the sprinkler nozzle, and the return flow pump should be capable of recycling all excess water from the shield to the supply tank. The circular sprinkler shield is designed to permit a revolving head sprinkler to operate normally inside the shield. The opening in the side of the shield

FIGURE C-11

LAYOUT OF SPRINKLER INFILTROMETER [14]



1 ton = 907.2 Kg
1 ft = 0.305 m
1 gal = 3.785 L

restricts the wetted area to about one-eighth of a circle. Prior to testing, the soil in the wetted area is brought up to field capacity. Rain gages are then set out in rows of three spaced at 5 ft (1.5 m) intervals outward from the sprinkler in the center of the area to be wetted. The sprinkler is operated for about 1 hour. The intake of water in the soil at various places between gages is observed to determine whether the application rate is less than, greater than, or equal to the infiltration rate. The area selected for measurement of the application rate is that where the application is equal to the infiltration rate, i.e., where the applied water just disappears from the soil surface as the sprinkler jet returns to the spot. At the end of the test (after 1 hour), the amount of water caught in the gages is measured and the intake rate is calculated. This calculated rate of infiltration is equal to the limiting application rate that the soil system can accept without runoff.

C.3.1.4.3 Cylinder Infiltrometers

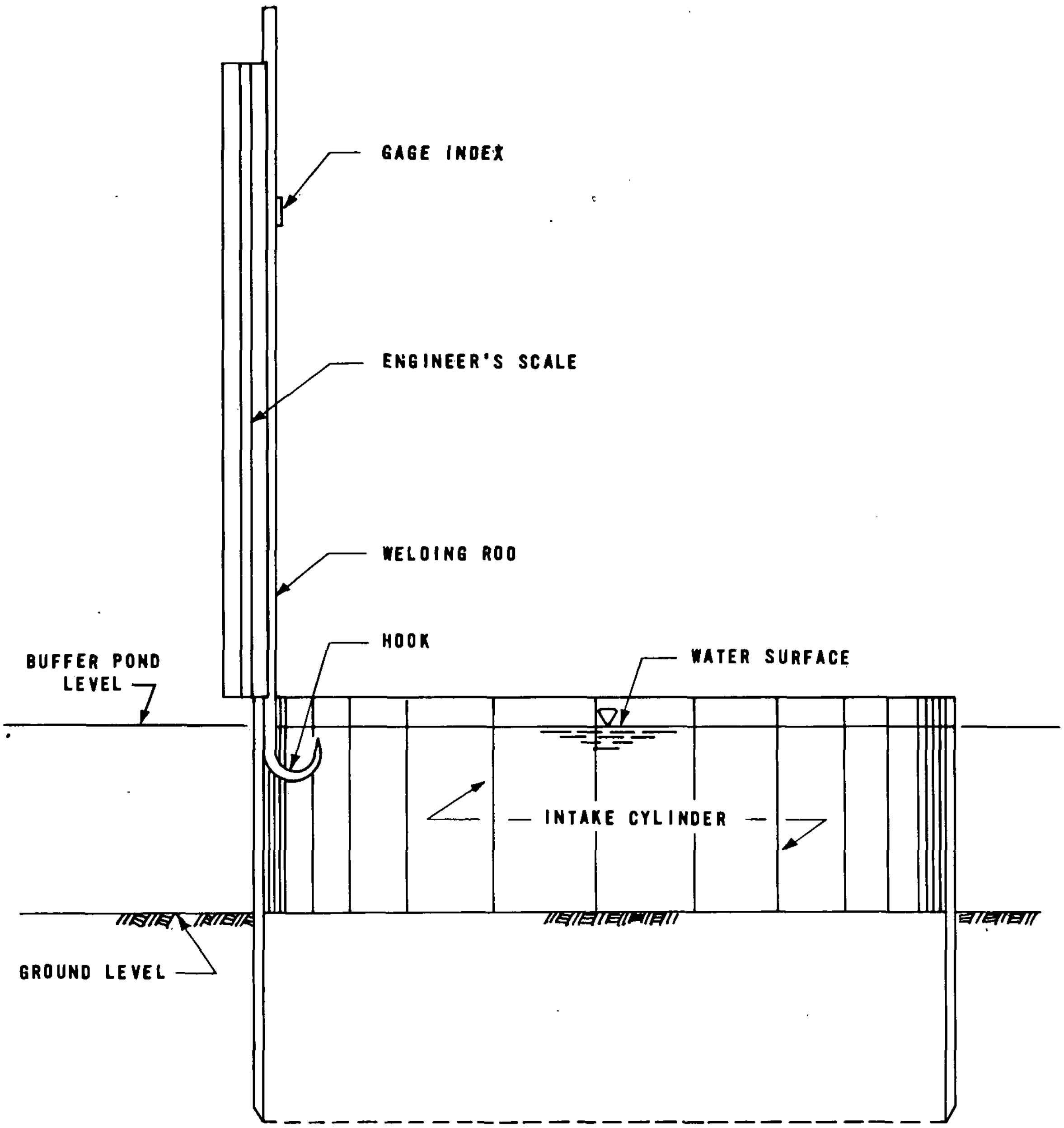
A useful reference on cylinder infiltrometers is Haise et al. [15]. The basic technique, as currently practiced, is to drive or jack a metal cylinder into the soil to a depth of about 6 in. (15 cm) to prevent lateral or divergent flow of water from the ring. The cylinder should be 6 to 14 in. (15 to 35 cm) in diameter and approximately 10 in. (25 cm) in length. Divergent flow is further minimized by means of a "buffer zone" surrounding the central ring. The buffer zone is commonly provided by another cylinder 16 to 30 in. (40 to 75 cm) in diameter driven to a depth of 2 to 4 in. (5 to 10 cm) and kept partially full of water during the time of infiltration measurements from the inner ring. Alternatively, a buffer zone may be provided by diking the area around the intake cylinder with low (3 to 4 in. or 7.5 to 10 cm) earthen dikes.

The quantity of water that might have to be supplied to the double-cylinder system during a test can be substantial and might be considered a limitation of the technique. For highly permeable soil, a 1 500 gal (5 680 L) tank truck might be needed to hold a day's water supply for a series of tests. The basic configuration of the equipment during a test is shown in Figure C-12.

This technique is thought to produce data that are at least representative of the vertical component of flow. In most soils, the infiltration rate will decrease throughout the test and approach a steady state value asymptotically. This may require as little as 20 to 30 minutes in some soils and several hours in others. The test cannot be terminated until the steady state is attained or else the results are meaningless (see Figure C-7).

FIGURE C-12

CYLINDER INFILTROMETER IN USE



The following precautions concerning the cylinder infiltrometer test are noted.

1. If a sprinkler or flood application is planned, the cylinders should obviously be placed in surficial materials. If rapid infiltration is planned, pits must be excavated to expose lower horizons that will constitute the bottoms of the basins.

2. If a more restrictive layer is present below the intended plane of infiltration and this layer is close enough to the intended plane to interfere, the infiltration cylinders should be embedded into this layer to ensure a conservative estimate.

3. The method of placement into the soil may be a serious limitation. Disturbance of natural structural conditions (shattering or compaction) may cause a large variation in infiltration rates between replicated runs. Also the interface between the soil and the metal cylinder may become a seepage plane, resulting in abnormally high rates. In cohesionless soils (sands and gravels), the poor bond between the soil and the cylinder may allow seepage around the cylinder and cause "piping." This can be observed easily and corrected, usually by moving a short distance to a new location and trying again. Variability of data caused by cylinder placement can largely be overcome by leaving the cylinders in place over an extended period during a series of measurements [7].

Knowledge of the ratio of the total quantity of water infiltrated to the quantity of water remaining directly beneath the cylinder is essential if one is interested only in vertical water movements. If no correction is made for lateral seepage, the measured infiltration rate in the cylinder will be well in excess of the "real" rate [16]. Several investigators have studied this problem of lateral seepage and have offered suggestions for handling it [16, 17, 18].

As pointed out by Van Schilfgaarde [19], measurements of hydraulic conductivity on soil samples often show wide variations within a relatively small area. Hundred-fold differences are common on some sites. Assessing hydraulic capacity for a project site is especially difficult because test plots may have adequate capacity when tested as isolated portions, but may prove to have inadequate capacity after water is applied to the total area for prolonged periods. Parizek has observed that problem areas can be anticipated more readily by field study following spring thaws or prolonged periods of heavy rainfall and recharge [20]. Runoff, ponding, and near saturation conditions may be observed for brief periods at sites where drainage problems are likely to occur after extensive application begins.

Although far too few extensive tests have been made to gather meaningful statistical data on the cylinder infiltrometer technique, one very comprehensive study is available from which tentative conclusions can be drawn. Burgy and Luthin reported on studies of three 40 by 90 ft (12.2 x 27.4 m) plots of Yolo silt loam characterized by the absence of horizon development in the upper profile [11]. The plots were diked with levees 2 ft (0.6 m) high. Each plot was flooded to a depth of 1.5 ft (0.5 m), and the time for the water to subside to a depth of 0.5 ft (0.15 m) was noted. The plots were then allowed to drain to the approximate field capacity and a series of cylinder infiltrometer tests--357 total--were made.

Test results from the three basins located on the same homogeneous field were compared. In addition, test results from single-cylinder infiltrometers with no buffer zone were compared with those from double-cylinder infiltrometers. The inside cylinders had a 6 in. (15 cm) diameter; the outside cylinders, where used, had a 12 in. (30 cm) diameter.

For this particular soil, the presence of a buffer zone did not have a significant effect on the measured rates. Consequently, all of the data are summarized on one histogram in Figure C-13. The calculated mean of the distribution shown is 6.2 in./h (15.7 cm/h). The standard deviation is 5.1 in./h (12.9 cm/h).

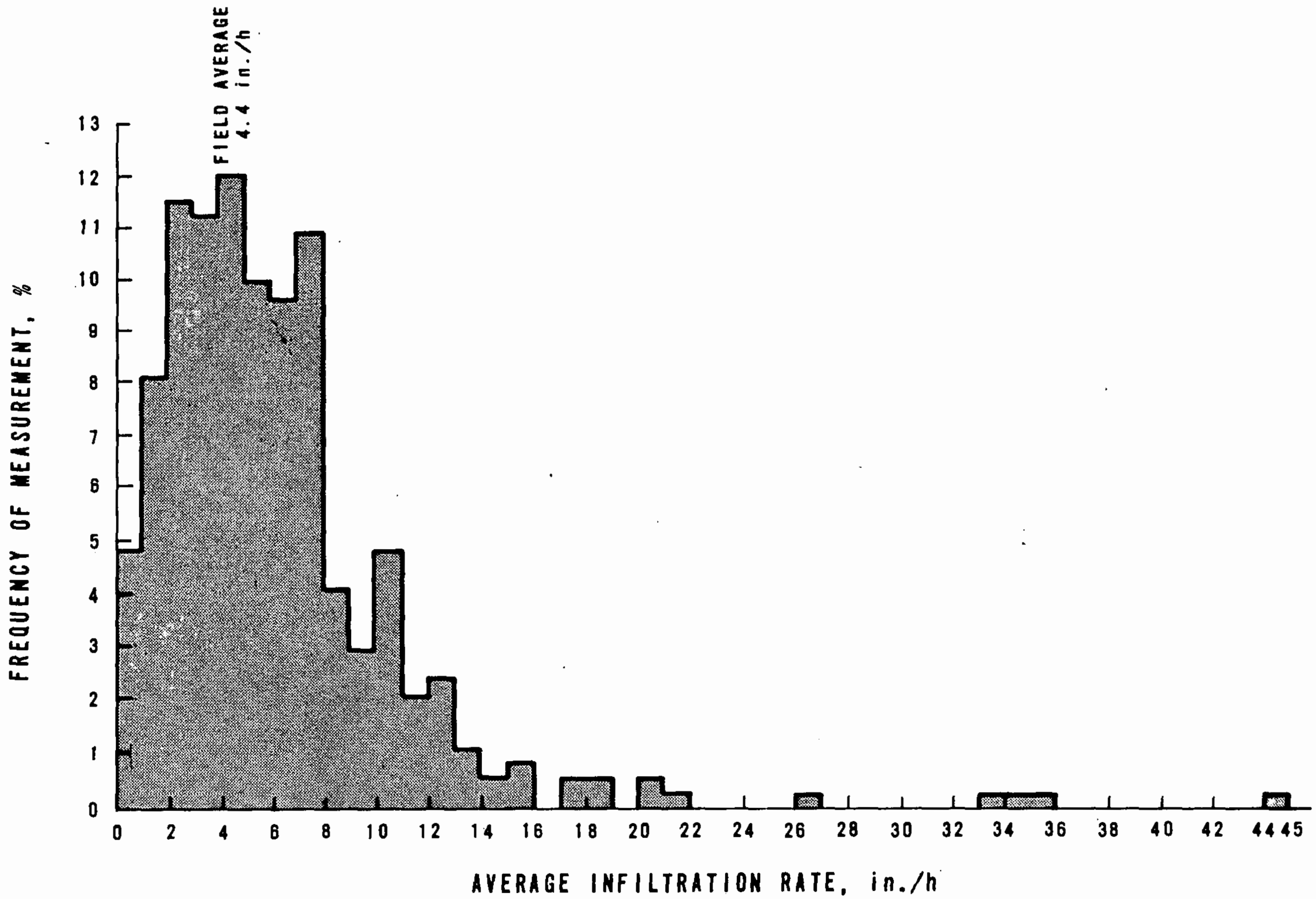
Burgy and Luthin suggest that the extreme high values, while not erroneous, should be rejected in calculating the hydraulic capacity of the site. Physical inspection revealed that these values were obtained when the cylinders intersected gopher burrows or root tubes. Although these phenomena had an effect on the infiltration rate, they should not be included in the averaging process as they carried too much weight.

As a criterion for rejection, Burgy and Luthin suggest omitting all values greater than three standard deviations from the mean value. They further suggest an arbitrary selection of the mean and standard deviation for this procedure based on one's best estimate of the corrected values rather than the original calculations. From inspection of the histogram, these values might be selected as about 5 in./h (12.7 cm/h) and 3.5 in./h (8.9 cm/h), respectively. Thus, all values greater than 5.0 + 3(3.5), or 15.5 in./h (39.4 cm/h) are arbitrarily rejected: a total of 12 of the 357 tests made (3.4%).

Because it is important to provide conservative design parameters for this work, however, it is recommended that all values greater than <u>two</u> standard deviations from the mean be rejected. For the example, this results in the rejection of all values greater than 5.0 + 2(3.5), or 12 in./h (30.5 cm/h) from the average. A recomputation using this

FIGURE C-13

VARIABILITY OF INFILTROMETER TEST RESULTS ON RELATIVELY HOMOGENEOUS SITE [20]



1 in./h = 2.54 cm/h

criterion provides a mean of 5.1 in./h (12.5 cm/h) and a standard deviation of 2.8 in./h (7.1 cm/h). This average value is within 16% of the "true" mean value of 4.4 in./h (11.2 cm/h) as measured during flooding tests of the entire plot.

The main question to be answered now is, how many individual tests must be made to obtain an average that is within some given percent of the true mean, say at the 90% confidence interval? The answer has been provided by statisticians using the Student "T" distribution. Details of the derivation are omitted here but can be found in most standard texts on statistics.

The results of two typical sets of computations are summarized in Figures C-14 and C-15. The two sets of curves are for 90% and 95% confidence intervals. The confidence interval and the desired precision are, of course, basic choices that the engineer must make. A 90% confidence in the measured mean, which is within 30% of the true mean, may be sufficient for small sites where neighboring property is available for expansion if necessary. On the other hand, 95% confidence that the measured value is within 10% of the true mean may be more appropriate for larger sites or for sites where expansion will not be easily accomplished once the project is constructed.

The coefficient of variation will have to be estimated from a few preliminary tests because it is the main plotting parameter in these figures. As an example, for the adjusted distribution of Burgy and Luthin's data with a coefficient of variation estimated at 0.55, at least 23 separate tests would be required to have 90% confidence that the computed mean would be within 20% of the true mean value of infiltration. Obviously, time and budget constraints must be considered in making the confidence and accuracy determinations; 3 to 4 man-days of work might be required to make 23 cylinder infiltrometer tests.

C.3.1.4.4 Lysimeters

Lysimeter studies, using either undisturbed cores (cohesive soils) or disturbed samples compacted carefully to, or near, the field bulk density of the undisturbed sample, may have potential for bridging the very large gap between short-term field tests with clean water and long-term pilot scale field studies with the actual wastewater. The configuration of a typical lysimeter is shown in Figure C-16. Smaller diameters, down to 3 or 4 in. (7.6 or 10.2 cm), have been used with success, especially for relatively undisturbed cores. The gravel layer shown in the figure is artificial and was provided only to prevent clogging at the outlet. Screens and perforated or porous plates have been used for the same purpose in other lysimeter designs.

FIGURE C-14

NUMBER OF TESTS REQUIRED FOR 90% CONFIDENCE
THAT THE CALCULATED MEAN IS WITHIN STATED
PERCENT OF THE TRUE MEAN

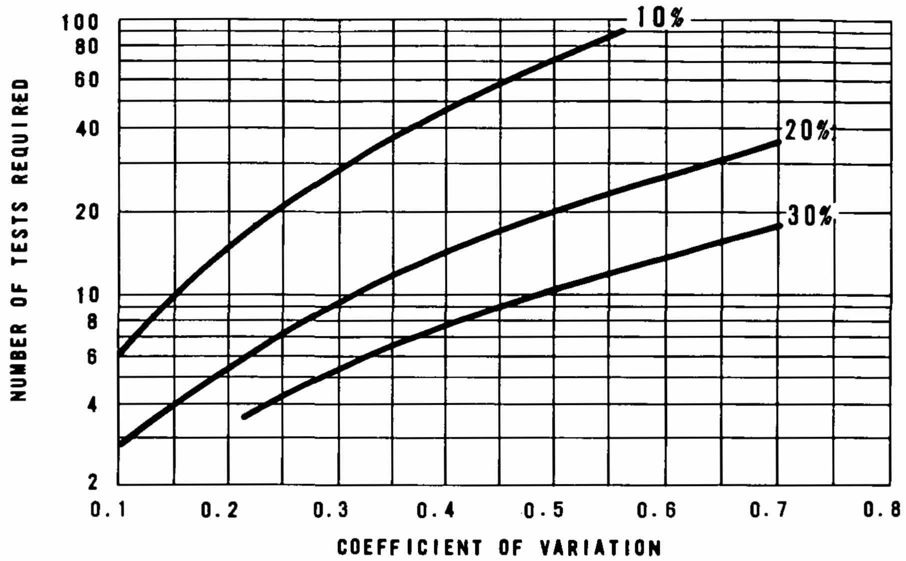


FIGURE C-15

NUMBER OF TESTS REQUIRED FOR 95% CONFIDENCE
THAT THE CALCULATED MEAN IS WITHIN STATED
PERCENT OF THE TRUE MEAN

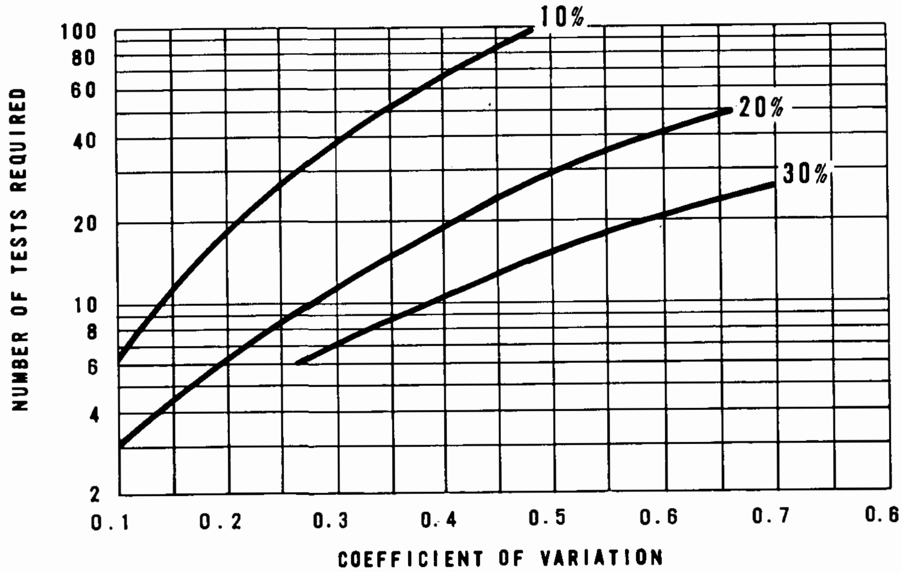
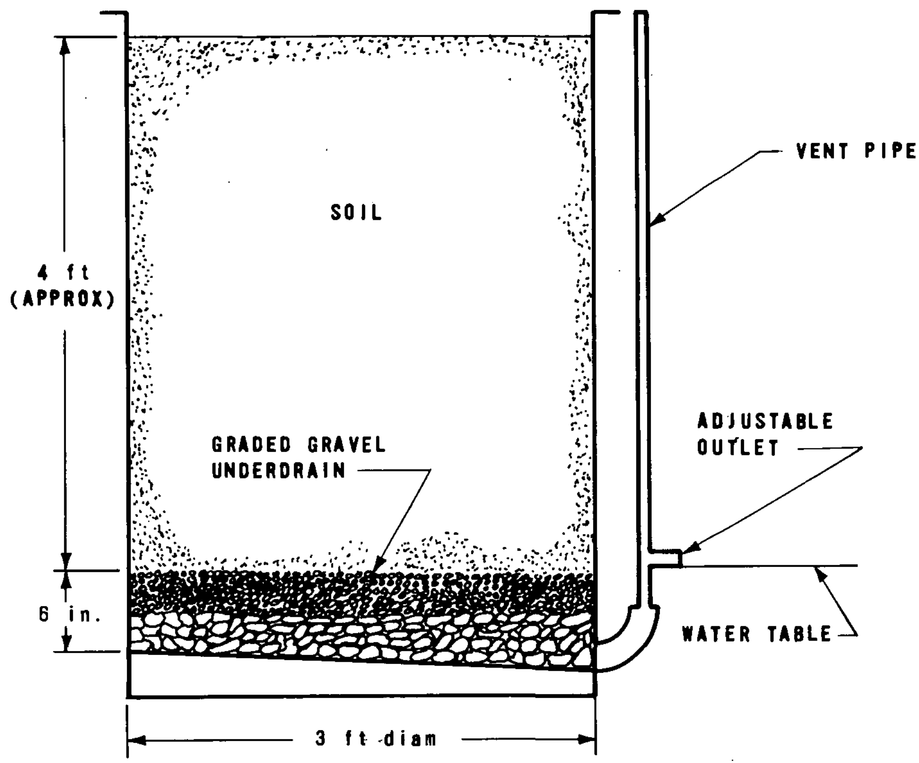


FIGURE C-16

LYSIMETER CROSS-SECTION



1 ft = 0.305 m
1 in. = 2.54 cm

If clean water is used in lysimeter tests, close agreement with the results of cylinder infiltrometer results should be obtained, provided that the infiltrometer tests were made carefully and with sufficient replicates (usually 6 to 12) [21]. With actual wastewater, however, the results will not match. In a study by Ongerth and Bhagat, the 18-inch (45.7 cm) diameter lysimeter, loaded at somewhat less than 100 lb of BOD/acre·d (112 kg/ha·d), averaged about 5 to 10% of the infiltration rates observed on the undisturbed soils using cylinder infiltrometers and clean water [22]. Follow-up studies on a pilot basin of approximately 0.25 acre (0.10 ha) showed that rates significantly higher than those observed in the lysimeters could be sustained [23]. After 1 year of operation the infiltration rate from the pilot basin has averaged about 25% of those measured by the original cylinder infiltrometer testing on this site. This is almost three times the rate predicted from the lysimeter tests. Exact reasons for these differences are not known, but the packing of the disturbed soil into the lysimeters is probably a major factor. This problem will be more critical for fine textured soils. Much more information from studies like the one by Ongerth and Bhagat is necessary before general conclusions can be drawn.

C.3.2 Relation Between Infiltration and Vertical Permeability

Percolation, the movement of water through the soil, is a distinctly different property from infiltration, the movement of water through the soil surface into the soil. The measurement of the vertical component of percolation is called the vertical permeability. In a study by Bouwer [24] it has been shown that the steady state value of infiltration of secondary wastewater effluents containing approximately 25 mg/L suspended solids is about one-half of the potential saturated vertical permeability.

Although the measured infiltration rate on the particular site may decrease in time due to surface clogging phenomena, the subsurface vertical permeability at saturation will generally remain constant. That is, clogging in depth does not generally occur. Thus, the short-term measurement of infiltration serves reasonably well as an estimate of the long-term saturated vertical permeability if infiltration is measured over a large area. Once the infiltration surface begins to clog, however, the flow beneath the clogged layers tends to be unsaturated and at unit hydraulic gradient.

C.3.3 Measurement of Soil Vertical Permeability

The rate at which water percolates through the soil profile during application depends on the average saturated permeability (K_s) of the profile. If the soil is uniform, K is constant with depth, and any differences in measured values of K are due to errors in the measurement technique. Average K then may be computed as the arithmetic mean:

$$K_{am} = \frac{K_1 + K_2 + K_3 \ldots K_n}{n} \qquad \text{(C-4)}$$

where K_{am} = arithmetic mean permeability

Many soil profiles approximate a layered series of uniform soils with distinctly different K values, generally decreasing with depth. For such cases, it can be shown that average K is represented by the harmonic mean of the K values from each layer [25]:

$$K_{hm} = \frac{D}{\frac{d_1}{K_1} + \frac{d_2}{K_2} + \ldots \frac{d_n}{K_n}} \qquad \text{(C-5)}$$

where D = soil profile depth

d_n = depth of nth layer

K_{hm} = harmonic mean permeability

If a bias or preference for a certain K value is not indicated by statistical analysis of field test results, a random distribution of K for a certain layer or soil region must be assumed. In such cases, it has been shown that the geometric mean provides the best estimate of the true K [25]:

$$K_{gm} = (K_1 \cdot K_2 \cdot K_3 \cdots K_n)^{1/n} \qquad \text{(C-6)}$$

where K_{gm} = geometric mean permeability

Methods available to measure vertical saturated permeability in the soil region above a water table, or in the absence of a water table, include the double tube [26, 27], the gradient intake [27, 28], and the air entry permeameter [29].

Each method requires wetting of the soil to obtain K values. During wetting, air is often trapped in the soil pores, and long periods of infiltration are necessary to achieve true saturation. Thus, the measured K value (K_r) is usually less than the saturated permeability (K_s). The air entry permeameter measures K in the wetted zone during wetting, and the measured K is about 0.5 K_s [29]. The double tube and gradient intake methods required longer periods of infiltration, and the soil becomes more nearly saturated. Thus, the K value determined with these two methods may be somewhere between 0.5 K_s and K_s.

In the gradient intake and double tube methods, permeability is measured at the bottom of an auger hole, so these methods can be used for measuring K at different depths in a profile. The air entry permeameter is a surface device, and pits or trenches must be dug if it is to be used for measuring K at greater depths.

C.4 Groundwater Flow Investigations

Groundwater movement through soil and rock is important to the design and operation of land treatment systems, especially rapid infiltration systems. Quantities and qualities of subsurface water will likely be altered by rapid infiltration. Estimating subsurface water flow by indirect surface methods and by more direct subsurface methods is described in this section. The use of hydraulic models to predict groundwater movement also is described briefly.

C.4.1 Groundwater Elevation Maps

Groundwater elevation maps are constructed by interpolating between measured elevations in wells and drawing contour lines of equal elevation. The movement of the groundwater is in a direction perpendicular to the contours from the higher to the lower elevations. The flowrate can be estimated using Darcy's law, by measuring the length dl between groundwater elevation contours dH .

Groundwater maps are used to identify zones of discharge and recharge. Since groundwater flows downgradient, domes, hills, and ridges represent recharge zones; and basins, troughs, and valleys represent discharge zones. Recharge and discharge zones are likely to occur where aquifers are exposed at the earth surface, where lakes and streams intersect shallow water tables, and where there is concentrated agricultural, industrial, or municipal land use. Groundwater discharge can occur because of vertical leakage along faults or other boundaries.

C.4.2 Surface Methods of Estimating Hydrologic Properties

Indirect surface methods can often be used to provide qualitative data on subsurface hydrologic properties. These methods include: (1) earth resistivity, (2) remote sensing, and (3) soil and geologic surveys.

C.4.2.1 Earth Resistivity

Earth resistivity surveys are useful in determining shallow water tables. These and other geophysical, geochemical, and geological surveys are reviewed by Maxey [30]. Resistivity surveys are inexpensive and yield qualitative subsurface data.

C.4.2.2 Remote Sensing

Remote sensing by aerial and satellite photography can be used to estimate subsurface hydrologic properties from interpretation of differences in vegetation, soil associations, and surface drainage. Aerial photographs are usually available from the SCS soil surveys or from land use surveys. Satellite photography, a more recent technique, and computer-generated maps can be used to estimate subsurface hydrology from interpretation of surface features, such as soil moisture.

C.4.2.3 Soil and Geologic Surveys

Existing soil surveys often include information on geologic features, depth to groundwater, and areas of poor drainage. Geologic surveys, as developed by USGS, include discussions of climate, land use, geography, physical geology, mineralogy, petrology, structural geology, historical geology, paleontology, and economic geology of an area. Methods of making geologic surveys are discussed by Lahee [31] and Compton [32]. Geologic surveys often include numerous measurements of rock thickness, texture, structure, attitude (dip, strike, plunge), and statistical analysis of the data may be given.

Published geologic surveys are useful in describing location, physical make-up, thickness, attitude, and boundaries of geologic units which may be aquifers. They are useful in identifying recharge and discharge areas, subsurface flow directions, surface drainage patterns, water quality problems, and potential hazards for land use. Surveys are produced by federal and state geological surveys and bureaus of mines. They are also available as reports for special engineering, scientific, and educational studies at universities and research centers.

C.4.3 Subsurface Methods of Estimating Hydrologic Properties

Logging methods, aquifer tests, and laboratory tests are among the subsurface methods used to estimate hydrologic properties. They require physical access to the subsurface through wells, pits, or drill holes.

C.4.3.1 Logging Methods

Logging methods are used to estimate texture, porosity, and groundwater circulation and quality. A log is a description of material properties with depth as determined by observations or measurements through a hole or with samples from a hole. Drillers' logs and electrical resistance and potential logs are most common. Changes in soils or soil materials can be correlated with spikes or peaks on the electric log printouts. Various logging methods are reviewed by Jones and Skibitzke [33] and Todd [34]. Professional logging companies publish detailed manuals and research papers. A summary of subsurface logging information obtained by various methods is provided in Table C-4.

C.4.3.2 Aquifer Tests

Aquifer tests by the pumped-well method are performed using a series of wells in the field. The approach is to discharge (pump) or recharge one well at a known rate and to measure the response of water levels in the other wells. Water level responses are then mathematically related to permeability (K) or transmissivity [6].

TABLE C-4

SUBSURFACE LOGGING INFORMATION
OBTAINED BY VARIOUS METHODS [34]

Method	Operation	Information
Drillers' log	Observe well cuttings during drilling	Rock contacts, thickness, description, or type texture. Samples for laboratory tests. Common method.
Drilling-time log	Observe drilling time	Rock texture, porosity.
Resistivity log	Measure electrical resistivity of media surrounding encased hole	Specific resistivity of rocks, porosity, packing, water resistivity, moisture content, temperature, groundwater quality. Correlate with samples for best results. Common method.
Potential log	Measure natural electric potentia, or self-potential	Permeable or impermeable, groundwater quality. Common method.
Temperature log	Measure temperature	Groundwater circulation, leakage.
Caliper log	Measure hole diameter	Hole diameter, rock consolidation, caving zones, casing location.
Current log	Measure current	Groundwater flow velocity, circulation, leakage.
Radioactive log	Measure attenuation of gamma and neutron rays	Consolidation, porosity, moisture content. Common in soil studies, clay or nonclay materials.

Bouwer summarized other variations including the auger-hole, piezometer, and tube methods [35]. In the tube method, the resultant K is for the vertical direction [25, 35]. For the piezometer method, the direction of measured K depends on the ratio of the piezometer height to its diameter. The auger-hole test, which measures principally horizontal permeability, is discussed further in Section C.5.2.2.

C.4.3.3 Laboratory Tests

Laboratory tests of subsurface flow properties are generally not as reliable as field methods because of the errors introduced in sampling and the change in properties due to disturbance in sampling. Laboratory determinations of soil-water characteristics and unsaturated conductivity curves are presented in Taylor and Ashcroft [1], Kirkham and Powers [3], Black [36], and Bouwer and Jackson [37]. Because of the tremendous variability of actual hydrologic properties, a great many laboratory tests must be conducted to provide statistical validity.

C.5 Groundwater Mound Height Analysis

C.5.1 Introduction

If water that infiltrates the soil and percolates vertically through the zone of aeration encounters a water table or an impermeable (or less permeable) layer, a groundwater "mound" will begin to grow. If the mound height continues to grow, it may eventually encroach on the zone of aeration to the point where renovation capacity is affected. Further growth may result in intersection of the mound with the soil surface, which will reduce infiltration rates. This problem can usually be identified and analyzed before the system is designed and built if the proper geologic and hydrologic information is available for analysis.

Parizek [20] and Spiegel [38] support the concept of hydrogeologic systems analysis to define the probable effects of a project on local and regional water table configurations. Modeling may be expected to delineate areas likely to be flooded, demonstrating the need for drainage facilities and their location. Thus, contingency plans to eliminate potential drainage problems can be formulated at the time projects are designed.

Practically all analyses of drainage problems have been limited to the behavior of the water table, which of course responds to a wetting event as shown in Figures C-17 and C-18. Irrigation experts recognize that water table position alone is not a satisfactory criterion in their work. If the present state of knowledge would permit, they might well redirect their attentions to the moisture content of the root zone. The situation is similar with respect to land application of wastewater. One is really less interested in the position of the water table at any time than in the onset of anaerobiosis in the soil voids and the breakdown of renovation capacity. Analysis of the latter is so complex, however, that we will have to be content for the present to simply be able to control the water table, or to know how high it will rise under given loading conditions.

Analysis of the growth and decay characteristics of groundwater mounds induced by percolating waters is a complex, mathematically sophisticated process. The problem has been attacked in several ways, including analytical, analog, and digital modeling. Several empirical equations representing gross approximations have also appeared. A complete review of all the work in this area is well beyond the scope of this appendix. Rather, the input data generally required for the analysis will be discussed, and a short review will be provided of published studies that should prove useful to the user searching for a method to suit a particular problem. Only simple geometries, known to recur frequently in practical applications, are covered by these references.

FIGURE C-17

MOUND DEVELOPMENT FOR STRIP RECHARGE [33]

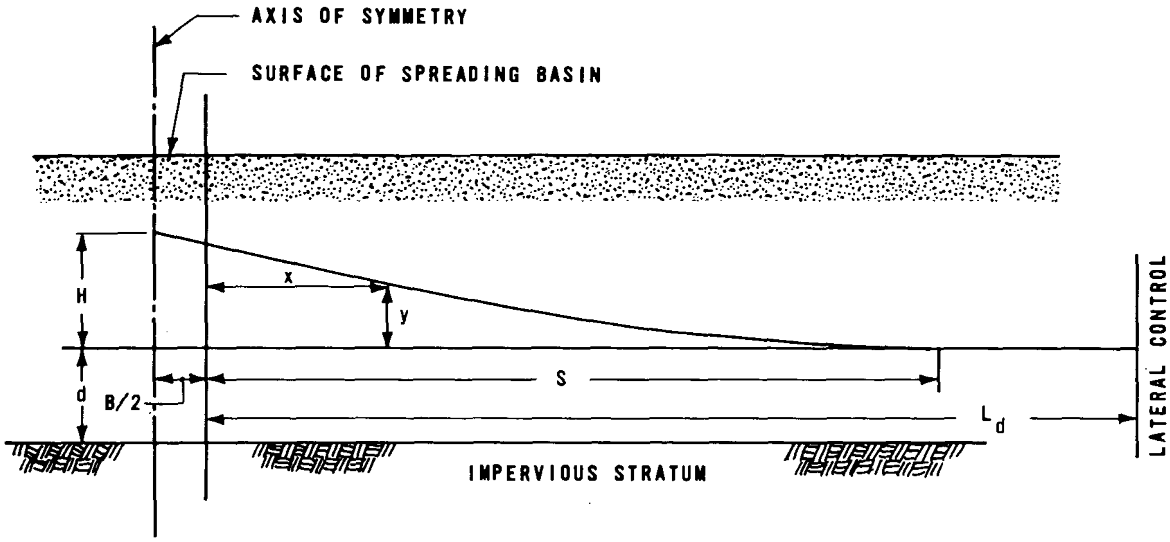
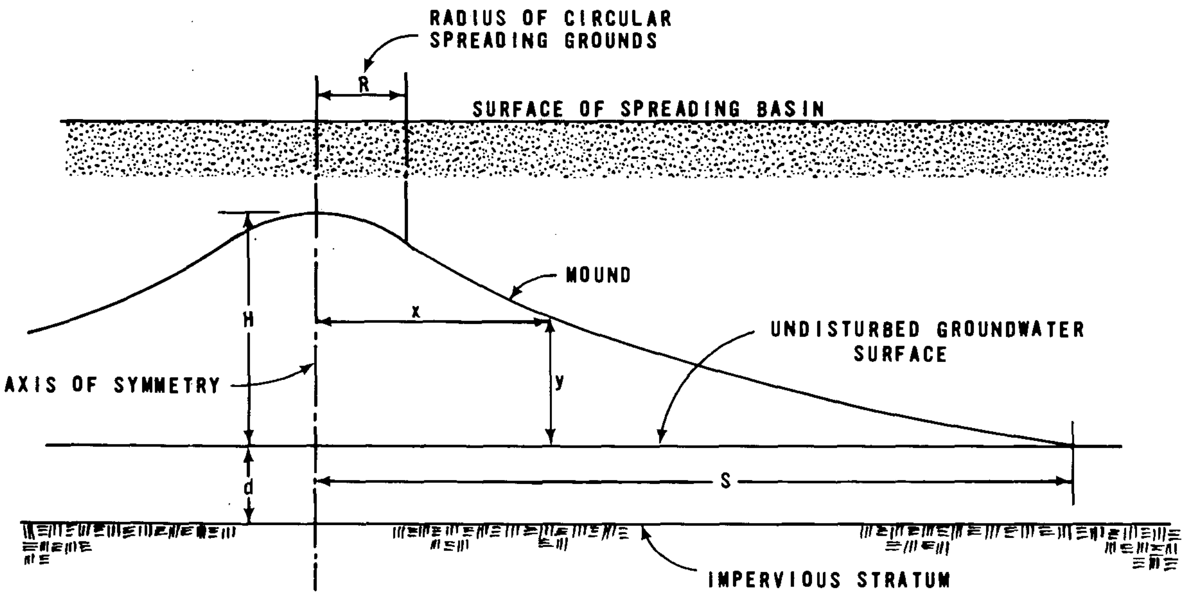


FIGURE C-18

MOUND DEVELOPMENT FOR CIRCULAR RECHARGE AREA



C.5.2 Data Requirements

Before proceeding with any analysis, a good deal of data must be collected. The following list of input information requirements is abstracted from the work of Baumann [39] and Parizek [20]. Pertinent comments by other investigators are included where appropriate to provide additional insight or clarification.

1. Coefficients of vertical and horizontal permeability. If the site is large and/or its soils heterogeneous, a spatial distribution of these values will be needed. Klute [40] and Papadopulos et al. [41] both stress the need for <u>extensive</u> rather than <u>intensive</u> methods of characterization, and meaningful average conductivity functions, together with probability statements as to deviations from the mean value.

2. Specific yield (drainable porosity) of deposits saturated or dewatered by water table fluctuations.

3. Vertical distances between initial groundwater surface and ground surface, and between initial groundwater surface and impervious stratum.

4. Slope of impervious strata.

5. Horizontal distance to a control or discharge surface.

6. Geometry of recharge area.

7. Rate and duration of spreading (infiltration). Although most, if not all, of the analytic expressions assume a steady supply (constant vertical percolation), Childs has shown that seepage of a series of intermittent recharges is equivalent to that of a single steady application [42].

8. Estimates of the evapotranspiration losses for the areas where groundwater tables will be near the surface.

C.5.2.1 Drainable Voids

The term drainable voids is synonymous with the term "specific yield" used in water well technology. It is the ratio of water that will drain freely from the materials to the total volume of the materials. It may be estimated from data on similar soils (Table C-5), or more preferably, it may be evaluated in the laboratory from soil moisture data at saturation and 0.3 bar tension on undisturbed soil fragments.

TABLE C-5

APPROXIMATE DRAINABLE VOIDS FOR MAJOR SOIL CLASSIFICATIONS

Material	Porosity, %	Drainable voids, %
Clay	45	3
Sand	35	25
Gravelly sand	20	16
Gravel	25	22

In some areas of the United States, the SCS has investigated the soil profile sufficiently to provide a reasonable estimate of drainable voids on a particular site. An outstanding reference, covering 200 typical soils in 23 states, is the USDA Agricultural Research Service Publication 41-144 [2]. This compendium of soil-moisture tension data gives bulk density, total porosity, and saturated vertical permeability values. It also gives soil moisture at 0.1, 0.3, 0.6, 3.0, and 15 bars tension for several depths in the soil profile (down to about 4 ft or 1.2 m in many cases).

Drainable voids can be calculated as the difference between total porosity and the volume percent of moisture at 0.3 bar tension. Other important hydrologic computations can be made as well, using the relationships in Holtan et al. [43]. One important factor that was not discussed in either Reference [2] or [43] was the inherent spatial variability of the basic measurements reported. Nielsen et al. conducted a set of experiments on a 370 acre (150 ha) site to determine the statistical variability of many soil properties affecting its hydrologic behavior [44]. A few of their results, shown in Table C-6, should be of value in developing a sensitivity for this variability.

C.5.2.2 Lateral (Horizontal) Flow

Horizontal permeability is a more difficult parameter to obtain. In field soils, isotropic conditions are rarely encountered, although they are frequently assumed for the sake of convenience. "Apparent" anisotropic permeability often occurs in unconsolidated media because of interbedding of fine-grained and coarse-grained materials within the profile. Such interbedding restricts permeability to vertical flow much more than it does lateral flow [25]. Although the interbedding represents nonhomogeneity, rather than anisotropy, its effects on the permeability of a large sample of aquifer material may be approximated by treating the "aquifer" as homogeneous but anisotropic. A considerable amount of data is available on the calculated or measured relationships between vertical and horizontal permeability for specific sites. The possible spread of ratios is indicated in Table C-7, which is based on field measurements in glacial outwash deposits (Sites 1-5) by Weeks [45] and in a river bed (Site 6) by Bouwer [46]. Both authors claim, with justification, that the reported values would not likely be observed in any laboratory tests with small quantities of disturbed aquifer material.

TABLE C-6

STATISTICAL VARIABILITY OF SEVERAL PHYSICAL PROPERTIES OF SOIL [44]

Property	No. of samples	Mean	Standard deviation	Coefficient of variation, %
Bulk density	720	1.356 g/cm^3	0.104 g/cm^3	7.7
Moisture at saturation, volume fraction	120	0.469	0.035	7.5
Moisture at 200 cm tension	120	0.346	0.072	20.8

It is apparent, then, that if accurate information regarding horizontal permeability is required for an analysis, field measurements will be necessary. Of the many field measurement techniques available, the most useful is the auger hole technique of Van Bavel and Kirkham [47]. Although auger hole measurements are certainly affected by the vertical component of flow, studies have demonstrated that the technique primarily measures the horizontal component [48]. A definition sketch of the measurement system is shown in Figure C-19 and the experimental setup is shown in Figure C-20. The technique is based on the fact that if the hole extends below the water table and water is removed from the hole (by bailing or pumping), the hole will refill at a rate determined by the permeability of the soil, the dimensions of the hole, and the height of water in the hole. With the aid of either formulas or graphs, the permeability is calculated from measured rates of rise in the hole. The total inflow into the hole should be sufficiently small during the period of measurement to permit calculation of the permeability based on an "average" hydraulic head. This is usually the case.

In the formulas and graphs that have been derived, the soil is assumed to be homogeneous and isotropic. However, a modification of the basic technique by Maasland allows determination of the horizontal and vertical components (K_h and K_v) in anisotropic soils by combining auger hole measurements with piezometer measurements at the same depth [48]. If the auger hole terminates at (or in) an impermeable layer, the following equation applies (refer to Figure C-19 for symbols):

$$K_h = 523\ 000\ a^2 \frac{\log_{10}(y_0/y_1)}{\Delta t} \qquad \text{(C-7)}$$

where a = auger hold radius, m
Δt = time for water to rise y, s
K_h = horizontal permeability, m/d
y_0, y_1 = depths defined in Figure C-19, any units, usually cm

If an impermeable layer is encountered at a great depth below the bottom of the auger hole, the equation becomes

$$K_h = \frac{1\ 045\ 000\ da^2}{(2d + a)} \quad \frac{\log_{10}(y_0/y_1)}{\Delta t} \qquad \text{(C-8)}$$

where d = depth of auger hole, m

TABLE C-7

MEASURED RATIOS OF HORIZONTAL TO VERTICAL PERMEABILITY

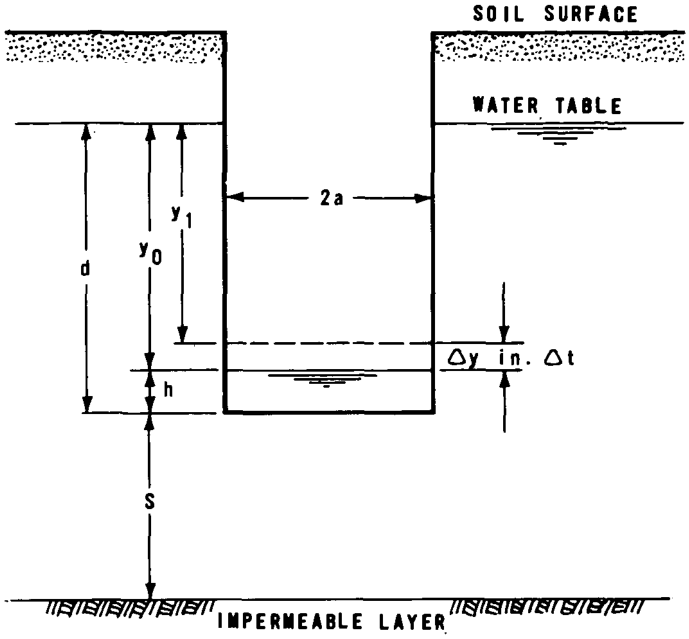
Site	Effective horizontal permeability, K_h, ft/d	K_h/K_v	Remarks
1	138	2.0	Silty
2	247	2.0	.....
3	183	4.4	.....
4	329	7.0	Gravelly
5	237	20.0	Near terminal moraine
6	236	10.0	Irregular succession of sand and gravel layers (from K measurements in field)
6	282	16.0	(From analysis of recharge flow system)

a. Sources: Data on Sites 1 through 5 [45]; data on Site 6 [46].

1 ft/d = 0.305 m/d

FIGURE C-19

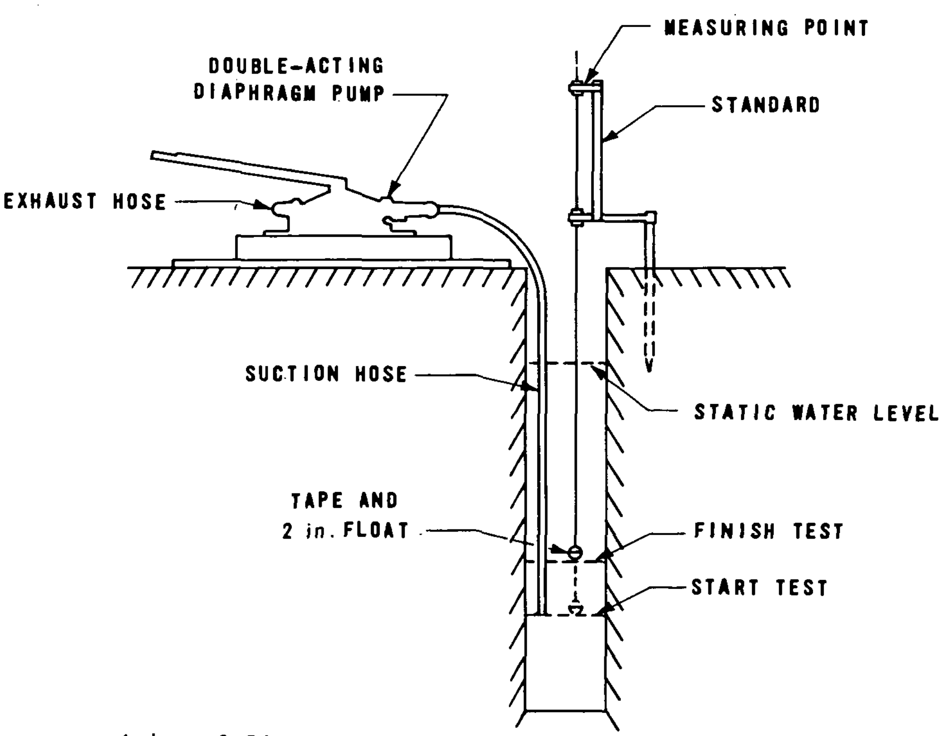
DEFINITION SKETCH FOR AUGER-HOLE TECHNIQUE



1 in. = 2.54 cm

FIGURE C-20

EXPERIMENTAL SETUP FOR AUGER-HOLE TECHNIQUE



1 in. = 2.54 cm

Charts for both cases were prepared by Ernst and are available in the text by Luthin [49]. An alternate formula, claimed to be slightly more accurate, has been developed by Boast and Kirkham [50]. Their equation employs a table of coefficients to account for depth of impermeable or of very permeable material below the bottom of the hole.

There are several other techniques for evaluating horizontal permeability in the presence of a water table. Slug tests, such as described by Bouwer and Rice, can be used to calculate K_h from the Thiem equation after observing the rate of rise of water in a well following an instantaneous removal of a volume of water to create a hydraulic gradient [51]. Certainly pumping tests, which are already familiar to many engineers, would provide a meaningful estimate. Glover presents a comprehensive discussion of pumping tests, as well as other groundwater problems. He also presents example problems and tables of the mathematical functions needed to evaluate permeability from drawdown measurements [52].

There are two limitations to full-scale pumping tests. The first is the expense involved in drilling and installation. Thus, if a well is not already located on the site, the pumping test technique would probably not be considered. If an existing production well fulfills the conditions needed for the technique to be valid, it should probably be used to obtain an estimate. However, this estimate may still require modification through the use of supplementary "point" determinations, especially if the site is very large or if the soils are quite heterogeneous. The possibility that the pumping test will not give results representative of the larger area is the second limitation.

Donnan and Aronovici developed a technique using a small well screen, carefully manufactured to a set of standard specifications [53]. Because these screens have a constant and reproducible flow geometry when inserted below the water table and pumped, standard curves prepared by the authors could be used to compute K_h from flowrate and pressure drop data.

Measurement of horizontal and vertical permeability may occasionally be necessary in the absence of a water table. A typical case might involve the presence of a caliche layer, or other hardpan formation near the surface, which is restrictive enough to vertical flow to result in a perched water table upon application of effluent. In this case, equipment and techniques developed by Bouwer will permit the determination [54]. The required equipment is known as the Tempe Double Tube Hydraulic Permeability Device. Water levels in the inner and outer tube are manipulated to give as estimate of the overall permeability which is some resultant of K_h and K_v (more of K_v). The true value of K_h can be evaluated by inserting piezometers in the double tube system to measure the true K_v. K_h can then be computed. Other methods for measuring K in absence of water table are: shallow well pump-in method (K_h), air entry permeameter (K_v) and infiltration gradient method (K_v) [4].

C.5.3 Models for Two Geometries of Practical Concern

Two recharge area geometries can be expected to recur frequently in practice. The first involves a rectangular area lying roughly perpendicular to the initial direction of groundwater flow. If the length is large relative to the width, the area can be considered a strip and the resulting mound will be two-dimensional, as shown in Figure C-17.

The growth and decay of the two-dimensional mound is given analytically by both Baumann [39] and Marino [55]. Baumann based his solution on the analogy of the flow of heat through a prismatic, nonradiating bar of length L_d with a constant heat source ν at the origin ($X = 0$) and constant temperature at $X = L_d$. The exact solution for the stable mound at $t = \infty$ is a Fourier-series expansion; however, an approximate solution for the mound coordinates can be obtained more simply from the expression

$$Y = \left(\frac{\nu}{Kd} - i\right) \quad (L_d - X) \qquad \text{(C-9)}$$

where ν = the average infiltration rate
i = the slope of the impermeable strata, usually zero

This expression allows calculation of the mound height at either edge of the spreading basin. The maximum height H , at the center of the strip, would have to be estimated from the geometry and the shape of the curve near $X = 0$.

If the spreading area can be readily approximated by a circle, the mound will be three-dimensional. The absence of a circular control for lateral flow makes the problem of defining a stable mound difficult, but solutions are available [39, 56, 57]. Referring to the definition sketch shown in Figure C-18, one solution for the maximum mound height (from reference [57]) is

$$H = \frac{\nu t}{\mu}\left[1 - e^{-u_1} + u_1\int_{u_1}^{\infty} \frac{e^{-u}\,du}{u}\right] \qquad \text{(C-10)}$$

where $u_1 = R^2/4\alpha t$
$\alpha = Kd/\mu$
μ = drainable void volume

and the integral above is the well-known exponential integral, values of which are tabulated in reference [52]. Graphical solutions to the equations of groundwater mound height analysis for several important geometries may be found in reference [58], a publication of the USDA - Agricultural Research Service.

C.6 Control of the Groundwater Table

The need for drainage will be established through some method of groundwater mound height analysis or when the natural fluctuations in the groundwater table are thought to bring it too close to the surface, the latter being a judgment of the designer. For some large projects that do not require a complete system of underdrainage, drainage at a few selected locations may be required. For other projects, drainage may be required only to prevent trespass of wastewater onto adjoining property by subsurface flow.

The drainage design consists of selecting the depth and spacing for placement of the drain pipes or tiles. As a frame of reference, practical drainage systems for wastewater applications will be at depths of 4 to 8 ft (1.2 to 2.4 m), at or in the water table, and spaced 200 ft (60 m) or more apart. Spacing may approach 500 ft (150 m) in sandy soils. Although closer spacings result in better control of the water table, the cost of moving the drains closer together soon becomes prohibitive except for a very few cases.

A definition sketch for the use of the Hooghoudt drain spacing method is shown in Figure C-21. The assumptions of this method are as follows [49]:

1. The soil is homogeneous and of permeability K (horizontal conductivity).

2. The drains are evenly spaced a distance S apart.

3. The hydraulic gradient at any point is equal to the slope of the water table above that point.

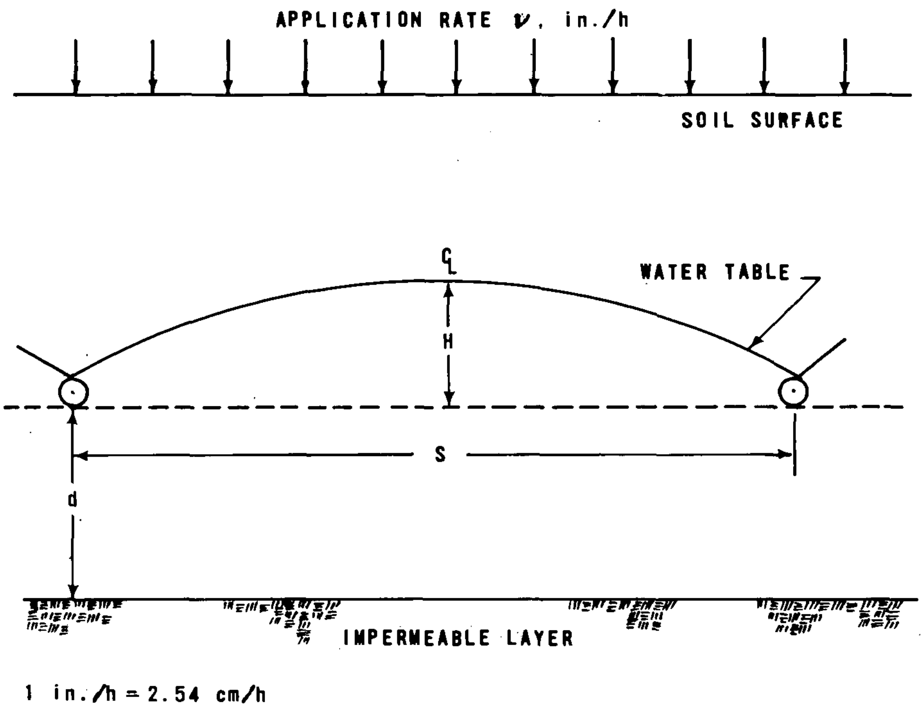
4. Darcy's law is valid.

5. An impermeable layer underlies the drain at a depth d.

6. The rate of replenishment (wastewater application plus natural precipitation) is ν.

FIGURE C-21

DEFINITION SKETCH FOR DRAIN SPACING FORMULA



Omitting all details of the derivation, the final spacing formula is given as

$$S^2 = \frac{4\ KH}{\nu}\ (2d + H) \qquad \text{(C-11)}$$

where H is the maximum height of water table allowable above the drains.

This equation is approximate, and several modifications are possible and/or necessary for particular field situations. In particular, the value of d in Equation C-11 is only equal to the actual depth when the depth is small. Hooghoudt developed a table of "equivalent" depths for large values of d which are to be substituted for the actual value of d in Equation C-11. Curves based on Hooghoudt's analysis are available in Luthin's text [49]. Additional details of drainage design may be found in Luthin and other references (see Section C.8). On occasion, pumped wells have been used for drainage and/or recovery of renovated effluent when the water table is too deep for the use of horizontal drains [46].

C.7 Relationship Between Measured Hydraulic Capacity and Actual Operating Capacity

The relationship between measured hydraulic capacity and actual operating capacity is an extremely important subject which is complicated by the fact that meaningful data are not generally available for analysis. In addition, not every site is hydraulically limited; in fact, most sites probably are not. In many cases, loadings are controlled by management approaches, nitrogen loadings, or other factors. However, for sites that receive relatively low organic loadings (say less than 200 lb/acre·d [224 kg/ha·d] of BOD) and that have no other limiting factors, it would be significant to know the relationship between the highest hydraulic loadings that did not cause problems and the original infiltration rates measured on the soils before the initial applications of wastewater began. Data from several systems are summarized in Table C-8, but they are from a very limited cross-section of soil types and wastewater characteristics, so it would be inappropriate to draw general conclusions from them. More data of this type are required before a meaningful pattern can emerge. As can be seen from the footnotes, only two of the systems are being loaded at or near their maximum acceptance rate in the absence of any other known constraints. Such situations make data interpretation very difficult. At present, it appears that loadings in the range of 5 to 25% of the measured infiltration rate will generally produce a satisfactory result in terms of system hydraulics, no other constraints existing. Operating rates reported in Table C-8 are calculated using total cycle times, including the time allowed for resting and drying. One further point of interest is the hydraulic loading for the Flushing Meadows project which averages about 25% of potential infiltration. It is believed that this may be the peak value (as a percentage) attainable anywhere because of the nearly ideal conditions at the Phoenix test site [24].

TABLE C-8

SUMMARY OF MEASURED INFILTRATION RATES AND OPERATING RATES FOR SELECTED LAND APPLICATION SYSTEMS [59]

System type	Soil texture class	Type of wastewater	Infiltration rate I, in./h	Operating rate ν, in./h	ν/I, %
Slow rate	Silt loam	Steam peel potato	0.8-0.9	0.03[a]	03.4
Slow rate	Loam	Secondary effluent meat packing	0.8-2.10	0.03[b]	1.4-3.8
Slow rate	Silt loam	Secondary municipal	0.2-0.3	0.01[c]	4.0
Rapid infiltration	Sand	Oily cooling water	9.0-14.4	2.8	19.0-31.0
Rapid infiltration	Gravelly sand	Secondary kraft mill	28.0-55.0	0.29[d]	0.5-1.0
Rapid infiltration	Loamy sand	Secondary kraft mill	1.5-9.7	0.19[d]	2.0-12.7
Rapid infiltration	Sand gravel	Secondary municipal	8.8-11.8	0.55	4.6-6.2

a. Limited by poor drainage (high water table).
b. Limited arbitrarily to irrigate larger acreage for hay production.
c. Limited by nitrogen loading considerations.
d. Limited by organic loading (biological clogging problems).

1 in./h = 2.54 cm/h

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APPENDIX D

PATHOGENS

D.1 Introduction

Because land treatment must be compared equitably to conventional wastewater discharge systems in facilities planning, a need exists to evaluate the relative health risks associated with land treatment versus conventional systems. Unfortunately, there are few data on this aspect. Since the level of enteric disease in the United States is relatively low, wastewater in the United States would be expected to contain low levels of pathogens compared with that in many regions of Asia, Africa, and South America. However, the continual occurrence of waterborne disease caused by wastewater-contaminated water in the United States [1] indicates that sufficient numbers of pathogens are present to be a public health concern.

D.2 Relative Public Health Risk

Sufficient data are not available to show whether or not land treatment is a greater health risk than conventional treatment and discharge systems. The paucity of information available on disease caused by wastewater treatment processes may reflect either the absence of a problem, lack of intensive surveillance, or the insensitivity of present epidemiological tools to detect recurrent small-scale incidents of disease. It should be emphasized, however, that no incidents of disease have been documented from a planned and properly operated land treatment system. Comparative epidemiological studies on human populations associated with conventional as well as land treatment systems are needed to provide sufficient and reliable data on which regulations could be based. Thus, evaluation of potential health hazards must rest on our knowledge of the occurrence of pathogens in wastewater and their fate during land treatment.

D.3 Pathogens Present in Wastewater

A large variety of disease-causing microorganisms and parasites are present in domestic wastewater. These include pathogenic bacteria, viruses, protozoa, and parasitic worms. The number of individual species of pathogens is high. For example, over 100 different types of viruses are known to be excreted in human feces. The relative concentrations of these pathogens are highly variable, being dependent on a number of complex factors, but pathogens are almost always present in untreated wastewater in sufficient numbers to be a public health concern. Thus, it is necessary to identify and put into perspective any potential routes of disease transmission involved in land treatment as well as conventional treatment of wastewater so that appropriate safeguards can be assured. It is also necessary to identify the relative risk of infection for humans and animals from these and other sources.

Most of the studies reported in this appendix involve applications of untreated wastewater (often simply referred to as wastewater) to the land in an unplanned manner, or deliberate artificial seeding of bacteria or viruses in high concentrations to the soil to determine their survival or movement under various conditions. Because each study had different objectives, types of soil, climatic factors, types of organisms, and methods of detection, the results must be interpreted accordingly. In citing a particular study for purposes of establishing safeguards or standards, all of the conditions relevant to that study should be defined. In general, safeguards should be established on a case-by-case basis so that the relative risk of disease transmission in each situation can be evaluated individually.

D.3.1 Bacteria

The most common bacterial pathogens found in wastewater include strains of *Salmonella*, *Shigella*, enteropathogenic *Escherichia coli* (*E. coli*), *Vibrio*, and *Mycobacterium*. The genus *Salmonella* includes over 1 200 different strains, many of which are pathogenic for both man and animals. Members of this group are commonly isolated from wastewater and polluted receiving waters. *Salmonella typhi* has been responsible for incidents of typhoid fever associated with wastewater-contaminated drinking water [1] and with the eating of raw vegetables grown on soil fertilized with untreated wastewater [2]. Other members of the group are associated with paratyphoid fever and acute gastroenteritis.

Shigella organisms are the most commonly identified cause of acute bacterial diarrheal disease in the United States [3]. Waterborne spread of the organisms can cause outbreaks of shigellosis, commonly known as bacillary dysentery, which occur frequently in undeveloped countries and occasionally in developed countries. Unlike *Salmonella*, *Shigella* organisms are rarely found in animals other than man.

Cholera is caused by the organism *Vibrio cholerae*. In Israel in 1970 cases of cholera were attributed to the practice of irrigating vegetable crops with untreated wastewater. This practice is contrary to regulations of the Ministry of Health [4]. There were no reported cases of cholera in the United States between 1911 and 1973, until a single case occurred in Texas with no known source. Though individual cases of cholera may arise in international travelers, the likelihood of cholera being transmitted by wastewater land application projects is minimal.

D.3.2 Viruses [5]

Viruses, the smallest wastewater pathogens, consist of a nucleic acid genome enclosed in a protective protein coat. Viruses that are shed in fecal matter, referred to as enteric viruses, are characterized by their ability to infect tissues in the throat and gastrointestinal tract, but they are also capable of replicating in other organs of the body. They include the true enteroviruses (polio-, echo-, and coxsackieviruses), reoviruses, adenoviruses, and rotaviruses, as well as the agent of infectious hepatitis. These viruses can cause a wide variety of diseases, such as paralysis, meningitis, respiratory illness, myocarditis, congenital heart anomalies, diarrhea, eye infections, rash, liver disease, and gastroenteritis. Almost all of these viruses also produce inapparent or latent infections. This makes it difficult to recognize them as being waterborne. Documented cases of waterborne viral disease have largely been limited to infectious hepatitis, mainly because of the explosive nature of the cases and the characteristic nature of the disease.

Knowledge of the actual number and concentration of human pathogenic viruses in wastewater is inadequate, due to lack of adequate and standardized sampling and analytical procedures. Furthermore, the methodology for detecting and monitoring many of these agents has not yet been developed. This probably accounts for the fact that almost 60% of all documented cases of disease attributable to drinking water in the United States has been reported to be caused by agents as yet not isolated in the laboratory. The lack of documentation reflects the difficulty in sampling an infectious agent in the carrier at the time of reported illness. It must be borne in mind that viruses as a group are generally more resistant to environmental stresses and chlorination than pathogenic bacteria.

Also present in wastewater are large numbers of bacterial viruses known as bacteriophages. These viruses are not pathogenic for man, but they have been studied as models for animal virus behavior because of the ease with which they can be detected. However, it should not be assumed that all studies using bacteriophages can be directly applied to human pathogenic viruses.

D.3.3 Other Pathogens

Protozoans pathogenic to man and capable of transmission in wastewater are _Entamoeba histolytica_, the agent of amoebic dysentery; _Naegleria gruberi_, which may cause fatal meningoencephalitis; and _Giardia lambia_, which produces a variety of intestinal symptoms. Waterborne cases of _Giardia lamblia_ have increased in the United States in recent years [1].

The eggs of several intestinal parasitic worms have been found in wastewater and have been shown to be a potential health problem to wastewater treatment plant operators and laborers employed on farms in India and East Germany where untreated wastewater is used for irrigation [6]. Modern water treatment methods have proved a very effective barrier against the waterborne spread of disease caused by protozoa and parasitic worms in developed countries, like the United States.

D.3.4 Concentrations of Pathogens in Wastewater

Evaluation of the relative risk of disease transmission associated with land application of wastewater requires knowledge of the number of pathogens in untreated and treated wastewater, as well as the number necessary to cause an infection in man or other animals. Unfortunately, data on the removal efficiency of all wastewater treatment methods for many pathogens are either nonexistent or largely based on laboratory studies by researchers who may overestimate the efficiency that can be obtained in actual practice [7]. From currently available information, Foster and Engelbrecht attempted to estimate the relative concentrations of pathogens in untreated wastewater and the relative efficiency of removal by primary and secondary treatment [7]. The results are shown in Table D-1.

TABLE D-1

ESTIMATED CONCENTRATIONS OF WASTEWATER PATHOGENS[a]

	Number of organisms/gal (3.78 L)			
Pathogen	Untreated wastewater	Primary effluent	Secondary effluent	Disinfection[b]
Salmonella	2.0×10^4	1.0×10^4	5.0×10^2	5×10^{-1}
E. histolytica	1.5×10^1	1.3×10^1	1.2×10^1	1.2×10^{-2}
Helminth ova	2.5×10^2	2.5×10^1	5.0×10^0	5×10^{-3}
Mycobacterium	2.0×10^2	1.0×10^2	1.5×10^1	1.5×10^{-2}
Human enterovirus (poliovirus, etc.)	4.0×10^4[c]	2.0×10^4	2.0×10^3	2×10^2

a. Adapted from Foster and Engelbrecht [7].

b. Conditions sufficient to yield a 99.9% kill.

c. As high as 4×10^6 per gal (3.78 L) have been reported [8].

Disinfection of wastewater, as commonly practiced today, is highly effective in achieving large reductions of bacterial pathogens, but it is much less effective against cysts and enteric viruses. For example, whereas a chlorine dose of 2 mg/L killed 99.9% of the coliform bacteria in 60 minutes in wastewater, 20 mg/L were required to achieve the same kill of poliovirus [9]. Another complicating factor in carrying over laboratory data obtained with pure viruses to wastewater is that polioviruses and other viruses are much more resistant to chlorine if organic matter is present [10]. For these and other reasons, chlorination as practiced today cannot be relied on alone to provide complete destruction of pathogenic bacteria and viruses.

The infective dose (the number of organisms necessary to cause disease in healthy humans or animals, some of whom may have had previous exposure), should also be considered when evaluating the disease potential. Infective doses for most bacterial and protozoan pathogens are relatively high. For instance, ingestion of 10^8 enteropathogenic *E. Coli* or *V. cholerae*, 10^4 to 10^9 *Salmonella*, and 10^1 to 10^2 *Shigella* organisms are necessary to cause infection in man [9]. The infective dose of a protozoan, such as *E. histolytica*, is believed to be as high as 20 cysts [7]. The infective dose for viruses varies with the type of virus and may range from 1 to 10^2 or more [11, 12]. The low infective dose of viruses gives importance to even relatively low concentrations of these agents in water.

D.3.5 Bacteriological and Virological Criteria for Wastewater Reuse

Additional research is necessary before guidelines based on specific data can be established concerning wastewater reuse for agricultural, recreational, and potable purposes. Bacteriological standards now exist for each type of reuse, but the relationship of these bacteriological standards to health risks from viral and other waterborne pathogens is arbitrary in all cases. Still, these standards should be considered when judging the effectiveness of land treatment for pathogen reduction and the effect on surface and subsurface water supplies.

The National Technical Advisory Committee on Water Quality has recommended that, for waters intended for agricultural use, the monthly average coliform bacteria counts should not exceed 5 000/100 mL and the fecal coliform concentration should be less than 1 000/100 mL [13]. According to a World Health Organization report on the reuse of wastewater effluents, only a limited health risk would result from unrestricted irrigation of agricultural crops if there were less than 100 coliforms/100 mL [14]. The National Technical Advisory Committee has recommended total coliform limits of less than 1 000/100 mL for recreational waters and 1/100 mL for drinking water [13].

Because enteric viruses have greater resistance to environmental stresses than bacteria, it has been suggested that bacterial standards may not give a realistic indication of the viral disease risk [5].

D.4 Bacterial Survival

To control the dissemination of pathogens among man and animals following land application of wastewater, it is imperative to know their persistence and movement in soil, overland runoff, groundwater, crops, and aerosols. The degree of retention by soil and the survival of pathogens therein will ascertain the chance of pathogen transfer to a susceptible host.

D.4.1 Bacterial Survival in Soil

The literature is replete with studies on bacterial survival in soil, and several reviews are available [2, 9, 15-17]. Among various pathogens of man and animals, survival in soil of *Brucella*, *Leptospira*, *Pseudomonas*, *E. coli*, *Erysipelothrix*, *Streptococci*, *Mycobacterium tuberculosis*, and *M. avium* have been investigated, and *Salmonella* in particular has been studied extensively. Survival of *S. typhi* was studied as early as 1889 when it was found to survive in soil for 3 and 5 months in two separate studies [18]. Survival times reported generally represent the maximum period after dosing the soil that a live organism could still be found.

Bacteria may survive in soil for a period varying from a few hours to several months, depending on the type of organism, type of soil, moisture-retaining capacity of soil, moisture and organic content of soil, pH, temperature, sunlight, rain, degree of contamination of wastewater being applied, and predation and antagonism from the resident microbial flora of soil. In general, enteric bacteria persist in soil for 2 to 3 months, although survival times as long as 5 years have been reported [15]. Under certain favorable conditions, applied organisms may actually multiply and increase in numbers. In general, however, land treatment using intermittent application and drying periods results in die-off of enteric bacteria retained in the soil.

Vegetative bacteria tend to die exponentially with time outside their host. The time of survival of these organisms therefore depends on the initial numbers applied as well as on the sensitivity of analysis and the size of the sample, not just on the adversity of the environment.

The influence of soil type on bacterial survival is important insofar as its moisture content, moisture-retaining capacity, pH, and organic matter content are concerned. It has been found by many workers that the survival of *E. coli*, *S. typhi*, and *M. avium* is greatly enhanced in moist rather than in dry soil. Survival time is less in sandy soil than in soils with greater water-holding capacity, such as moist loam and muck. Bacteria survive for a shorter time in strongly acid peat soil (pH 2.9 to 3.7) than in limestone-derived soil (pH 5.8 to 7.7). Increasing the pH of peat soil resulted in extended survival of enterococci. Bacterial persistence is related to the effect soil pH has on the availability of nutrients or the inhibitory agents present in the soil.

An increase in the longevity of bacteria in soil is often associated with increased organic content of the soil. Tannock and Smith demonstrated that populations of *Salmonella* declined rapidly when they were applied to pastures with wastewater containing no fecal matter, as compared to wastewater contaminated with feces [19]. Under natural conditions, the buildup of organisms may be greater in soils with high moisture and high organic content.

Both pathogens and indicators survive longer under low winter temperatures than in summer. In one study, it was reported that *S. typhi* survived as long as 2 years at constant freezing temperatures. In fact, the self-cleansing property of soil is slowed down in the Russian Arctic where winters are prolonged [20]. Microorganisms disappear more rapidly at the soil surface than below the surface, apparently because of desiccation, effects of sunlight, and other factors at work at the soil surface.

Another important factor is the competition and antagonism the alien enteric bacteria face from the resident soil microflora. Thus, organisms applied to sterilized soil survive longer than they would in unsterilized soil. Factors that influence the survival of bacteria in soil are listed in Table D-2.

D.4.2 Bacterial Survival in Groundwater

Pathogenic organisms generally are removed rapidly in most soils, but they may pass through coarse materials and fractured rocks like limestone. Only limited information is available on the survival of bacteria in groundwater, and there is wide variation in the reported duration of bacterial viability in underground waters. It should be noted, however, that pathogens are expected to survive longer in groundwater than on the soil surface because of low temperature, nearly neutral pH, absence of sunlight, and absence of antagonistic bacteria. From the few studies that have been made, it appears that bacteria may persist in underground water for months. *E. coli* have been found to survive up to 1 000 days in subsoil water, whereas a 50% reduction in number occurred within 12 hours in well water [16].

TABLE D-2

FACTORS THAT AFFECT THE SURVIVAL OF ENTERIC BACTERIA AND VIRUSES IN SOIL

Factor		Remarks
pH	Bacteria	Shorter survival in acid soils (pH 3 to 5) than in neutral and alkaline soils
	Viruses	Insufficient data
Antagonism from soil microflora	Bacteria	Increased survival time in sterile soil
	Viruses	Insufficient data
Moisture content	Bacteria and viruses	Longer survival in moist soils and during periods of high rainfall
Temperature	Bacteria and viruses	Longer survival at low (winter) temperatures
Sunlight	Bacteria and viruses	Shorter survival at the soil surface
Organic matter	Bacteria and viruses	Longer survival (regrowth of some bacteria when sufficient amounts of organic matter are present)

D.4.3 Bacterial Survival on Crops

At the turn of the century, cases of typhoid fever attributed to the eating of raw vegetables grown on soil fertilized with untreated wastewater [2] led to extensive studies on the survival of enteric bacteria on such crops. Some important facts emerged from these studies:

1. The surfaces of fruits and vegetables growing in soil irrigated with raw wastewater can be contaminated with bacteria that are not easily removed by ordinary washing [2]. Furthermore, bacteria can penetrate broken, bruised, and damaged portions of vegetables, but not the healthy surfaces.

2. Crops grown on fields may become contaminated directly during irrigation with wastewater and indirectly through contact with polluted soil or field workers. Kruse reported that heavy rainfall did not wash away coliforms from clover that was irrigated with settled wastewater [21].

3. Bacteria survive longer in dense grass than in sparse grass, and longer in leafy vegetables than in smooth vegetables, apparently because of protection from the lethal effect of sunlight. Low temperature and adequate moisture also favor bacterial survival [2].

4. Although the length of survival depends on several factors, including weather, type of vegetable, and type of organism present, a period of 30 to 40 days is most common [22].

D.5 Virus Survival

Although little information is currently available, studies are underway on the survival of enteric viruses in soil, on crops, in groundwater, or in aerosols.

D.5.1 Virus Survival in Soil

Laboratory studies indicate that virus survival in soil depends on the nature of the soil, temperature, pH, moisture, and possibly antagonism by soil microflora. Viruses readily adsorb to soil particles, and this has been reported to prolong their survival time in aqueous marine environments [23]. Such viruses bound to solids are as infectious to man and animals as the free viruses by themselves [24].

In studies on the survival of f2 bacteriophage and poliovirus type 1 in sand saturated with tapwater and oxidation pond effluent, it was observed that 60 to 90% of the viruses was inactivated at 20°C within 7 days [25]. After this initial large kill, the viruses became inactivated at a much slower rate and polioviruses could still be detected at 91 days. The f2 viruses survived longer than 175 days. At lower temperatures, as many as 20% of the polioviruses survived after 175 days.

Considerable stability and prolonged survival of several enteric viruses in loamy and sandy loam soils in the Soviet Union have been reported [16]. Virus survival was found to vary from 15 to 170 days, depending on various environmental factors and the type of virus. The degree of soil moisture had a marked influence on the survival time of the viruses. In air-dried soils, the viruses survived only 15 to 25 days, but in soil that had a 10% moisture content, they survived up to 90 days.

Sullivan et al. have studied the survival of poliovirus type 1 in outdoor soil plots irrigated with wastewater sludge and effluent during the spring and winter in Ohio [26]. During the winter, some viruses survived 96 days in sludge-irrigated soil and 89 days in effluent-irrigated soil. During the spring, viruses survived less than 16 days in both sludge- and effluent-irrigated soils. The higher temperature and solar radiation levels in spring apparently accelerated viral die-off.

In field experiments in Hawaii, seeded poliovirus type 1 at very high concentrations was found to survive at least 32 days at the soil surface that had been sodded and irrigated with wastewater effluents [27].

From these studies it appears that viruses survive for times as short as 7 days or as long as 6 months in soil, and that climatic conditions, particularly temperature, have a major influence on survival time. Some of the factors that influence the survival of viruses in soils are listed in Table D-2.

D.5.2 Virus Survival in Groundwater

Enteric viruses can also survive for long periods of time in water. A survey of the literature indicates that enteric viruses can survive from 2 to more than 188 days in fresh water [28], but little information on their survival in groundwater is available. Again, temperature is the most important factor in virus survival in water; survival is greatly prolonged at lower temperatures. In studying a land treatment site in Florida where wastewater was being applied to a cypress dome, Wellings et al. were able to detect enteric viruses in monitoring wells 28 days after the last application of wastewater to the surface [29]. The wells were 10 ft (3 m) deep and the lateral distance was 23 ft (7 m). It should be noted that periods of heavy rainfall preceded virus detection.

D.5.3 Virus Survival on Crops

Generally speaking, virus survival on crops under field conditions can be expected to be shorter than in soil, because the viruses are more exposed to deleterious environmental effects. Artificially seeded viruses have been shown to contaminate vegetables and forage crops during sprinkler irrigation with wastewater [30], although this is undoubtedly a function of irrigation practices. The most common type of contamination occurs when wastewater comes in contact with the surface of the crop. There is also evidence that, in rare events, the translocation of animal viruses from the roots of plants to the aerial parts can occur [31]. However, in general, the pathogens associated with municipal wastewaters do not enter the plant substance. A number of factors, such as sunlight, temperature, humidity, and rainfall, are known to affect virus persistence on vegetation. There is also evidence that virus survival varies with the type of crop.

Larkin et al. studied the persistence of artificially seeded poliovirus type 1 after sprinkler irrigation of wastewater onto lettuce and radishes during two growing seasons in Ohio [30]. The viruses survived on these vegetables from 14 to 36 days after irrigation under field conditions, although a 99% loss in detectable viruses was noted during the first 5 to 6 days. In Israel, the surfaces of tomatoes and parsley were contaminated with oxidation pond effluent containing polioviruses and then exposed to sunlight [32]. No viruses were detectable after 72 hours on the surface of the vegetables. Sunlight was believed to be a major factor because massive inactivation of viruses occurred when the solar energy exceeded 0.35 $cal/cm^2 \cdot min$. Thus, virus survival is probably minimal on the parts of the plants that receive direct sunlight, but prolonged survival could be expected on the moist, protected parts of plants. Animal viruses readily adsorb to plant roots, and some investigators have reported that viruses apparently penetrate the surfaces of roots, resulting in internal contamination of the plant [33]. No information is currently available on the survival of animal viruses within the edible parts of plants.

It also should be pointed out that once the crops are harvested, viruses can survive for prolonged periods of time during commercial and household storage at low temperatures. For example, polioviruses and coxsackieviruses artificially applied on the surfaces of vegetables have survived for more than 4 months in a refrigerator [34].

D.6 Movement and Retention of Bacteria in Soil

Once pathogenic bacteria present in wastewater are applied to the land, it is necessary to know to what extent they are retained by the soil. This is important in order to determine if, and to what extent, they are capable of contaminating groundwater.

D.6.1 Laboratory Studies

Pathogen removal is a function of characteristics of the soil, such as particle size, particle shape, and surface properties, as well as aggregation and packing of soil particles. Most bacteria appear to be removed after brief passage through heavy-textured clay soils and consolidated sands as a result of filtration and adsorption. In contrast, bacteria can travel longer distances through highly fractured rock, such as limestones or basalts.

Wastewater bacteria are effectively removed by percolation through a few feet of fine soil by the process of straining at the soil surface, and at intergrain contacts, sedimentation, and sorption by soil particles. Adsorption of bacteria to sand depends on pH and zeta potential of the soil and is reversible. Factors that reduce the repulsive forces between the two surfaces, such as the presence of cations, would be expected to allow closer interaction between them and allow adsorption to proceed.

Adsorption plays a more important role in the removal of microorganisms in soils containing clay because the very small size of clays, their generally platy shapes, the occurrence of a large surface area per given volume, and the substitution of lower valence metal atoms in their crystal lattices make them ideal adsorption sites for bacteria in soils [35].

As a result of mechanical and biological straining, and the accumulation of wastewater solids and bacterial slimes, an organic mat is formed in the top 0.2 in. (0.5 cm) of soil. This mat is capable of removing even finer particles by bridging or sedimentation before they reach and clog the original soil surface. Butler et al. observed the greatest removal of bacteria on the mat that formed on the soil surface, followed by a subsequent buildup of bacteria at lower levels [36]. Their results indicated that a limiting zone is slowly built up in the soil and that its depth below the surface depends on the nature of the liquid applied and the surface treatment of the soil. Under various operating conditions studied, this zone occurred at 3.9 to 19.5 in. (10 to 50 cm) below the soil surface and was not related to the particle size of the soils studied.

Other complex and interlocking factors determine the distance of travel. Generalizations are difficult, but movement is related directly to the hydraulic infiltration rate and inversely to the particle size of the soil and to the concentration and cationic composition of the solute. Retention and subsequent survival also depend on the rate of groundwater flow, oxygen tension, temperature, and availability of food.

It is apparent from the foregoing discussion that the upper layers of the soil are most efficient for removing microorganisms. Once these organisms are retained, the primary consideration is the length of their survival in the soil matrix, where they are inactivated following exposure to sunlight, oxidation, desiccation, and antagonism from the soil microbial population.

D.6.2 Field Studies

The first major field studies on bacteria removal during wastewater percolation through soil were performed at Whittier and Azusa, California. At Whittier, coliform concentrations were reduced from 110 000/100 mL to 40 000/100 mL after percolation through 3 ft (0.9 m) of soil in 12 days, and none appeared at greater depths. When treated wastewater effluent containing 120 000 organisms/100 mL was allowed to percolate in Azusa soil, the percolates produced at 2.5 and 7 ft (0.75 and 2.1 m) contained 60 organisms/100 mL [17]. At Lodi, California, coliform levels were observed to decrease below drinking water standards within 7 ft (2.1 m) of the surface when undisinfected wastewater effluent was applied to sandy loam soil, but in one case, coliforms were detected at a depth of 13 ft (3.9 m) [16].

In a thorough study at the Santee Project near San Diego, California, it was found that most of the bacteria removal occurred within the first 200 ft (60 m) of horizontal travel, with little additional removal occurring in the next 1 300 ft (390 m). The median value of fecal streptococci in the oxidation pond effluent was 4 500/100 mL, while median values from wells at 200 ft (60 m), 400 ft (120 m), and 1 500 ft (450 m) were 20, 48, and 6.8/100 mL, respectively. The medium consisted of coarse gravel and sand confined in a river bed.

At the Flushing Meadows Project near Phoenix, Arizona, wastewater (with 10^5 to 10^6 coliforms/100 mL) was applied to infiltration basins that consisted of 3 ft (0.9 m) of fine loamy sand underlain by a succession of coarse sand and gravel layers to a depth of 250 ft (75 m). With a wastewater infiltration rate of 330 ft/yr (99 m/yr), the total coliforms decreased to a level of 0 to 200 organisms/100 mL at 30 ft (9 m) from the point of application when basins were inundated for 2 weeks followed by a dry period of 3 weeks. When 2- to 3-day inundation periods were used, however, the total coliform levels were reduced to 5/100 mL, a reduction of 99.9% [16].

D.6.3 Potential for Groundwater Contamination

It is generally believed that percolation through a porous medium, such as 5 to 10 ft (1.5 to 3 m) of continuous fine soil, removes most bacteria. This removal, however, has its own limitations. Different soils have different capacities to remove bacteria. While pathogens may be removed rapidly in most soils, they may reach groundwater in regions where subsurface fissures are common. Adequate site investigation would show the presence of areas with fissured subsurface geology.

Although land treatment systems have never been implicated as a cause of diseases due to contaminated groundwater, it would seem prudent to maintain some type of surveillance in high-risk areas to establish travel of pathogens through the soil. It should be noted that bacteria do not travel significant distances in all directions from a concentrated source, but are carried only with the groundwater flow.

D.7 Movement and Retention of Viruses in Soil

Unlike bacteria, where filtration at the soil-water interface appears to be the main factor in limiting movement through the soil, adsorption is probably the predominant factor in virus removal by soil. Thus, factors influencing adsorption phenomena will determine not only the efficiency of short-term virus retention but also the long-term behavior of viruses in the soil. Viruses are composed of a nucleic acid core encased in a protein coat, and thus mimic the colloidal characteristics of proteins. It has been shown that adsorption of such hydrophilic colloids is strongly influenced by the pH of the media, the presence of cations, and the ionizable groups on the virus [37].

The pH is of considerable importance relative to adsorption. At the pH at which the isoelectric point of the virus occurs, the net electric charge is zero. The virus has a positive charge below the isoelectric point, and a negative charge above the isoelectric point. Viruses are strongly negatively charged at high pH levels and strongly positively charged at low pH levels. The isoelectric pH for enteric viruses is usually below pH 5; thus, in the pH range of most soils, enteroviruses as well as soil particles retain a net negative charge. In general, virus adsorption to surfaces is enhanced at a pH below 7 and reduced at a pH above 7 [38]. It is important to note that viruses once adsorbed to solids at a low pH are readily desorbed by a rise in pH.

While the actual mechanism of viral adsorption to solids is not known, two general theories have been proposed. Both are based on the net electronegativity of the interacting particles. Carlson et al. found that in solution bacteriophage T2 adsorption to common clay particles was highly dependent on the concentration and type of cation present [39]. It was shown that maximum adsorption of T2 was about 10 times greater for a divalent cation than a monovalent cation at the same concentration in solution. In addition, no definite relationship between the degree of virus adsorption to clay particle and electrophoretic mobility was evident. This led Carlson et al. to conclude that a clay-cation-virus bridge was operating to link the two negatively charged particles. Thus, a reduction in cation concentration results in a breakdown of the bridging effect and desorption of the viruses. They also demonstrated that organic matter in solution competed with viruses for adsorption sites, resulting in decreased virus adsorption or elution of adsorbed viruses from the clay.

From the foregoing analysis, it can be concluded that virus adsorption cannot be considered a process of absolute immobilization of the viruses from the liquid phase. Any process that results in a breakdown of virus association with solids will result in their further movement through porous media.

D.7.1 Laboratory Studies

Laboratory soil column studies on virus removal have demonstrated that most of the viruses in wastewater are removed in the top few centimetres, but work has been limited to only a few soil types, and broad generalizations on virus removal cannot be made at present. Presently, no existing models are available to quantify virus behavior, but with additional research on the mechanisms of virus adsorption to soils, predictive models on virus removal efficiency may be determined for land treatment sites.

Drewry and Eliassen, who performed some of the earliest work on virus movement through soil, conducted experiments with bacteriophages and nine different soils from California and Arkansas [40]. Batch tests indicated that virus adsorption in distilled water showed typical Freundlich isotherms, indicating that physical adsorption was taking place. The effect of the pH of the soil-liquid slurry on virus adsorption for five California soils is shown in Figure D-1. Virus adsorption was found to decrease at pH values above 7 because of increased ionization of the carboxyl groups of the virus protein and increasing negative charge on the soil particles. In most soils tested, virus adsorption increased with increasing cation concentration, but in some soils, no effect was observed. Other batch studies indicated that, in general, virus adsorption by soil increased with increasing ion exchange capacity, clay content, organic carbon, and glycerol-retention capacity, but exceptions were found with at least one soil type. In studies in which viruses suspended in distilled water were passed through columns of 16 to 20 in. (40 to 50 cm) of sterile soil, over 99% removal of the viruses was observed. Radioactivity tagging experiments indicated that most of the viruses were retained in the top 0.8 in. (2 cm) of the column.

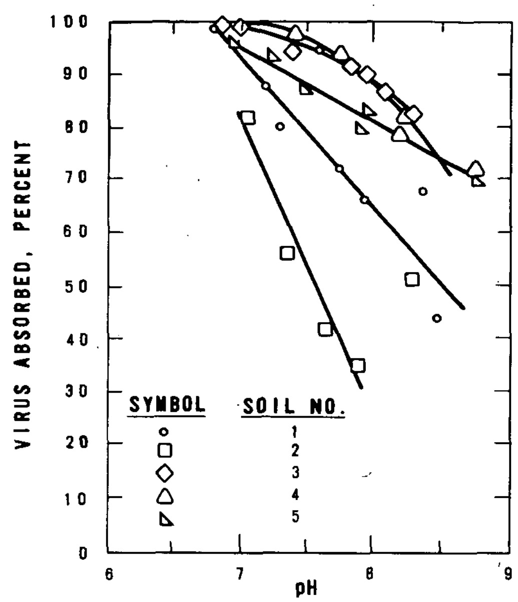
This pattern of virus removal has been found to be similar for both bacteriophages and animal viruses, although in some soils bacteriophages appear to be removed more efficiently.

Laboratory studies also indicate that rainfall can have a dramatic effect on the migration of viruses through soil [41]. Alternating cycles of rainfall and effluent application result in ionic gradients that enhance the movement of virus. Rainfall reduces the ionic concentration of salts in the soil after wastewater application. Such changes in ionic strength have been found to be closely linked with the elution of viruses near the soil surface [41]. This is seen as a burst of released viruses in soil columns when the specific conductance of the water in the soil column begins to decrease after the application of rainwater (simulated in the laboratory by the use of distilled water). This same elution effect can also be seen if a rise occurs in the pH of the water applied to the surface of the soil; that is, a rise in pH from

7.2 to 8 or 9 results in the elution of viruses adsorbed to soil [41]. Viruses are also capable of elution even after remaining in columns saturated with wastewater for long periods of time.

FIGURE D-1

VIRUS ADSORPTION BY VARIOUS SOILS AS A FUNCTION OF pH [40]



Studies by Lance et al. have indicated that certain management practices may prove useful in limiting virus migration through soil [42]. Using 98 in. (250 cm) columns of sandy loam soil, they found that many of the viruses eluting near the soil surface after addition of 4 in. (10 cm) of distilled water were later adsorbed near the bottom of the column and that migration of the viruses could be minimized if the columns were flooded with wastewater shortly after the simulated rainfall. In addition, allowing the columns to drain (i.e., soil not saturated with effluent) for at least 5 days before application of the distilled water resulted in no apparent virus movement through the soil. This led the authors to suggest that if a heavy rainfall occurred at a land treatment site within 5 days after application of wastewater, the area could be reflooded with wastewater to restrict subsurface virus migration through the soil.

These same authors also found that flooding of soil columns continuously for 27 days with wastewater seeded with approximately 30 000 infectious units (plaque-forming units) of poliovirus/mL did not saturate the adsorption capacity of the top few centimetres of soil. Removal of viruses below the 2 in. (5 cm) depth could be expressed by the following equation:

$$\frac{dC_v}{dz} = -kC_v \qquad \text{(D-1)}$$

where C_v = virus concentration detected at any depth in the column below 2 in. (5 cm), PFU/mL

z = column depth, cm

k = removal constant

In the sandy loam soil used in these experiments, k was found to be equal to 0.046 cm^{-1}.

D.7.2 Field Studies

Field studies on virus travel through soil have been hampered until recently by the lack of techniques necessary for the concentration of viruses from large volumes of water. This was the main limitation of the few early studies, such as the one at the Santee Project, on virus movement in groundwater.

At the Santee Project near San Diego, California, attempts were made to isolate viruses, using swab techniques, from observation wells located 200 to 400 ft (60 and 120 m) from a wastewater infiltration site [16]. Viruses were never isolated from the wells, even after larger amounts of vaccine strain polioviruses were seeded into the wastewater percolation beds.

Viruses were studied at Whittier Narrows, California, during the time of the Sabin polio vaccine program [17]. The one to four litre collected samples showed concentrations of 102 to 252 plaque-forming units (PFU) per litre in the applied effluents, but no viruses were detected after passage through 2 ft (0.6 m) of soil. All plaques in the applied effluent were identified as a polio type III.

Recently, Wellings et al. reported on the travel of viruses through soil at a wastewater reclamation pilot project near St. Petersburg, Florida [43]. At a 10 acre (4 ha) site, chlorinated secondary effluent was applied by a sprinkler system at the rate of 2 to 11 in./wk (5 to 28 cm/wk). The soil consists of Immokalee sand with little or no silt or clay. On one side of the test plot, an underdrain of tiles was placed at a depth of 5 ft (1.5 m) on top of an organic aquitard. This subsurface drain directs the percolated waters through a weir where gauze pads were placed for the collection of viruses. Both polioviruses and echoviruses were isolated from the weir water, demonstrating that viruses must survive aeration and sunlight during spraying as well as percolation through 5 ft (1.5 m) of sandy soil. Viruses were also isolated in wells 10 and 20 ft (3 and 6 m) below the soil surface when 50 to 150 gal (189 to 567 L) of percolate were sampled. No viruses were detected in these wells for the first 5 months of the study. Only after two heavy rains was poliovirus type I isolated. The viruses were first detected in the 10 ft (3 m) well and some time later appeared in the 20 ft (6 m) well, indicating that viruses were migrating though the soil. The authors reasoned that the high rainfall resulted in a large increase in the soil-water ratio, which led to increased solubility of portions of the organic layer and thus desorption of attached viruses. The observation of viruses as a "burst" after the rainfall was cited as evidence that the rainfall was responsible for the presence of viruses in the wells.

This same group of investigators also reported on the detection of viruses in groundwater after the discharge of secondary effluent into a cypress dome in Florida [29]. The soil under the dome consisted of black muck and layers of sand and clay. Viruses were isolated from wells 10 ft (3 m) deep, again after periods of heavy rainfall. The viruses had traveled laterally 23 ft (7 m) in the subsurface to reach the observation wells. Another important observation made during this study that is not generally recognized is the failure to detect fecal coliform bacteria in the well samples found to contain viruses.

A virus study was conducted at a rapid infiltration site at Fort Devens, Massachusetts, where primary effluent was applied. Very high concentrations of viruses were added to the effluent. Virus travel through the very coarse sand and gravel was observed [44].

In contrast to these findings, field studies at the Flushing Meadows rapid infiltration project near Phoenix, Arizona, indicate limited virus movement through the soil [45]. At this site, basins in loamy sand are underlain at a 3 ft (0.9 m) depth by coarse sand and gravel and are intermittently flooded with secondary effluent at an average hydraulic loading rate of 300 ft/yr (90 m/yr). Although viruses were detected in the wastewater used to flood the basins, no viruses were detected in wells 20 ft (6 m) deep, located midway between the basins. These results indicated that at least a 99.99% removal of viruses had occurred during travel of secondary treated wastewater through 30 ft (9 m) of sandy soil--20 ft (6 m) vertically and 10 ft (3 m) laterally. The loamy sand at this site may have resulted in better conditions for virus removal than at other land treatment sites studied to date (see foregoing discussion of work by Lance et al. [42]).

D.7.3 Potential for Groundwater Contamination

The results of recent field and laboratory studies reviewed in the previous sections of this appendix indicate that, under certain conditions, enteric viruses can gain entrance into groundwater. The type of land treatment system, climatic conditions, soil type, and possibly management practices of soil flooding, appear to be the dominant factors in controlling virus migration through soil. The greatest danger to groundwater appears to be in areas that receive high periodic rainfalls, which allow adsorbed viruses to be eluted as a "burst" or a wave of infectious particles. Limited research results indicate that flooding sites with wastewater after a rainfall may limit virus removal, but much more work needs to be done in this area before such practices can be recommended. Some factors that should be considered when evaluating a site for the potential of groundwater contamination by viruses are shown in Table D-3.

TABLE D-3

FACTORS THAT INFLUENCE THE MOVEMENT OF VIRUSES IN SOIL

Factor	Remarks
Rainfall	Viruses retained near the soil surface may be eluted after a heavy rainfall because of the establishment of ionic gradients within the soil column.
pH	Low pH favors virus adsorption; high pH results in elution of adsorbed virus.
Soil composition	Viruses are readily adsorbed to clays under appropriate conditions and the higher the clay content of the soil, the greater the expected removal of virus. Sandy loam soils and other soils containing organic matter also are favorable for virus removal. Soils with a low surface area do not achieve good virus removal.
Flowrate	As the flowrate increases, virus removal declines, but flowrates as high as 32 ft/d (9.6 m/d) can result in 99.9% virus removal after travel through 8.2 ft (2.5 m) of sandy loam soil.
Soluble organics	Soluble organic matter competes with viruses for adsorption sites on the soil particles, resulting in decreased virus adsorption or even elution of an already adsorbed virus. Definitive information is still lacking for soil systems.
Cations	The presence of cations usually enhances the retention of viruses by soil.

D.8 Potential Disease Transmission Through Crop Irrigation

The type of crop (vegetation) and irrigation practice determines the extent of crop contamination and plays a significant role in the evaluation of health risks following land treatment. Wastewater irrigation of fodder and fiber crops presents the least health risk; while irrigation of food crops, particularly, those eaten raw, poses the greatest potential risk. For clarification, the term crop is used to include all vegetation that forms an integral part of the waste treatment system. This includes grain, seed, fodder, and fiber crops. Sprinkling and flooding wet the low-growing vegetation as well as the soil, but direct contact between the wastewater and the vegetation is avoided by the use of subsurface irrigation or the flooding techniques. The greatest health concern is with low-growing crops, such as vegetables, which have a greater chance of contamination and are often eaten raw. Contamination of orchard or other crops whose edible portion does not come into contact with the soil or wastewater during irrigation would be expected to be small.

Although bacteria do not enter healthy and unbroken surfaces of vegetables, they can penetrate broken, bruised, and unhealthy plants and vegetables. Once vegetables are contaminated, they are not easily decontaminated by rinsing with water or disinfectant. Therefore, it appears that a greater risk is associated with truck and garden crops grown with wastewater and eaten raw than with vegetables eaten only after cooking or processing.

D.8.1 Limitations on Crop Use

Different standards have been put forward regulating the use of land treatment of wastewater. The states of California and Arizona were among the first to promulgate such standards. Arbitrary waiting periods are sometimes imposed on the use of crops grown on treated land. The reviews by Rudolfs et al. [15], Krishnaswami [46], and Geldreich and Bordner [47] are interesting in this regard. Some limitations on crop use put forth by these authors and others are summarized as follows:

1. Crops that are eaten after they are cooked, or industrial crops that are eaten after satisfactory processing, may be irrigated with treated wastewater.

2. Oxidized and disinfected wastewater effluent may be used to irrigate fruit and vegetable crops. Vegetables should not be sprinkler irrigated for 4 weeks prior to harvest. Similarly, application on pasture and hay should stop 2 weeks before pasturing or harvesting. (This also provides a drying period for farm equipment access.)

3. Reclaimed water used for the surface or spray irrigation of fodder, fiber, and seed crops shall have a level of quality no less than that of primary effluent.

D.8.2 Risks to Grazing Animals

Arbitrary preapplication treatment limitations are usually imposed on wastewater-irrigated land to preserve aesthetics, to minimize health risks, and to protect crops meant for human consumption, but it is equally undesirable to infect animals. A number of cases of disease in animals have been attributed to their unintentional exposure to wastewater, but relatively less is known about the risks to animals grazing on pastures irrigated with wastewater. The use of untreated wastewater for the irrigation of grazing land has been practiced on a large scale in Europe and Australia. The use of treated wastewater for the irrigation of grazing lands has also been practiced in the United States (see Chapter 7, Sections 7.2 and 7.5), for many years, with seemingly little threat to the health of farm animals under normal conditions. However, the transmission of disease to domestic animals from wastewater-contaminated water and pasture has been known to occur [48-50], and carefully controlled experiments and field data need to be compiled to develop effective guidelines.

Whether or not animals grazing on a wastewater irrigated pasture will become infected may depend on many factors, including persistence and concentration of pathogens, the health of the animals, and the interval between irrigation and grazing. Preventing grazing on pastures immediately after flooding with wastewater will allow time for significant reduction in the levels of any pathogens applied. Most pathogenic bacteria and viruses are quickly inactivated during desiccation and when exposed to sunlight.

In cases of salmonellosis in a dairy herd, the source of infection was found to be rye contaminated with domestic wastewater effluent overflowing onto grazing land. In this study, Bicknell isolated _S. aberdeen_ from 22 cows, wastewater, materials inside the wastewater pipeline, pond mud, a cess pit, and dung in the farm yard [55]. Nottingham and Urselmann found _S. typhimurium_ in pasture soil at a farm in New Zealand where acute salmonellosis had occurred during the preceding 9 months [48].

Risks of infection among animals are not limited to _Salmonella_. _Pseudomonas aeruginosa_, _Mycobacterium tuberculosis_, and _Leptospira_ organisms may exist in waste applied to pasture and may present a risk to the health of dairy cows and calves, but no documented evidence exists at present to indicate that a risk exists. Calves that grazed pastures to which 10^6 _S. dublin_ organisms/mL of slurry had been applied on the previous day became infected, but no infections resulted when the contamination rate was decreased to 10^3 organisms/mL of slurry [51]. These limited results indicate that Salmonella may only be a concern in unusual circumstances when high concentrations of these organisms are present.

D.9 Potential Disease Transmission By Aerosols

Aerosols containing bacterial and viral pathogens may be infectious on inhalation. During sprinkler irrigation of wastewater, approximately 0.1% of the liquid is aerosolized [52]. Thus, there is a possibility of producing a potential health risk by the process [53]. This feature, plus the limited amount of information available on the subject, makes the evaluation of health implications of aerosols from any form of wastewater treatment difficult to assess.

Aerosols have been defined as particles in the size range of 0.01 to 50 μm that are suspended in air. When an airborne water droplet is created, the water evaporates very rapidly under average atmospheric conditions, resulting in a nucleus of the originally dissolved solids plus the microorganisms contained in the original droplet [54]. The high rate of evaporation results in the die-off of many of the original organisms that were aerosolized, but the remaining resistant organisms may persist for a long time.

Humans may be infected by biological aerosols primarily by inhaling the aerosol or secondarily by contacting material on which the airborne droplets have settled (i.e., clothes). The infectivity of an aerosol depends on the depth of respiratory penetration and the presence of pathogenic organisms. Larger droplets (2 to 5 μm) are mainly removed in the upper respiratory tract and do not gain entrance to the alveoli of the lungs, although they may find their way into the digestive tract because of the ciliary action [54]. Thus, if gastrointestinal pathogens are present, infection may result. However, a much higher rate of infection occurs when respiratory pathogens are inhaled in smaller droplets (about 0.2 to 2 μm) that do reach the alveoli of the lungs [54]. Also important is the fact that some pathogens found in wastewater have a lower infective dose (i.e, number of organisms necessary to cause an infection) in aerosol form than when ingested directly [9].

Most of the information available today on wastewater aerosols concerns their generation by wastewater treatment facilities, such as activated sludge treatment plants. Aeration of the wastewater during this process has been shown to produce biological aerosols that can be carried considerable distances dependent on local climatic conditions. Airborne coliform bacteria have been recovered at night as far as 0.8 m (1.3 km) from a large trickling filter plant [52]. Factors that have been found to affect the survival and dispersion of bacteria and viruses in such aerosols are summarized in Table D-4.

TABLE D-4

FACTORS THAT AFFECT THE SURVIVAL AND DISPERSION OF BACTERIA AND VIRUSES IN WASTEWATER AEROSOLS

Factor	Remarks
Relative humidity	Bacteria and most enteric viruses survive longer at high relative humidities, such as those occurring during the night. High relative humidity delays droplet evaporation and retards organism die-off.
Wind speed	Low wind speeds reduce biological aerosol transmission.
Sunlight	Sunlight, through ultraviolet radiation, is deleterious to microorganisms. The greatest concentration of organisms in aerosols from wastewater occurs at night.
Temperature	Increased temperature can also reduce the viability of organisms in aerosols mainly by accentuating the effects of relative humidity. Pronounced temperature effects do not appear until a temperature of 80°F (26.7°C) is reached.
Open air	It has been observed that bacteria and viruses are inactivated more rapidly when aerosolized and when the captive aerosols are exposed to the open air than when held in the laboratory. Much more work is needed to clarify this issue.

There is little quantitative information on the spread of biological aerosols from land application of wastewater by sprinkler irrigation [52]:

> In 1957, Merz investigated the hazards associated with sprinkling treated wastewater onto a golf course. Air was sampled downwind from a covered sedimentation basin, a wastewater aeration tank, and a sprinkler by using a sampling instrument with a rectangular orifice that impinged air onto the surface of liquid collection media. The sampler fluid was assayed for coliform organisms. Coliforms were reported to have been recovered only downwind from the sprinkler and close enough [135 ft (41.1 m)] that the spray could be felt. Merz concluded that hazards from sprinkling wastewater were limited to direct contact with unevaporated droplets. Merz's study (now out of print) is the only published U.S. field study that could be found that addressed airborne microorganisms from land application of wastewater although some foreign language articles and unpublished materials do address the

subject....Reploh and Handloser, by using agar settling plates, found airborne dispersion of coliform bacteria downwind from sprinklers discharging [untreated] wastewater....They estimated that the viable aerosol could be carried 400 m downwind by a 5 m/s wind and recommended that large land areas and the planting of hedges be used as safety measures.

Bringmann and Trolldenier, by using Endo agar settling plates, investigated the airborne spread of bacteria downwind from sprays discharging settled wastewater that was not disinfected. They found that the downwind travel distance of the viable aerosol increased as relative humidity and wind speed increased and ecreased as ultraviolet radiation increased. They estimated that coliform organisms may remain viable as far as 400 m downwind from the source under conditions of darkness, 100 percent relative humidity, and a wind speed of 7 m/sec. Sepp measured the airborne spread of total and coliform bacteria downwind from sprayers discharging ponded and chlorinated activated sludge tank effluent. Coliform bacteria were recovered as far as 10 ft (3.0 m) downwind from the spray limits in a dense brushy area and up to 200 ft (61 m) downwind from the spray limits in a sparsely vegetated area. Shtarkas and Krasil'shchikov recovered bacteria on settling plates 650 m downwind from sprinklers discharging settled wastewater and recommended a 1,000-m sanitary zone around such installations.

Katzenelson and Teltch have recently studied the bacterial aerosols generated by the sprinkler irrigation of water from a small stream contaminated by untreated domestic wastewater [55]. They used Anderson and glass impingers to collect coliform bacteria in air at distances up to 1 310 ft (400 m) from an irrigation line and 820 ft (250 m) from an aerated lagoon. The coliform concentration of 820 ft (250 m) from the sprinklers and lagoon were from 0 to 17 coliforms/m^3, and 0 to 4 coliforms/m^3, respectively. In addition, of the 45 colonies evaluated only one colony showing the characteristics of Salmonella infentis was isolated 197 ft (60 m) from the sprinklers. Bausum et al., using Anderson samplers and high-volume electrostatic precipitors, detected tracer bacterial viruses 2 067 ft (630 m) from the wetted zone at a sprinkler irrigation site [56].

The first epidemiological evidence of a disease risk associated with wastewater irrigation has been reported recently by Katzenelson et al. [57]. The incidence of enteric disease in agricultural communal settlements in Israel that practiced wastewater irrigation with partially treated, nondisinfected wastewater (similar to that of raw domestic wastewater), was compared with similar settlements that did not practice wastewater irrigation. The incidence of shigellosis, salmonellosis, typhoid fever, and infectious hepatitis was found to be 2 to 4 times higher in those communities practicing wastewater irrigation. No difference in the incidence of disease not transmitted by wastewater was observed between the communities, nor were differences observed for shigellosis and infectious hepatitis rates during the winter when irrigation with wastewater was not practiced. These authors claim [57]:

> These findings, although of a tentative nature, point out that the health hazards associated with wastewater irrigation may be greater than previously assumed. In the case of the kibbutzim studied the distance between the areas spray irrigated with wastewater and the residential areas vary from 100-3,000 meters. No direct evidence is available at this time as to the actual concentrations of pathogens in the air at the residential areas....It is also possible that the pathogens from the wastewater irrigation areas can reach the kibbutz population by an alternate pathway, on the bodies and clothes of the irrigation workers who live in the community and return from the fields at mealtime and at the end of the day.

The potential health effects related to the production of wastewater aerosols have yet to be fully established. The recent work of Katzenelson et al. indicates that the sprinkling of untreated wastewater may be a health risk to irrigation workers and possibly to persons residing nearby [57]. Biological treatment and disinfection may largely eliminate any possible pathogen transmission by aerosols, but validation of this is necessary. The use of buffer zones, control of sprinkling operations to minimize the production of fine droplets, elimination of sprinkling during high winds, and sprinkling only during daylight hours should be considered as alternative control measures in the production of biological aerosols.

D.10 References

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APPENDIX E

METALS

E.1 Introduction

An important consideration in modern treatment of wastewater is to produce an effluent that can be used on land without leading to significant problems either at once or later on. Nearly all wastewater delivered to treatment facilities contains metals or trace elements. Industrial plants are an obvious source; but wastewaters from private residences can have metal concentrations many times those of seawater, groundwater, or domestic waters. The important trace metals are copper, nickel, lead, cadmium, and zinc. Some wastewater influents have high concentrations of other trace elements that must be dealt with in a special manner.

A few of the metals in wastewater, being essential to life, may enrich a soil at a land treatment site. Zinc is the metal most likely to provide an environmental benefit, because large areas of land have too little zinc for the growth of some crops and because average dietary zinc intake by humans is marginal. Nevertheless, such essential-to-life metals (and others too) can accumulate and pose potential long-term hazards to plant growth or to animals or humans consuming the plants. Copper, zinc, nickel, and cadmium are examples of metals that can accumulate in soils and decrease plant growth (phytotoxicity). Cadmium and copper (to a lesser extent) can become hazardous at high concentrations to people or animals who eat the plants. The aquatic chemistry of metals in wastewater, ranges of the properties of soils that have an important influence on the behavior of metals in them, and a summary of benefits and hazards from metal accumulation in soils are discussed in this appendix.

E.2 Metals in Wastewater

E.2.1 Concentrations

The concentration of a metal in soil is probably the most important factor to consider. The short-term behavior of metals is influenced by the forms or species in the wastewater, but most metals are relatively immobile in soils. Thus, the assessment of the status of a metal in soil can be simplified to two major considerations: (1) the total mass input to a soil, and (2) vertical distribution of that mass when it is in a relatively steady state condition in the soil.

Metal concentrations in wastewaters, as affected by their sources and treatments, are important to a land application project because they may shorten the lifetime of the site through a cumulative total of one or a combination of metals in excess of a biological toxicity threshold. Page has reviewed and summarized some earlier published values for metals in wastewaters, and the effectiveness of standard treatment processes for their removal [1]. He makes the point that metal concentrations vary greatly in both soils and wastewater. High values in soils can arise from natural geo- or pedo-chemical accumulation processes [2].

Not only do individual treatment plants differ greatly in the concentrations of influent metals but Oliver et al. note that certain metals show rapid increases and decreases within waters of a treatment plant, which they attribute to sporadic industrial discharge of minimally treated metal-containing wastewater [3]. The wide range of trace metal concentrations in influents to municipal treatment plants is strikingly shown by the fact that biological activity in digesters can be inhibited by either metal concentrations that are too high [4] or to trace element deficiencies [5].

Sludge in conventional plants is generally found to retain much of the metals contained in the influent [3, 6-13]. Although proper operation of such systems can retain 50 to 75% of most metals in the sludge, lead and (especially) nickel are often not efficiently removed [13]. Poor settling of solids at one plant caused high carryover metal into the final effluent [13].

Advanced wastewater treatment processes (such as lime or chemical coagulant addition, carbon or charcoal filtration, and cation and anion resin exchange) can remove over 90% of metals from influent wastewater [14-17]. Effective processes convert the metals to separable solids by precipitation and/or adsorption. Mercury can be removed by these processes but participates in reactions that lead to gaseous losses as both dimethyl mercury and metallic mercury [18]. Effective metal removal, especially at the industrial discharge site, would slow the development of environmental hazards and extend the safe operating lifetime of a land treatment site, making the effluent more acceptable to potential users.

The problems of the municipal plant are considerably diminished by identifying the sources of wastes containing highly concentrated metals, and either treating or excluding them. Klein [19] reports that 25 to 49% of the metal in New York City wastewater influent is from domestic rather than industrial sources, but others note that certain metals traceable to specific industrial sources fluctuate dramatically [13, 20].

E.2.2 Species

The total concentration of a metal (M_T) in a volume element of unfiltered wastewater is the sum of the concentrations of the species of the metal:

$$M_T = MA + MB + MC + \ldots MZ. \qquad (E-1)$$

The metal associates, A, B, C, ...Z, have a large number of possible identities, some of which are catalogued and classified in Table E-1.

TABLE E-1

CLASSIFICATION OF SUBSTANCES WITH WHICH METALS MAY FORM CHEMICAL AND/OR PHYSICAL ASSOCIATIONS IN FRESH WATERS AND WASTEWATERS [21]

Complex and ion pair formers	Chelates	Precipitants	Adsorbents
H_2O	$R(COO^-)_x$	OH^-	Clay minerals
NH_3	Fulvates	CO_3^{2-}	Hydrous oxides (Al, Fe, Mn, Si)
OH^-	Humates	PO_4^{3-}	Humates
Cl^-	Polypeptides	S^{2-}	Fulvates
HCO_3^-	Polyaminosaccharides	SO_4^{2-}	Bio-remnants
CO_3^{2-}	Polyuronides		Calcium carbonates
SO_4^{2-}	Proteins		Iron sulfides
$RCOO^-$	Polyphosphate		Calcium phosphates
RSO_3^-			

Metal species are important in that they differ in chemical properties. The classes of soluble metal species include complexes, ion pairs, and chelates. Complexes and ion pairs are chemically similar. Structurally they have the metal ion at the center, which then coordinates or bonds or closely attracts to it one or more of the liquands listed in Column 1 of Table E-1.

An important complex of a metal ion in water is the aquo ion. It can be visualized as an ion such as a divalent zinc ion together with the water of hydration coordinated about it. Although this species may be an important intermediate in conversion from one species or form to another, it is often only a small fraction of M_T.

The formation of a monochloro- and a dichloro-metal complex ion is represented by the reactions:

$$Zn^{2+} + Cl^- = ZnCl^+ \qquad \text{(E-2)}$$

$$ZnCl^+ + Cl^- = ZnCl_2 \qquad \text{(E-3)}$$

These reactions should be interpreted to indicate that, at equilibrium, the solution contains some aquo metal ion (Zn^{2+}) as well as some of each of the complexes ($ZnCl^+$) and ($ZnCl_2$). Note that the complexes have a lower + charge than the aquo ion and thus are less likely than the aquo ion to be adsorbed by clays, oxides, or organic matter.

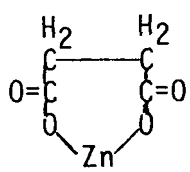
Chelates are similar to complexes and ion pairs in that the metal ion bonds or attracts around itself the "functional" groups of the chelate. Common functional groups of chelates in soils and waters include carboxylate, amino, and phenolate. The distinctive feature of the chelate is that two or more groups are connected by chains or bridges of atoms. Thus, acetate will form a complex with a metal ion because it has only one functional group, the carboxyl:

$$CH_3 - COO^- \qquad \text{(E-4)}$$

Succinate has two such groups

$$^-OOC - CH_2 - CH_2 - COO^- \qquad \text{(E-5)}$$

connected through a carbon chain and is thereby a chelate. Through the process of chelation, the succinate and the metal ion become an uncharged soluble metal chelate:



(E-6)

Equilibrium expressions can be written to quantitatively describe aquatic solution behavior of complexes, ion pairs, and chelates through stoichiometric expressions such as Equations E-2 and E-3 and the formation "constant." These constants are reported for the substances in Column 1, Table E-1, and for most simple organic acids and amino acids [21-23] as well as for higher-molecular-weight moieties such as fulvates in soils [24-26] and in wastewater solids [27]. Because most functional groups in chelates as well as most complex and ion pair formers are weak acids, the stability of the metal-moiety complex is often pH-dependent, with little association in acid media. The degree of association increases with pH to a maximum often determined by some competing alternative reaction, such as precipitation.

The important consequences of the formation of ion pairs, complexes, and chelates with metal ions in aqueous solutions are:

1. Total soluble metal concentrations are often greater than would be predicted from solubility considerations [1]. This is because solubility is a function of solubility product ("free" metal concentration or activity times "free" concentration or activity of precipitant). Total solution concentration is the sum of free, complex, ion pair, and chelated ion concentrations.

2. Uncharged ligands (H_2O, NH_3, RNH_2) do not diminish the positive charge of cationic metals but may result in a more polarizable cation, reduce the charge density, and increase the distance of closest approach to negatively charged surfaces.

3. Anionic ligands form metal associations that have lower positive charge than the "free" metal cation. The resulting association may be uncharged or it may have an overall net negative charge. The decrease in positive charge can thus reduce adsorption to negatively charged surfaces such as clay minerals in soils, thereby increasing the probability of leaching through soils to groundwater.

The full quantitative description of metals in solution phase of wastewater through analysis and computation is costly. Lagerwerff et al. [28] have proposed a resin-column procedure that estimates the concentrations of cationic, anionic, amphoteric, and uncharged forms of each metal in the original solution. Such an approach to characterizing metal species in wastewater effluents may be more economical than complete analysis--and still be precise enough.

The substances which form by association of metals with materials listed in Columns 3 and 4 of Table E-1 are particulate or high molecular weight. If settling, flocculation, and filtration are inefficient, however, they may remain suspended or dispersed in the wastewater and exit in the final effluent. The precipitants listed in Column 3 of Table E-1 may be present in the influent (SO_4^{2-}, PO_4^{3-}), may form by anaerobic processes ($SO_4^{2-} \rightarrow S^{2-}$), may be added as a treatment ($CaO \rightarrow OH^-$), or may be created during recarbonation ($2OH^- + CO_2 = CO_3^{2-} + H_2O$). During precipitation, the phosphates, sulfides, carbonates, and hydroxides or hydrous oxides may affect heavy metals by coprecipitating them and/or adsorbing them on solution-accessible surfaces of the precipitates. Parts of dead bacterial and other cells also have some capacity to adsorb metals, as do the humates and fulvates formed in microbial decay [29]. Clay minerals and clay-sized alumino-silicates and oxides entering in the influent also sorb metals. Column 4 of Table E-1 is thus a catalog of some of the materials that make up wastewater sludges and that account for much of the capacity of sludges to retain metals.

E.3 Receiving Soils

During a land treatment operation, changes and, especially, rates of change of the chemical state of the soil profile will be measured through monitoring, calculation, and projection to avoid hazards to future users from excessive accumulation. These efforts will require detailed knowledge of the "base level" initial soil composition before application begins. Some general discussion of vertical and horizontal metal distribution and variation in soils may be helpful in designing plans for sampling and analysis before, during, and after wastewater utilization.

E.3.1 Soil Analysis for Metals

Analytical philosophy can be divided into two categories: total and extractable. In following changes in soil composition, measurement of the total is preferred, for several reasons. First, the final results will probably be the most reproducible with complete breakdown and solubilization of all metal, no matter the distribution among mineralogical compartments. Second, many geological and pedological analyses report totals. Third, there is little correlation between total values and any extractable value [30]. However, full decomposition of the sample sometimes increases interferences in the final quantitation step. The greatest disadvantage is that the decomposition step for a "total analysis" is time-consuming and greatly increases laboratory costs per sample.

At the opposite extreme are procedures to extract small fractions of an element, often with the purpose of estimating "available" levels of nutrients essential to plants. Such procedures include extraction with dilute mineral or organic acid or synthetic metal chelates such as EDTA or DTPA, and measurement of "exchangeable" metal. Those procedures may admirably serve their original purpose, but their actual behavior in soils having high levels of metal have not been tested sufficiently. They have the distinct disadvantage that they will not fully extract alien metal introduced to soils through wastewaters and thus they cannot be used in mass balances.

A reasonable compromise is to extract soils with moderately concentrated hot solutions of mineral acids. These procedures extract far more of the total metal in a soil than the procedures for "available" metals, but they do not extract the metals in resistant minerals. Page [1] shows that the procedure of Andersson and Nilsson [31] for 2-molar HCl extraction at 100°C of soils receiving sludge applications for 20 years recovers or extracts high percentages of most of the applied metals. Such procedures simplify the final quantitation step because very little of the silt and sand dissolves, thus lowering the amounts of potential interferences, as would be the case in a "total" analysis.

In any extraction technique, scrupulous attention should be paid to using exactly the same procedural details from day to day. Early in the project, a large sample of soil from the project area should be prepared and stored for regular inclusion with each sample batch to ensure that changes in operator and operator technique do not cause systematic drift or variation in analytical output during the project.

E.3.2 Base Levels

The values in Table E-2, especially the averages, give a preliminary indication of whether the soil at a prospective site is near the norm or has an usual concentration of one or more elements. It should be emphasized that the given values are "totals." The data were selected to exclude samples taken near mineral deposits.

TABLE E-2

AVERAGE AND RANGES OF SOIL CONCENTRATIONS OF SELECTED ELEMENTS [32] mg/kg

Element	Average	Range
As	6	0.1-40
Cd	0.06[a]	0.01-0.7
Cr	100	5-3 000
Cu	20	2-100
Hg	0.03	0.01-0.3
Mo	2	0.2-5
Ni	40	10-1 000
Pb	10	2-200
Zn	50	10-300

a. Insufficient data reported. Values may need to be revised.

Fleischer discussed the mechanisms that lead to accumulations of metals by natural processes [33]. Soils and vegetation can have high levels of metals at considerable distance from a mineral deposit or ore body because the process which originally created the deposit was pervasive and/or because surface exposure of the deposit permitted erosive transport and deposition. A site proposed for wastewater application may thus be in the halo of a mineralized area. Application of metals in the wastewaters could rapidly bring the soils to a phytotoxic threshold.

For example, the Coast Range in California has at least two types of mineral deposit. Mercury in the form of cinnabar has been mined from a number of locations. Another type of "mineral" deposit is serpentine, exposed in a great number of locations of various sizes. This material is high in nickel and chromium. Erosion from the mountains and deposition in coastal valleys and on the west side of the Central Valley have probably caused some of those soils to have higher than average levels of these metals. The soils may thus have unexpectedly small capacities to accept nickel before declining in productivity.

The important point is that soils tend to reflect the chemistry of the "geochemical province" [34], so that they may have unusually high (or low) concentrations of specific metals even without mines or mining operations nearby. Exceptionally high levels of metals for any reason reduce the capacity to accept additional metal without exceeding environmental quality thresholds.

E.3.3 Vertical Distribution

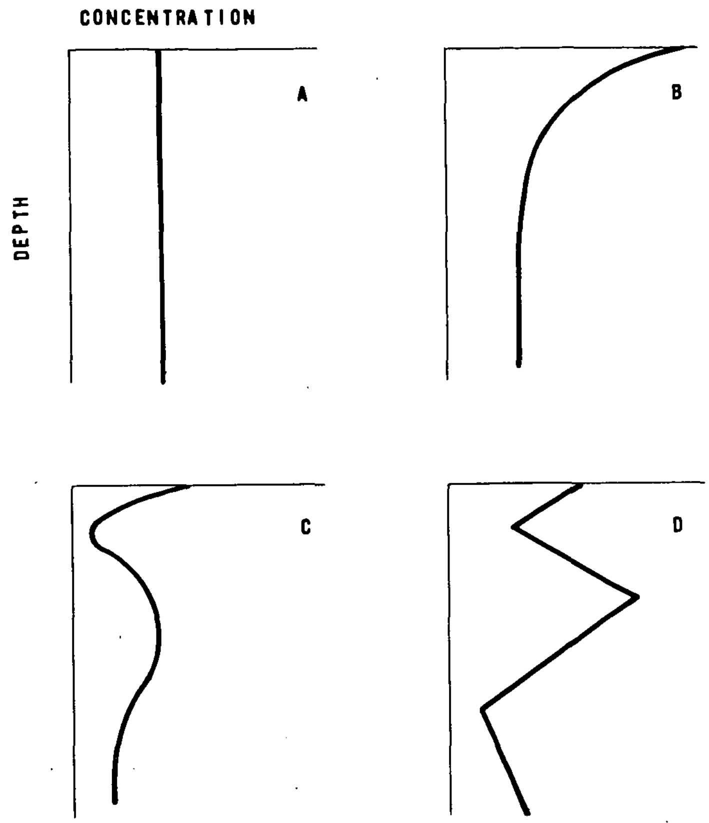
Not only do soils exhibit differences in metal concentration from one area to another (sometimes in surprisingly short distances) but the concentration often changes with depth in a given profile (Figure E-1). A profile is occasionally observed as shown in Figure E-1A. Such a condition might be observed in high rainfall areas, in tropical soils, or where leveling has removed the original surface soil for irrigation. Such profiles are relatively rare.

Figure E-1B is typical of the distribution of many metals in the soil profile. The apparent buildup or accumulation near the surface could be due to atmospheric input over a relatively long time, as with lead deposition near heavy highway traffic [35] or with deposition of lead, cadmium, and zinc from smelter smokestack [36, 37].

This same distribution is shown also by temperate-zone soils that have not been polluted. The mechanism involved is sometimes termed "plant pumping." During the thousands of years of soil development, plant roots take up the metal and translocate it to the aboveground leaves and stems. When this metal-containing biomass dies and falls to the

FIGURE E-1

PATTERNS OF DISTRIBUTION OF TRACE ELEMENTS WITH DEPTH IN SOIL PROFILES



ground, the soil insects ingest it and physically carry some of it down into the upper parts of the profile, where the mineralization process is completed by microorganisms. The apparent accumulation near the soil surface persists because the soil strongly adsorbs the metal, preventing it from leaching out. Such a mechanism is frequently ascribed to zinc, although it could possibly apply also to other trace metals taken up by plants, whether biologically essential or not [38].

A distribution that might occur in forested areas of higher rainfall is shown in Figure E-1C. The higher concentration at the surface is in the forest litter. The depleted zone has been leached by organic substances derived from decaying plant litter and moved downward a short distance, where it appears as an accumulation. Hodgson presents similar diagrams for distribution of individual metals in podzol soil profiles [39].

The profile in Figure E-1D will obviously be the most difficult to deal with because it is a series of discontinuities that might have developed by alluvial deposition of sediment from sources that have changed frequently.

E.4 Chemistry of Metals in Soil

The properties of a soil or sediment with respect to a metal can be characterized by the total concentration of the metal in the system. The importance of this parameter is based on the fact that a metal in a system will behave quite differently and be controlled by or participate in different reactions or mechanisms when in trace concentration than when at high concentrations. In most soils, which are neither extremely coarse nor very old nor highly leached nor highly polluted by geochemical or industrial activity, the zinc concentration ranges from 20 to 100 ppm, averaging about 50 ppm (Table E-2). Many laboratory chemicals have off-the-shelf contaminant concentrations of zinc greater than that value. Such soils, especially if in the range pH 6.5 to 8.0, have very high affinities for zinc, maintaining soil-solution phase concentrations of "free" zinc (aquo Zn^{2+}) in the range of 0.01 to 10 μg/L [40]. In fact, in some early studies, solutions of common reagents were sometimes purified of their zinc contaminants by passing them through soils. It is also important that, even at these low soil-solution concentration values, plants still acquire zinc through their roots fast enough to satisfy their biochemical demand.

Nevertheless, as more zinc (as a soluble zinc salt) is added to the soil-water system, much of the zinc "disappears" from the solution phase by mechanisms discussed below. The important points are that solution-phase concentrations of zinc increase and added zinc participates in more than a single reaction mechanism, distributing among several coexisting solid phase or interfacial states. It is important also that metal will accumulate unless water applied to soil has exceptionally low concentrations of metals.

Another familiar example from geochemistry is that of cadmium. Although carbonate and sulfide minerals of cadmium are known, most mineral deposits from which cadmium is obtained include cadmium as an impurity, probably coprecipitated with the major metal component (lead or zinc) at the time of crystallization. Therefore, we should not be surprised to find that, at trace levels in soils, metals exist in diffuse dependent states, not as discrete identifiable crystalline forms. As the total amount of metal in or added to a soil increases, the latter condition becomes more probable.

E.4.1 States of Metals in Soils

A summary classification of the states of metals in soils is presented in Table E-3 and is discussed in the following paragraphs. Several reviews discuss states and reaction mechanisms of metals in soils [1, 39-42].

TABLE E-3

CLASSIFICATION OF STATES OF METALS IN SOILS

Aqueous	Aqueous-solid interface	Solid
Soluble	Exchangeable	Biological
Dispersed or suspended	Specifically adsorbed	Precipitated
	Interfacial precipitate	Atom-proxied

E.4.1.1 Aqueous

The aqueous phase of soils includes only two important states, the soluble and the dispersed or suspended. The soluble state includes all forms of each metal previously discussed as aquatic species in wastewaters: aquo ion, complex ion, complex molecule, ion pairs, and low-molecular-weight metal chelates.

The dispersed or suspended state includes high-molecular-weight particles with metals adsorbed onto solution-accessible outer surfaces or included internally. These particles can peptize and move with the solution phase until electrochemical conditions change and flocculation again renders them immobile. Metals in the aqueous phase are subject to movement with soil water. They also participate in equilibria and chemical reactions with the solid phase.

E.4.1.2 Aqueous-Solid Interface

A very important region is the aqueous-solid interface, in which we can distinguish the exchangeable, the adsorbed, and the interfacially precipitated states.

E.4.1.2.1 Exchangeable

The surface of clays, oxides, and organic matter are negatively charged. That is, they have spots of negative charge at solution and cation-accessible locations on their surfaces. The sum of this charge is referred to as the cation exchange capacity (CEC). Positively charged cations such as calcium and magnesium, loosely held in the vicinity of these spots, are referred to as exchangeable ions. They are thought to be fully hydrated, to be in (thermal) motion, and to be "dissociated" from the surface [43]. Another characteristic of the exchangeable state is that insertion of a foreign cation (in a salt) into the solution phase readily (and predictably) displaces some of the "domestic" exchangeable cations.

E.4.1.2.2 Specifically Adsorbed

Some authors refer to this state simply as the adsorbed state. It is distinguished from the exchangeable state by having more binding between the metal and the surface. It includes the extra binding due to the covalency of the bonds that form during chelation by soil organic matter [24, 27, 44]. Metals specifically adsorbed at mineral surfaces are apparently held by electrical forces as well as by additional forces possibly including covalent bonding, Van der Waals forces, partial to complete dehydration, and steric fit at the site.

The term specific implies that other metal cations do not effectively compete or displace the specifically adsorbed metal cation. That is the practical distinction between the exchangeable and the specifically adsorbed state.

Specific adsorption is the most important mechanism controlling soil-water concentrations of metal ions at low amounts of metal in the soil-water system. As more metal ions are added, a specific adsorption capacity or limit is apparently reached, and incoming metal ions enter exchange positions. This concept is illustrated by comparison of studies of Blom with those of Bittell and Miller. Total cadmium was less than 1% of measured CEC in Blom's clay and soil samples; highly specific cadmium adsorption was observed [45]. On the other hand, Bittell and Miller also studied cadmium reactions with clays but at 10 to 90% occupancy of CEC [46]. They report selectivity coefficients of approximately 1.0 for calcium-cadmium systems, which clearly indicated no selectivity or specificity for cadmium.

The importance of the above is that metal ions will be relatively immobile and unaffected by high concentrations of "macro" salt cations such as calcium, magnesium, or sodium when specific adsorption is the dominant state. On the other hand, a metal cation will undergo greater leaching when the solution concentrations of macro cations are increased and the metal cation concentration in soil solution is controlled by the exchangeable state.

E.4.1.2.3 Interfacial Precipitate

This state is related both to adsorption and to precipitation. It is sometimes thought to arise through a process called heterogeneous nucleation [21]. In essence, already existing surfaces of clay minerals provide a host surface on which the cluster of ions can grow to become a crystallite. The precipitation process thus avoids a supersaturation step.

This theory implies that the resulting precipitate is identical in solubility to one produced through a supersaturation step. An extension of the theory suggests that the host surface in some cases affects the precipitate by making it more insoluble [47-51]. Data for copper and zinc equilibria in soils presented by Lindsay [40] can be interpreted as being due to interfacial precipitates of metal hydroxides. If the identity of the interfacial precipitate is known, it can presumably be managed like any other precipitate.

E.4.1.3 Solid

In addition to the aqueous and interfacial phases, the solid phase is important. This phase can be subdivided into the biological, the precipitated, and the atom-proxied states.

E.4.1.3.1. Biological

The metals which have passed across cell membranes into living cytoplasm are in this category. The organisms include microorganisms, plant roots, and the many insects and animals in soils. The state is important because it can temporarily sequester significant amounts of some metals and especially because it can cause transfer and accumulation of metals. This includes the uptake of metals by roots of plants and translocation to aboveground plant parts, thus tending to counteract downward leaching. On the other hand, earthworms and other saprophytes consume vegetative litter and distribute their decomposition products within the upper parts of the soil profile.

E.4.1.3.2 Precipitated

Precipitates include oxides, hydrous oxides, carbonates, hydroxy carbonates, phosphates, and, in reducing environments, sulfides. Clay minerals also can form by "precipitation." A precipitate can form only when the system contains sufficiently large quantities or high solution activities of the components of the solid. In the early stages of development of a land treatment project, such bulk precipitates are not likely to be an important factor in the control of soil-solution concentrations of the metals because the quantities of metals inserted into soil are still small and the adsorption process is favored energetically [52]. Various references [23, 40, 53] include thermodynamic data, applications, discussions, and diagrams of phase, pC-pH, Eh-pH, etc.

E.4.1.3.3 Atom-Proxied

Many metals are not major cationic components of precipitates but occupy structural crystal lattice positions of the major cation. This is called isomorphous substitution or atomic proxying by the trace foreign ion. The condition can develop in at least three distinct ways related to time. First, the precipitate may be forming rapidly while the trace metal adsorbed on the growing surfaces is coprecipitated.

Rapidly formed fresh precipitates often have low crystallinity and high specific surface and will often have greater solubility than aqed precipitates or precipitates formed slowly from "homogeneous solutions." Thus fast precipitates tend to dissolve upon aging and reform into more insoluble forms often containing less of the trace coprecipitate or atom proxy.

Slow precipitation is the second way and results in different distributions and quantities of the trace metals in the precipitate [54]. Some slow precipitates, such as clay minerals and some manganese oxides, have solution-stable forms with substantial amounts of proxying of octahedral cations. Krauskopf considers that copper, cobalt, and zinc are incorporated in aluminosilicate clay minerals as they form over geological time [55]. Nickel is proxied for magnesium in garnierite [56], and cobalt is closely associated with the manganese oxides in soils [57].

The third way for incorporation is by solid-state diffusion of the trace or foreign ion from the surface of an existing crystal into its interior (or vice versa), depending on concentration gradients. There has not been extensive study, however, of the degree and rate of such interchange between existing octahedral clay mineral cations and aquatic cations.

The coprecipitated or atom-proxied form of a metal has been looked upon as a sink into which metals can move, and thus it may extend the capacity of soil to accept metals before metal levels in the aquatic phase exceed tolerable threshold values. Unfortunately, the reaction rates and the parameters controlling reaction rates of this postulated mechanism are essentially unknown for any of the metals. Until the information becomes available, it would be wise to take a conservative position and discount the influence of coprecipitation as a sink.

E.4.2 Effect of pH

Trace metal concentrations in solution generally increase with decreasing pH. That is because most precipitant anions are weak acids and become soluble through protonation and displacement of metal cations in the solid phase. In addition, most specific adsorption sites (including interfacial hydroxy precipitates and chelates with soil organic matter) are pH dependent so that as pH declines the number of possible attachment sites diminishes. This is particularly true for hydrolyzable metal ions.

On the other hand, as pH rises, the solubility of a metal such as zinc passes through a minimum and then rises because of the formation above pH 9 of the soluble hydroxy complexes $ZnOH^+$, $Zn(OH)_2^0$, $Zn(OH)_3^-$, $Zn(OH)_4^{2-}$. Molybdenum (orthomolybdate ion) is an element which shows a general increase in solubility with increase in pH throughout the range of pH in natural sediments.

The general shape of the curve relating adsorption to pH while the amount of metal and soil or colloid is kept constant is shown in Figure E-2 [51]. In the lower pH range (A), adsorption increases with pH although the positive slope is relatively small (this may be specific adsorption). In the range B, adsorption increases abruptly. This occurs not only when massive amounts of metal are in the system, with the rise clearly due to precipitation of bulk hydroxides [58], but also when the aqueous system is clearly undersaturated with respect to bulk hydroxides [50, 51].

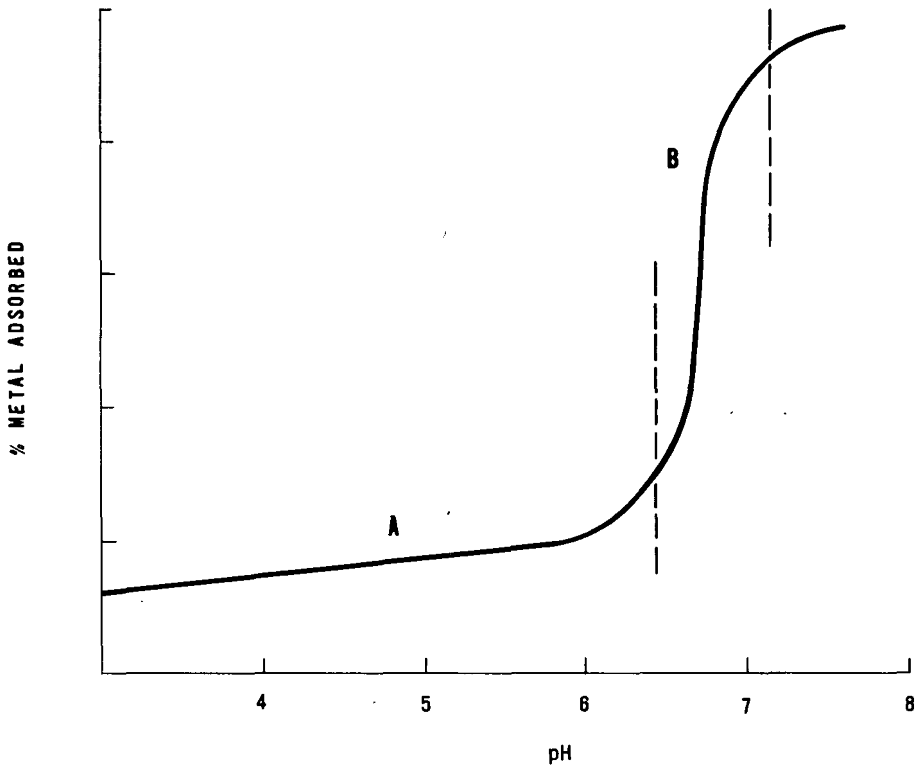
Thus, there is an inverse relation between pH and the capacity of a volume element of a soil or sediment to adsorb or precipitate metals. In some cases it may be advisable to lime the soil at the land treatment site to increase the capacity to retain metals and/or to counteract acidification of the system which sometimes results from nitrification of ammonium ion.

E.4.3 Adsorption-Desorption Isotherms

An isotherm consists of a series of laboratory observations at constant temperature. The effort is to obtain fundamental data on the interfacial, usually equillibrium, behavior of sorbates and sorbents in

FIGURE E-2

EFFECT OF pH ON ADSORPTION OF METAL
BY OXIDES AND SILICATES



two-phase systems. The two important phases are the aqueous liquid and the solid, especially the fraction of the solid that is finely particulate and thus has a significant quantity of surface which in the mixed system is the solid-liquid interface where "adsorption" occurs. The general objective of these studies is to fit the data to some model or mathematical function which will relate the mass or concentration of metal sorbed by the solid phase to some solution phase parameters such as concentration (or activity) of the metal (sorbate) ion, pH, concentration or activity of competing ions, etc.

Three major models or computational approaches have been used: the Freundlich expression, the Langmuir model, and the exchange model. The first two approaches are discussed by Ellis and Knezek with respect to trace elements in soil-water systems [59]. The Freundlich expression is:

$$x/m = kC^{1/n} \qquad \text{(E-7)}$$

where x/m = the mass (x) of the element adsorbed from solution per unit mass of adsorbent (m)

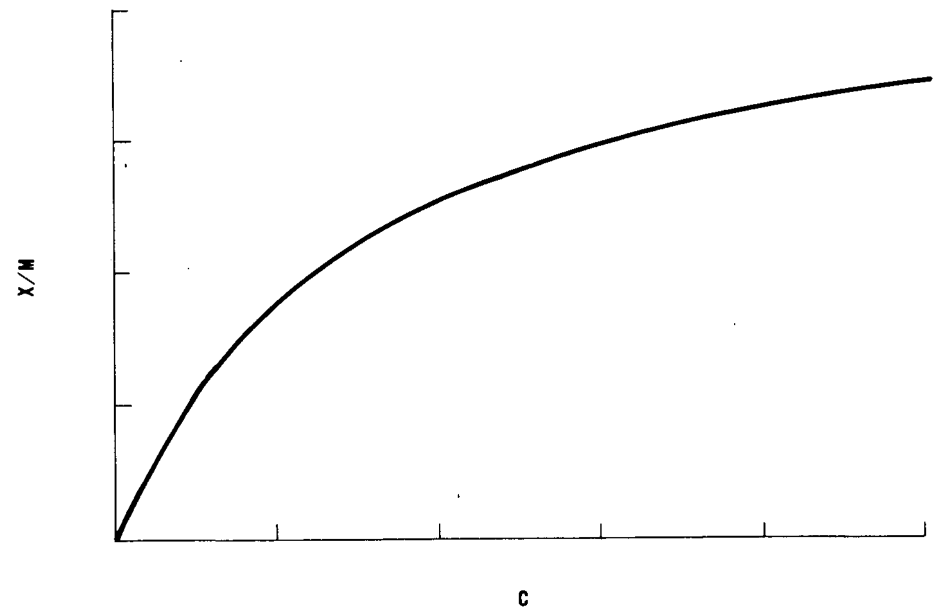
k,n = constants fitted from the experimental data

C = the concentration or activity of the metal ion in solution phase at equilibrium

This expression is essentially empirical but has the virtue of having an appropriate form to fit the graphical shape of many metal adsorption data. An example is shown in Figure E-3.

FIGURE E-3

TYPICAL ADSORPTION ISOTHERM FOR METAL SALT
ADDITION TO A SOIL- OR SEDIMENT-WATER SYSTEM



A second approach (Langmuir) assumes that no more than a monolayer of the adsorbing species will "attach" to the available surface, resulting in a maximum capacity generally designated by the symbol b , having the same dimensions as x/m. When the theory is applied to gas adsorption to "clean" surfaces, the spots or sites where the molecules attach are either occupied or vacant. Aqueous systems may contain competitors for occupancy, including water molecules, bound protons, or other chemical entities in solution phase. The theory assumes that the K in the expression below is constant over the working range of x/m. K is closely related to heat of adsorption in the solid-gas systems to which the theory was initially applied. The assumption of constancy of K is tantamount to assuming a single type of site or that all sites have equal energy.

The working Langmuir expression is:

$$\frac{x}{m} = \frac{KbC}{1+KC} \qquad \text{(E-8)}$$

Ellis and Knezek list a number of studies in which either Equations E-7 or E-8 have been reported to fit trace metal adsorption isotherm data [59]. It should be noted that at low C the expression is linear, which is essentially true for many hydrolyzable metal ions.

The third major approach is to consider adsorption as an exchange process and then to express the relationships in usual exchange functions such as the selectivity coefficient (K_B^A). Babcock [43] and Helfferich [60] describe the numerous exchange expressions and the nomenclature of the field. The model is one of a section of aqueous fluid (the film of water and ions in the interface between the surface and bulk aqueous solution) that is designated as the exchange "phase," while the interstitial fluid (the aqueous solution that can move under the influence of gravity or hydraulic gradients) is designated as the solution "phase." The two phases can be distinguished experimentally by defining as solution phase the equilibrium fluid and its ionic composition passing through a column of soil and determining the exchange "phase" composition by difference or by direct analysis. In fact, the boundary between the two "phases" in the column is probably diffuse.

Early in the operating life of a land treatment system, adsorption models will be more appropriate than ion exchange models. As more metal is incorporated, the exchange model may need to be included. At high pH and/or at high activity of other precipitant anions, solubility products may be appropriate. Such multi-equilibria computation models have been proposed and given limited testing under high loading with metals [52].

Most laboratory studies are adsorption studies, meaning that the metal is furnished to samples in increasing amounts, the mixture is equilibrated, and distribution measured. Far fewer studies examine desorption, i.e., a loaded adsorbent is subjected to successive volumes of aqueous solutions to determine again the equilibrium amount of x/m versus equilibrium solution concentration. At high loadings, the desorption curve is often very similar to the adsorption curve. However, significant hysteresis has been observed during desorption measurements [61, 62]. At low levels of cadmium, Blom found hysteresis in the direction of considerably "lower solubility" during desorption [45].

It is clear that the desorption behavior is of great importance in predicting leaching or downward movement of metals. The scarcity of such data, as well as of accurate data in the range of low loading, is a serious obstacle to predicting environmental behavior of many of the metals.

E.4.4 Kinetics of Metal Adsorption

Leeper uses the term "reversion" to describe the change of a soluble metal ion in soil to insoluble "second-" or "third-class" forms [41]. He also indicates that these processes can vary from very slow to very fast. Simple ion-exchange processes are ordinarily quite fast, having first-order "half-reaction" times of a few minutes [60]. Reaction rate curves for metal salt additions to soils tend to reach an apparent steady state in 1 to 2 hours [45]. A few studies suggest, however, that a slow reaction(s) exists which continues over long periods and converts some of the initial product to this secondary form [39, 41]. Both indirect and direct evidence suggests that much, if not all, of the modest fertilization applications to soil of zinc sulfate remains for several years in readily available but immobile forms [63, 64].

It thus appears that more facts are needed to establish mechanisms and rates of the slow reaction of metals in soils particularly over a range of soil metal content. Although evidence suggests that significant percentages of the small amounts of zinc applied as zinc sulfate fertilizers remain in the specifically adsorbed, labile form, the important question is whether the same is true for much higher application amounts of zinc and other metals.

E.5 Environmental Benefits From Metal Addition to Soil

A soil "deficiency" of an essential plant nutrient can be defined as the condition where rate of supply to the roots of the plant is insufficient to fulfill the functional demand of the plant. The deficiency is expressed in various ways, including reduced growth, lowered photosynthetic or respiration rates, and anatomical distortions and changes in dominance of plant pigments. Most deficiencies are specific as to site and/or plant species. Small applications of the deficient metal to either the soil or the foliage are often sufficient to overcome the deficiency. Any additional application has little apparent effect on plant growth up to a drastic diminution in growth from phytotoxicity.

The width of this plateau in the dose-response curve is dependent on species, soil, and the metal in question [32].

Among the trace elements that can be deficient in soils are boron, copper, molybdenum, and zinc. Classical techniques for correcting such deficiencies were recently reviewed and summarized by Murphy and Walsh [65]. Most amendments used for this purpose contain high concentrations of the element and are applied at rates usually less than 22 lb/acre (25 kg/ha) of metal for copper and zinc, less than 2.2 lb/acre (2.5 kg/ha) of boron, and less than 0.4 lb/acre (0.5 kg/ha) of molybdenum.

Hence, with trace-nutrient deficiencies in plants, only small quantities of the element are required to alleviate the problem. Any additional input through continued use of the wastewater is unnecessary.

The secondary benefit of metal addition to soil is improvement of the quality of the plant or plant part as a food for animals and humans. Human diets are only marginally sufficient in zinc [66]. Some individuals may have much higher requirements than the general population. Increased zinc contents of plants in response to zinc additions to soil are generally more rapid in the foliar portions than in the fruits or seeds [62, 66, 67]. Generalized accumulation of zinc in plant tissues as the zinc content of soil is increased through application of soluble or available forms of zinc is presented in Figure E-4.

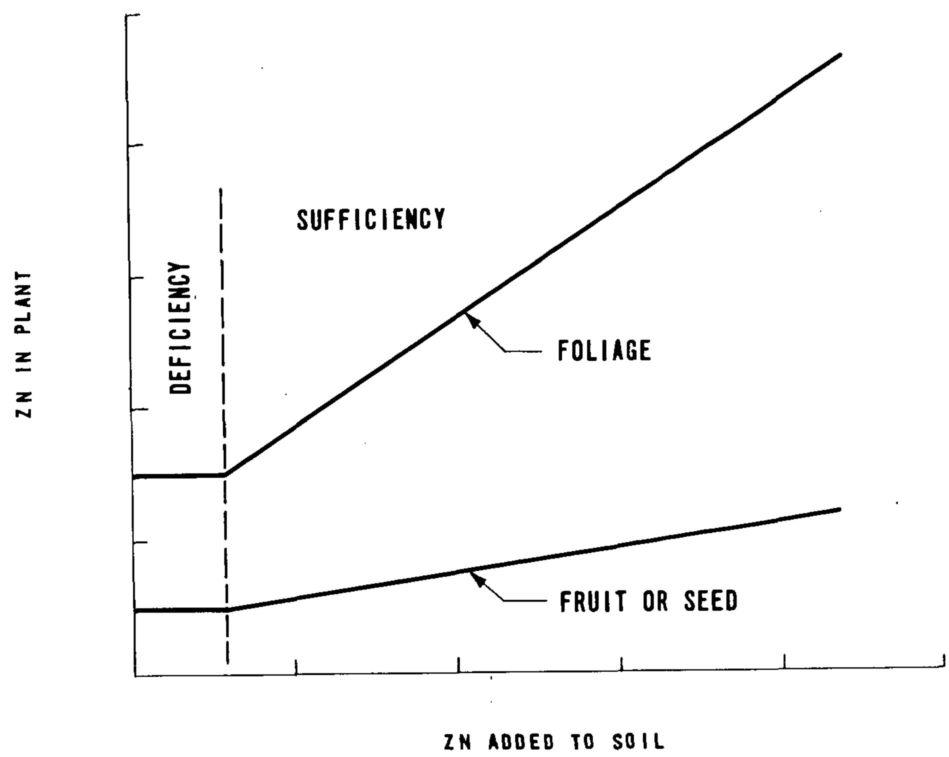
This performance pattern has many exceptions but is a reasonable first approximation for discussion. First, most foliar tissues have higher contents of the metal than fruits, seeds, or grains. Second, at deficiency levels, since small increments of added zinc cause additional growth, the extra biomass tends to dilute the extra metal taken up from the substrate, keeping foliar concentrations relatively constant. When the soil level is raised above the deficiency threshold, plant tissue concentrations of zinc tend to increase linearly [66]. The slope of this increase is much greater for foliage than for reproductive tissues, which sometimes have no detectable increase. The slope of the foliar curve is very different for different plant species. At very high levels of addition, uptake may become curvilinear and tends to reach a maximum. To a degree, the dose-response performance discussed above applies as well to other metals, including copper, nickel, and cadmium.

In generalizing about plant performance, it is also important to recognize that some metals are not efficiently taken up by plant roots. These metals include lead, arsenic, chromium, and, to a lesser extent, copper.

The increased zinc content of directly consumed plant parts could be of some benefit to humans. It is doubtful that grazing animals would

FIGURE E-4

SCHEMATIC RESPONSE OF A TYPICAL PLANT SPECIES TO INCREASING ZINC ADDITION TO A SOIL



benefit much, for zinc deficiency of animals is often the result of "ineffective digestion and use of dietary zinc..." [34]. These authors also point out that most plant tissue, even from plants suffering from zinc deficiency, is 10 ppm or greater. They cite studies which show optimum animal growth at 5 ppm in the diet.

The potential benefits from metals (or trace elements) in wastewater applied to land include enhancement of cobalt availability. Cobalt deficiency of ruminant animals is reported most frequently in the eastern and, especially, southeastern United States [34]. The usual treatment is to supply cobalt in salt or as a rumen bolus, and fertilization with cobalt has been reported to increase cobalt levels in forage to adequate levels [68]. Thus, cobalt in wastewaters might raise availability in soils where grazing ruminants suffer from cobalt deficiency, although some highly treated wastewaters may be too low in cobalt to affect soil cobalt levels measurably.

Another trace element food chain benefit could be the incorporation of selenium into pastures growing forages deficient in this element. Although deficiency of this element causes "white muscle disease" in sheep and cattle, antiquated legal restrictions prevent dietary management of selenium. Large areas of the United States are involved, including the west, northwest, and much of the east [34]. As with cobalt, however, it is questionable whether all treated wastewaters would contain sufficient quantities of selenium to affect soil levels and, in turn, forage contents.

E.6 Environmental Impact From Metal Addition to Soil

E.6.1 Input Greater Than Removal

The rate of input of metal in wastewater is the product of concentration and volume. Removal of metal from the soil (rhizosphere) can be by volatilization, by leaching or erosive runoff, and by removal of the plant (or animal) biomass which has acquired the metal by root uptake and subsequent internal transfer.

Volatilization can be a significant factor only for selenium, arsenic, and mercury. The mechanism can involve biological methylation of all three elements as well as simple reduction to the volatile metal for mercury [69]. Low oxygen or reducing conditions from flooding are generally necessary for a significant amount of either reaction.

Removal by physical erosion of soil particles to which metals are adsorbed is site specific and can be controlled by suitable barriers such as terraces and/or grassed waterways to carry tailwaters.

Removal by leaching downward through the soil with pore water will generally be negligible [1], unless:

1. The soil is very coarse textured and contains little clay or organic matter.

2. The specific adsorption capacity of the upper layers of soil is approached and pore water concentrations begin to rise.

3. There are present either significant concentrations of strong complex-forming ions (e.g., chloride, alkyl sulfonate, polyphosphates) or of low-molecular-weight organic chelates such as fulvates.

Removal of metal by plant uptake followed by export of the biomass from the site can be significant. Sidle et al. [70] define an accumulation index (A) as:

$$A = \frac{(Mw-Mp)\ 100}{Mw} \qquad \text{(E-9)}$$

where Mw = total quantity of heavy metal applied
Mp = total quantity removed annually by harvest

The index, which reflects the percent of applied metal remaining in the soil, was evaluated for copper, zinc, cadmium, and lead added in wastewater to a corn growing site and an area of Reed canary grass. It ranged from a low of 75.6% for zinc in the corn area to 99.1% for cadmium, also in the corn area. Of the 40 reported indexes, 34 were greater than 90%. Even in these instances of relatively small soil loadings, input was much greater than removal by crops.

Removal will obviously be much greater if the plants grown are selected for their ability to accumulate and thus remove metals from soils [71]. (Any use or disposal of such plants should be done in a manner that is environmentally nonhazardous.) Not only do species of plants vary greatly in their capacity to accumulate metals [32] but metal concentrations may be orders of magnitude higher in some parts of a given plant than in others [72]. Thus, removal by plants can range from essentially zero to a substantial amount.

The following simple model and mathematical analysis may be helpful in designing or predicting system behavior. The assumptions are that the yearly application rate of metal in wastewater is (k_1) in dimensions of g/ha·yr of the metal, and that the removal by plants is linear with respect to metal concentration in soil as expressed in the plant removal coefficient (k_2) having dimensions of 1/yr.

The differential equation for the model is:

$$\frac{dC}{dt} = k_1 - k_2C \qquad \text{(E-10)}$$

where C = concentration of metal in soil, g/ha of metal
t = time, yr

This equation emphasizes the concept that input is not always much greater than removal. Input will equal removal when the soil has reached a concentration equal to k_1/k_2. Therefore, the system will become steady-state at lower soil concentration if k_1 is minimized.

This can be achieved by lowering concentrations of the metal in the wastewater and/or diminishing the rate of wastewater application. It can be achieved also by maximizing k_2, that is, by selecting accumulator plants and/or by selecting plants with high rates of production of harvestable biomass.

Some assumptions about a soil-plant-wastewater system leading to values of k_1 and k_2 are given in Table E-4. Plant uptake rate is high and would be valid only for an accumulator plant. Some plants take up cadmium at one-tenth this rate [73]. The biomass production is also high and assumes vigorous growth throughout the spring, summer, and fall. The removal or harvest includes all aboveground portions of the plant.

TABLE E-4

ASSUMPTIONS AND SELECTED PERFORMANCE VALUES USED TO CALCULATE CROP REMOVAL COEFFICIENTS k_2 AND YEARLY APPLICATION OF METAL IN WASTEWATER k_1 FOR EVALUATION OF CADMIUM

Assumptions	
1. Plant uptake rate = $\frac{1 \text{ mg cadmium/kg plant}}{1 \text{ mg cadmium/kg soil}}$ [73].	$k_2 = 0.0033/\text{yr}$
2. Plant biomass harvested and removed = 10^4 kg/ha·yr.	
3. Cadmium mixed into upper 20 cm soil	
4. Low cadmium soil below 20 cm has no effect on cadmium uptake.	
5. Soil mass (1 ha x 20 cm) = 3 x 10^6 kg.	
1. Wastewater appplication rate = 100 cm/yr	k_1 = 1 000 g cadmium/ha·yr
2. Cadmium in wastewater = 0.1 mg cadmium/L	

With these assumptions and selections, the maximum concentration per unit soil area (C_{max}) at $t = \infty$ will be:

$$C_{max} = \frac{k_1}{k_2} = \frac{1\ 000}{0.00333} = 3 \times 10^5 \text{ g/ha} \qquad \text{(E-11)}$$

In 20 cm of this soil, the gravimetric C_{max} = 100 mg/kg. At such a high value, some of the simplifications in the model would be violated. First, the metal would not remain confined to the upper 20 cm but would move downward somewhat. In addition, phytotoxicity from cadmium and the copper, zinc, and nickel also added in the wastewater would probably decrease biomass production rates.

On the other hand, if cadmium concentration in the wastewater were diminished to 0.01 mg/L, C_{max} would be 10 mg cd/kg, a soil concentration tolerated by many plant species [74]. If the wastewater were effluent treated to contain less than 0.001 mg/L, the soil C_{max} would be less than 1 mg cd/kg. This is the value considered by Fleischer et al. to be an upper limit for unpolluted soils [33].

This analysis does not explicitly include time as a variable. Therefore the differential equation (E-10) may be integrated specifying that $C = C_i$, the initial soil content, at $t = 0$.

$$C = \frac{k_1}{k_2} + (C_i - \frac{k_1}{k_2}) \exp(-k_2 t) \qquad \text{(E-12)}$$

From this it is clear that C will never exceed k_1/k_2 at any time. Even though C_{max} is fixed by the defining differential equation, the t when $C = C_{max}$ is indeterminate. Even so, an impression of the time behavior of the system can be gained by calculating t_f, which is the time when $C = fC_{max}$. Selecting $k_2 = 0.0033/yr$ and setting $C_i = 0$ (which is tantamount to the case where $C_i << C_{max}$), we obtain the relationship plotted in Figure E-5. The calculated accumulation of cadmium in the soil at constant annual input of 1 kg/ha·yr is shown under two conditions: (1) no removal of cadmium from the field and (2) removal at a rate controlled by the biomass removed, assuming that cadmium concentration in the plant material is a linear function of cadmium in the soil. Numerical values for the time required to achieve selected fractions of C_{max} are as follows:

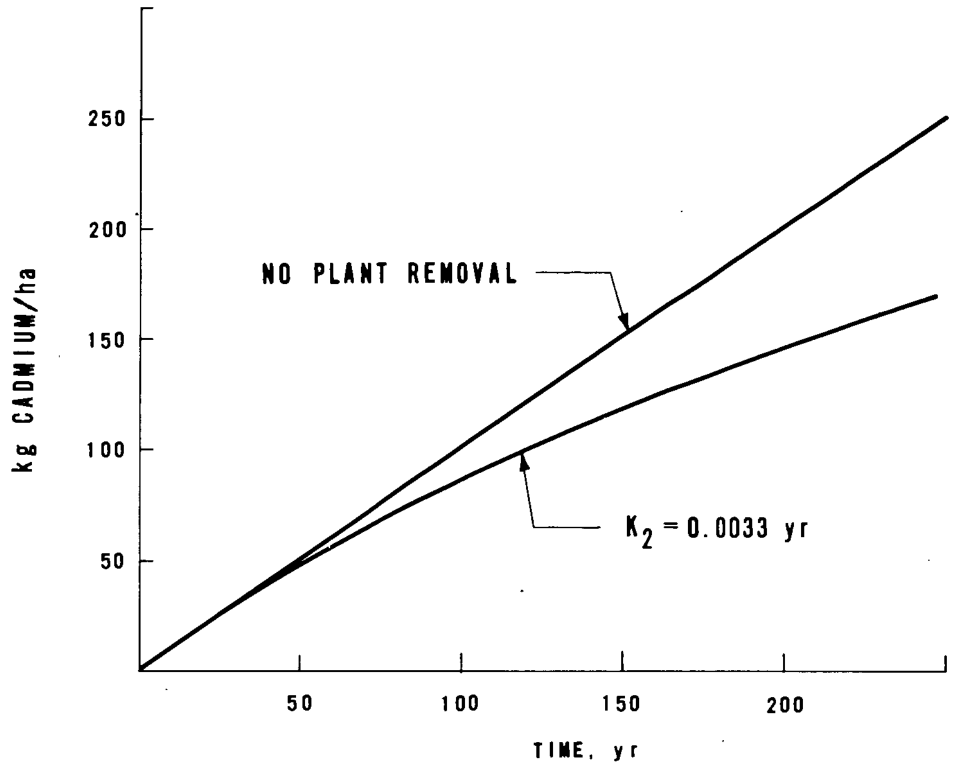
f	t_f (yr)
0.99	1 382
0.50	208
0.25	86

These values change slightly with the value chosen for C_i and the degree to which the assumption of negligibility of C_i is violated.

This analysis for the metal cadmium suggests that removal by plants (which includes harvest and biomass removal) may not be negligible. It may also be possible to extend this approach to other metals, such as zinc and nickel. The uptake of these metals by some plants also increases as soil metal content increases. Thus, the performance of a land treatment system might be roughly managed by judicious selection of plant species.

FIGURE E-5

CALCULATED ACCUMULATION OF CADMIUM IN SOIL
AT CONSTANT ANNUAL INPUT



E.6.2 Phytotoxicity

According to general principles of biological behavior, any substance administered in sufficient quantity and by an appropriate route can become toxic to a given organism. In the classic dose-response curve [32], at low substrate levels of an essential mineral element, the organisms or one or more biochemical subsystems in the organism operate suboptimally. As the substrate level or concentration is raised, a threshold plateau is reached, and at some still higher level toxicity sets in. That is true for the biologically essential trace metals, such as copper and zinc. For cadmium, however, lowered biological performance cannot be demonstrated at very low levels of cadmium in the substrate (nutrient solution, soil, etc.), so cadmium is among the elements which "is not known to be essential." On the other hand, cadmium, like copper and zinc, is easily demonstrated to be toxic to plant growth.

Bowen presents summary information on toxic levels of nearly all elements [32]. Instances of natural phytotoxicity are known. The most widespread and well known are phytotoxic levels of nickel in serpentine soils [75] and of boron in arid zone soils [76].

A number of reports detail the creation of phytotoxic conditions in at least a part of the rhizosphere from copper residues from Bordeaux sprays [77], from zinc sprays [78], and the classic situation of persistent phytotoxicity from arsenic residues in soils of old apple orchards sprayed with lead arsenate to control codling moth in the era before DDT [32]. Even when the chemical(s) are no longer used, the phytotoxic condition is often reported to persist for decades; and in the case of nickel in serpentines, the persistence is for centuries and millennia. This background would certainly suggest prudence in developing standards for maximum soil accumulation levels, especially in currently productive soils that are projected to be used in perpetuity for food production.

The literature is replete with discussion about potential phytotoxicity developing from application of wastewaters and wastewater solids to soils [79-87]. The metals most frequently regarded as potential phytotoxicity hazards in such materials are copper, zinc, and nickel [79], with recent reports also emphasizing cadmium [73, 74]. Hinesly et al. detect decreasing availability of cadmium after incorporation with sludge and suggest that the problem of phytotoxicity may have been "greatly overstated" [88].

A recent report by the Council for Agricultural Science and Technology examines the potential effects on agricultural crops and animals by heavy metals in wastewater sludges applied to cropland [89]. The report concludes that many metals are not a significant potential hazard, either because they are generally present in low concentrations, are not readily taken up by plants under normal conditions, or are not very toxic to plants and/or animals. Several metals (particularly Cd, Zn, Mo, Ni, Cu) are labeled as posing a potential serious hazard under certain circumstances, however, with cadmium presently being the metal of most concern.

For wastewaters, boron should be added to the phytotoxicity list. Unlike the metals, boron as H_3BO_3 or as $B(OH)_4^-$ is relatively less persistent, and phytohazardous soil concentrations can be removed from soil by leaching. That is likely to be the situation in most cases.

An additional caution is that many other possible metallic constituents exist in all wastewater and any unusual use or disposal in the collection system may result in special toxicity hazards. Chromium as anionic chromium VI is very phytotoxic [80]. However, any chromium entering a treatment system in this form will almost certainly be reduced to chromium III, which is very insoluble and much lower in phytotoxicity.

Looking at the question in the long term, the most valid observations or experiments will be those that can simulate conditions and properties of soils that have had metals added in organic combination but now have only vestiges of the organics because of bio-oxidation over time. Furthermore, any tendency for sequestration by coprecipitation or solid state diffusion into clay mineral structural lattice positions will be reasonably advanced, if not at virtual equilibrium, and vertical distribution should be reasonably stable. Such a condition might be achieved 1 to 5 decades after metal-plus organic input to the soil ceases. Observation of long-term treatment sites should be valuable in this regard.

The following can be concluded from the literature:

1. Phytotoxic buildup will never occur in some soils receiving wastewaters, because they contain too little clay and organic matter to serve as a nucleation surface for accumulation. These highly rocky, gravelly, or sandy soils will simply transmit the metals to underground regions of finer textures or to underground waters.

2. Metals may eventually accumulate to phytotoxic levels in all other soils. The threshold will depend on the plant species, soil pH, surface area, and the combined levels of the metals accumulated. The critical factor is obviously the rate of metal input, which is the integral of volume applied and concentration in the wastewater. Depending on the quality of the treatment process, this time might range from 50 years to infinity if advanced wastewater treatment processes are used.

Threshold standards have been proposed for preventing phytotoxic buildup of metals. One of the early proposals was the zinc-equivalent concept of Chumbley, which states that the soil should have a pH >6.5 and that the maximum addition of metal to a soil should not exceed 250 ppm zinc or its combined equivalent of zinc plus copper plus nickel [90]. An equivalent of copper was calculated by taking double the actual gravimetric addition of copper, while for nickel the multiplicative factor was eight. This was based on assumptions that the toxicity of the three metals in combination was additive and that copper was twice as toxic, while nickel was eight times as toxic as zinc. Leeper noted that the formula lacked any factor to account for the differential capacity of different soils to accept metals [41]. A further difficulty is a general lack of an experimental data base to support the assumptions of additivity and of relative contribution to toxicity. King and Morris do suggest that their data show an additive effect for copper and zinc [86].

Other proposals to calculate a limiting application quantity include those of DeHaan [83] and Water Quality Criteria [91]. No existing experimental evidence gives impressive support to one or another approach to calculating a threshold for maximum all-time input to prevent future phytotoxicity.

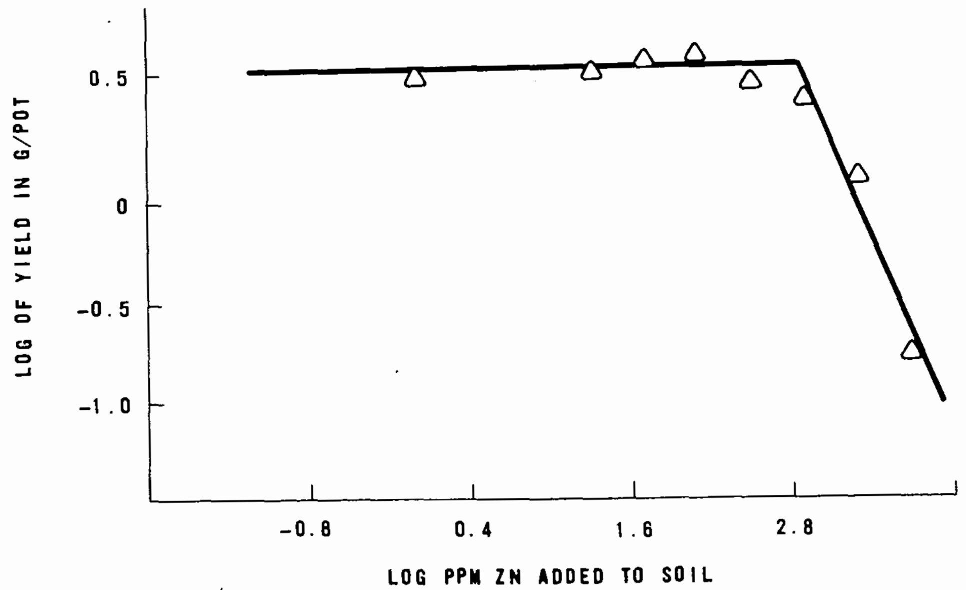
One question in studying toxicity is the point in the response curve which clearly indicates toxicity. In any greenhouse study, plant growth and performance varies considerably between pots of the same treatment (including the check), thus making mandatory statistical experimental design and treatment of the resulting data. Field experimentation data usually have higher coefficients of variability because a wider range of parameters affecting plant growth are not under experimental control.

One technique has been to select a given yield decrement (percent decrease below the maximum yield) as a point in the dose-response curve where a toxic response is statistially defensible. Bingham et al. [73, 74] select a 25% yield decrement as a phytotoxicity index for cadmium, while Boawn and Rasmussen [92] select a 20% decrement for the same purpose in zinc phytotoxicity studies. Such indexes are very valuable for discussion but tend to obscure the fact that some degree of toxicity occurred at lower system loading. This toxicity could perhaps have been detected with more sensitive experimental and statistical techniques.

A somewhat more conservative approach is to define a "threshold value" from a log-log plot of the dose-response curve shown in Figure E-6. The general approach is based on the empirical observation that many of these plots appear to be made up of two linear segments, which then suggests using the antilog of the x-axis value at the intersection of the lines as the "threshold." Although it is somewhat clumsy, a combination of regression and/or analysis of variance could be used to fit the data to two straight lines from which the intersection could then be calculated and used as an index.

FIGURE E-6

LOG-LOG PLOT OF DRY MATTER PRODUCTION OF SWEET CORN AS A FUNCTION OF ZINC (ZINC ACETATE) ADDITIONS TO SOIL [93]



Before this approach is adopted, another factor should be discussed. The nearly horizontal line at application rates below the threshold sometimes has a slope not statistically different from zero. At other times, the slope is significant and negative. This can be interpreted as indicating an early mechanism of toxication, perhaps different from the catastrophic mechanism seen as the steep negative slope after the threshold. A further suggestion would then be to define a second threshold as the intersection of the shallow-slope line (if it exists) with a horizontal line equal to a maximum performance value. The first (lower) intersection could be named the no-toxicity threshold limit value, while the higher might be named the acute-threshold limit value. Only the latter will be found consistently.

This raises the issue of safety factor. Even though this acute-threshold value can be measured or evaluated experimentally for a plant species growing on a soil, it is clearly a function of both. That is, plant species (and perhaps cultivars) vary greatly in susceptibility to metal toxication [73, 74, 92], and soils certainly vary in capacity to adsorb metals. Furthermore, soils may degrade in this capacity particularly through a decrease in pH. Therefore, thresholds should be measured with sensitive plants and at different soil pH values unless it can be guaranteed that the soil will be chemically managed to have a pH above some arbitrary minimum. Some tolerance formulas contain the proviso that the soil have a pH value greater than 6.5.

However, it is difficult to guarantee that future users of the contaminated land will manage the soil to have higher pH values. Through neglect, pH values of agricultural soils may fall through pedogenic factors such as high rainfall and high temperatures, causing acidification. In other cases, soil pH may be deliberately lowered through applications of sulfur or other acid-generating chemicals to inhibit plant pests as in the control of scab in potatoes. Since (for pedogenic reasons) soils that are acid are likely to differ considerably from neutral or alkaline soils that are (or can be) acidified, it is recommended that threshold values (or performance curves) be estimated or observed at two or more pH values. (Some soils, because of a high lime content, cannot practically have their pH lowered below 7.)

Still another index or diagnostic indicator to toxicity is plant tissue composition at or near the threshold of toxicity. Such values for several metals are presented and discussed by Leeper [41] and for cadmium by Bingham et al. [73, 74] and Iwai et al. [94]. Page points out that the concept has some value in determining the cause of diminished plant performance; but since a positive diagnosis is, by definition, after the fact, it has little value in designing systems to prevent phytotoxicity [1].

E.6.3 Food Chain Hazard

The food chain involves acquisition of the metal by plant roots, transport into edible portions of the plant, and then consumption by the primary consumer. The primary consumers may be humans as in consumption of grains, vegetables, and fruits, or they may be animals that eat the forages and grains. Humans are secondary consumers when they ingest animal products.

The question is whether the metal can be transferred in quantities or at rates that would pose a chronic or an acute toxicity hazard to primary or secondary consumers. Except for certain accumulator species [32], plants are excellent biological barriers [84]. That is notably true for nickel, copper, and lead [66, 70]. Although lead toxication of animals near smelters is frequent, toxic concentrations of lead in pasture forage are generally believed to be accumulated primarily from atmospheric deposition rather than by root uptake and translocation [36, 84, 95].

An exception to the plant barrier rule enunciated above is the potential for toxication of ruminants consuming forages having either a very high or a very low ratio of molybdenum to copper [1, 66]. Page concludes that molybdenum accumulation in soils from wastewater solids application and subsequent increase in forage molybdenum content is a potential hazard to grazing ruminants, especially where soils are neutral or alkaline in reaction [1].

For the other metals mentioned above, the plant root is generally an effective barrier. With lead, nickel, and copper, the root provides the barrier since uptake and, especially, translocation are low. Baumhardt and Welch show a significant but small increase of lead in corn stover from lead acetate applications to soil (3 200 kg/ha) although the corn grain content was only 0.4 mg/kg of lead and not affected by application rate [96]. No evidence of phytotoxicity was observed.

Nickel and copper have the added protective mechanism of preeminence of phytotoxicity. Leeper cites recent respective literature values for copper and nickel of 30 and 25 mg/kg plant tissue in plants at the phytotoxic threshold [41]. Thus, not only is uptake and translocation of these elements low but the plant dies or fails to grow long before it can accumulate a metal content toxic to a mammalian consumer.

Zinc, in contrast, is more readily translocated to foliar tissues of plants. Boawn and Rassmussen show 770 mg/kg of zinc as a high tissue concentration in spinach at the 20% yield decrement (toxicity) threshold in plants grown on neutral and alkaline soils spiked with various amounts of zinc nitrate (Table E-5) [92]. The mg/kg of metal concentration values for animal forage tissues at the 20% decrement were much lower, being 460 for corn, 475 and 570 for sorghum, 540 for barley, 560 for wheat, 295 for alfalfa, and 252 for clover. Underwood cites several reports of animal toxicity feeding trials with zinc and concludes that rats, pigs, sheep, poultry, and cattle exhibit considerable tolerance to high zinc intake, depending on the composition of the diet [66]. Most of the reports show no pathological symptoms at a 1 000 mg of zinc per kg of diet except in lambs, heifers, and steers, which showed reduced gains thought to be due to reduced feed consumption because the high zinc content made the diets less palatable. Higher dietary levels caused detectable pathology. In some cases, anemia developed in addition to depressions in cytochrome oxidase and catalase activity. The apparent lower zinc toxicity threshold for ruminants was suggested to be due to effects on rumen microflora. Zinc levels of 4 000 mg/kg of diet caused internal hemorrhages in weanling pigs, and 10 000 mg/kg caused heavy mortality in rats. Underwood does not report information on zinc toxicity in humans [66].

TABLE E-5

CONCENTRATION OF ZINC IN PLANT TISSUES AND INTERPOLATED APPLIED ZINC CONCENTRATION IN SOIL AT THE 20% YIELD DECREMENT [92]
mg/kg

Crop	Tissue content	Soil
Corn	460	286
Sweet corn	400	231
Sorghum	475	175
Sorghum	570	200
Barley	540	200
Wheat	560	324
Alfalfa	295	456
Peas	420	429
Lettuce	430	415
Spinach	770	400
Sugar beet	670	447
Tomato	450	452
Beans	257	500
Clover	252	500
Peas	490	500
Potato	327	500
Potato	346	500

Thus, the question of food chain transfer of toxic quantities of zinc cannot be dismissed as easily as with lead, copper, and nickel. It appears that accumulator plants, at the 20% yield decrement toxicity threshold, can acquire concentrations of zinc that might affect primary consumers either through loss of appetite or in ruminants, through negative effects on rumen microflora. At still higher soil zinc levels, plant biomass production will diminish and zinc content will probably rise. It is possible that the combination of a soil having in excess of 1 000 mg/kg of zinc could produce forage which would give overt toxicity symptoms in primary consumers. This statement is supported by the data in Table E-5. Boawn and Rasmussen, who generated the data, used a Shano coarse silt loam soil having pH values ranging from 7.0 to 7.5 [92].

Therefore, soil levels of added zinc in excess of 500 mg/kg may cause, at least, a decline in forage quality through lowered palatability, and, at worst, some overt toxicity symptoms. It is also important to recognize that some of those species exhibited toxicity below the 250 ppm threshold [90] even though the soil was above pH 6.5.

Humans are probably protected from food-chain transfer toxicity because their diet ordinarily includes fruits, grains, and animal meat. In all cases, zinc transfer from substrates high in zinc is much lower into these tissues than into foliar tissues of plants.

Cadmium is currently the element of greatest concern as a food chain hazard to humans. Its acute toxicity has been reported at 75 mg cadmium per kg diet of Japanese quail [97]. Acute toxicity to humans has been reported from consuming acidic foods prepared or served in cadmium-plated containers [98]. The more general alleged hazard to humans, however, is one of chronic toxicity, expressed only after long exposure. Several recent reviews present salient facts and thinking about cadmium [98, 99, 100].

Nordberg summarizes the known safety values for cadmium, including the recommendation of a joint committee of the World Health Organization (WHO) and the Food and Agriculture Organization (FAO) for permissible weekly cadmium intake of 400 to 500 μg/wk of cadmium (57 to 71 μg/d of cadmium) [100]. United States regulatory agencies have not established threshold tolerance dietary intake values either for individual foods or for the diet in general [101], except for the USPHS upper tolerance value of 10 μg/L of cadmium for domestic water supplies [91].

Most of human intake of cadmium is through the diet except during industrial exposure, where the inhalation route may be significant [99]. A great number of pathological conditions are alleged to be caused by or exacerbated by excessive cadmium intake [98, 99] including the widely publicized Itai-Itai disease of Japanese women. Friberg et al. suggest that the most sensitive organ in mammals is the kidney, which accumulates much of the cadmium absorbed into the blood from the gut. They claim that an established threshold toxicity limit value is cadmium at 200 mg/kg in the kidney cortex [99].

Base level average dietary intake of cadmium has been reported to range from 25 to 75 μg/d for an adult consuming 1.5 kg/d of food [99]. Intake rates are increased by eating organ meats as well as by eating some sea foods, notably shellfish [98]. (Fulkerson and Goeller consider that some reported cadmium values in tissue may be invalid because of analytical problems in the laboratory.)

Friberg et al. propose a model of human toxicity in which chronicity is projected over a 50 year period [99]. It is essentially computational. The major question is whether soil levels of cadmium can reach some upper value which would allow the transfer of cadmium into foods at levels that would cause toxicity in adult humans. A number of recent reports [67, 73, 74, 102, 103] study the concentrations of cadmium in plant tissues that result from additions of cadmium salts to soils (or to sludge amended soils). Other reports present tissue content sampled from contaminated as well as uncontaminated soils [33, 98, 99, 100]. The studies generally show increased plant content with increased soil content except in rice grown under flooded conditions. Uptake is more strongly a function of plant species and plant organ than of soil properties, including pH and "exchange capacity." Another tendency is for uptake to be approximately a linear function of soil cadmium content, but with exceptions. Since cadmium tends to be partially excluded from fruits, foliar tissues tend to be higher than fruits, seeds, or grains, and roots tend to have the highest concentrations. (However, the edible portions of root crops are not always much higher in cadmium than corresponding foliar tissues.)

The aforementioned studies do not directly answer the question of "food chain hazard." All the same, we have the Japanese experience in the Jintzu Valley, where it is legally recognized that cadmium contamination of food-producing soils by metal-ore wastes was the primary cause of some unusual disease symptoms as well as many cases of premature death [98, 99]. Soil levels as well as dietary intake are not extensively reported, although Friberg et al. report data of Fukuyama and Kubota [99] that contaminated paddy soils in the Jintzu River Valley contained cadmium at 0.2 to about 4 mg/kg. The contaminated Jintzu River water was used not only to flood the rice paddies but also as a water source for the inhabitants of the valley.

With the present United States system of food production and distribution, it is highly unlikely that produce from the relatively small land areas receiving cadmium in wastewater would pose any more of a hazard to the national diet than foods naturally high in cadmium such as shellfish. This presumes that foods enter a system that mixes products from many parts of the country during distribution to retail grocery outlets.

On the other hand, if soils contaminated with high levels of cadmium were used by individuals or families for gardens in which a high proportion of the family vegetable diet was produced, the situation would become similar to that in the Jintzu Valley. The inhabitants of that valley were apparently highly self-sufficient with respect to food and thus became unwitting victims of their deteriorating environment. Unless the land that might be contaminated by cadmium is dedicated to other land uses, the long-term hazard to intensive, self-sufficient food production must be considered in design of a land treatment system.

In summary, direct application over extended periods of raw wastewaters that are high in metals can induce phytotoxicity in soils, diminishing their productivity. Food-chain hazards to ruminant animals may result from copper-molybdenum imbalances in forage. Present systems of general production and distribution make unlikely chronic cadmium toxicity to humans from cadmium transferred from soil to vegetables or other directly consumed plant parts purchased in the market. It is a potential problem, however, if the soil becomes a "backyard garden."

However, treated wastewaters that have lower metal concentrations achieved by separation of the metals into sludges or by advanced treatment techniques can probably be applied to land for many decades without the creation of any foreseeable hazard.

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APPENDIX F

FIELD INVESTIGATION PROCEDURES

F.1 Introduction

During the facilities planning or design stages, it may be necessary to conduct detailed field investigations to verify or refine design criteria. When data on crop water requirements are not available, estimates can be made on the basis of prediction equations. When soil surveys do not provide adequate data, it may be necessary for a soil scientist to conduct soil investigations, including soil mapping, sampling, and testing. In this appendix, several methods of estimating crop water requirements are described, and the steps involved in soil mapping and sampling are outlined. The significance of physical and chemical soil properties as they relate to system design is discussed; hydraulic properties were discussed in Appendix C. Procedures for measuring physical and chemical properties of soil are given, and guidelines for the interpretation of soil test results are presented along with suggested information sources. Techniques for evaluating subsurface hydrologic properties to estimate groundwater flow were presented in Section C.3.

F.2 Crop Water Requirements

The water requirement of crops, or evapotranspiration, is a major part of the overall water balance for slow rate systems. In areas where historical evapotranspiration data are not available, it will be necessary to make estimates or to take measurements. Measurement of evapotranspiration for specific crop requirements under field conditions prior to design is not usually practical because of time limitations. The designer must therefore rely on prediction equations.

All predictions are based on an empirical correlation between evapotranspiration and various measured climatological parameters. Over 30 methods have been developed internationally for different agronomic and environmental conditions, and these are detailed in a recent ASCE publication [1]. On the basis of recommendations made in a recent publication of the Food and Agriculture Organization (FAO) of the United Nations [2], four methods appear to have potential for widespread use:

1. Modified Blaney-Criddle method

2. Radiation method

3. Modified Penman method

4. Evaporation pan method

The method selected for a given project will depend on the types of climatological data available. Complete details on the applications of these methods are presented in the FAO publication [2]*. The procedures in this publication are recommended because they allow corrections to be made for local climatic conditions and are thus more universally applicable. In addition, the procedures have been developed for use by the practicing engineer who does not have a background in meteorology, soil physics, or plant physiology. For an in-depth discussion of the fundamentals of evapotranspiration, the user is refered to the ASCE publication [1].

Prior to selecting the prediction method, data from completed climatological and agricultural surveys, specific studies, and research on crop water requirements in the area of investgation should be reviewed. Available measured climatic data should also be reviewed. If possible, meteorological and research stations should be visited, and the environment, siting, types of instruments, and observation and recording practices should be appraised to evaluate the accuracy of available data. The prediction method may then be selected on the basis of the types of usable meteorological data available and the level of accuracy desired. The types of data (measured or estimated) needed for each method are summarized in Table F-1. The methods are described very briefly here along with some general criteria for their selection and use.

TABLE F-1

TYPES OF DATA NEEDED FOR VARIOUS EVAPOTRANSPIRATION PREDICTION METHODS

Factor	Modified Blaney-Criddle	Radiation	Modified Penman	Evaporation pan
Temperature	Measured	Measured	Measured	....
Humidity	Estimated	Estimated	Measured	Estimated
Wind	Estimated	Estimated	Measured	Estimated
Sunshine	Estimated	Measured	Measured	....
Radiation	....	Measured	Measured	....
Evaporation	....	....	....	Measured

*Available from UNIPUB, Inc., 650 First Avenue, P.O. Box 433, Murray Hill Station, New York, N.Y. 10016.

The Blaney-Criddle method, as modified in the FAO publication [2], is recommended when only air temperature data are available and is best applied for periods of one month or more. It has been used extensively in the western United States and is the standard method used by the SCS [3]. In the eastern United States, it is less widely used and often yields estimates that are too low [1].

The radiation method is recommended when temperature and radiation or percent cloudiness data are available. Several versions of the method exist, and because they were mainly derived under cool coastal conditions, the resulting evapotranspiration generally tends to be underestimated [1].

The modified Penman method is probably the most accurate one when temperature, humidity, wind, and radiation data are available. Along with the radiation method, it offers the best results for periods as short as 10 days.

Evaporation pans offer the advantage of responding to the same climatic variables as vegetation. Depending on the location and surrounding environment of the pan, pan data may be superior to data obtained by other methods. Because of the influence of the surrounding environment and the pan condition on measured evaporation, the data must be used with caution. Pan evaporation data are best applied for periods of 10 days or more.

As mentioned previously, many other prediction methods may be used. Some are based on correlations with certain climatic conditions and cannot be easily adapted to other conditions. For example, the Thornthwaite method, in which temperature and latitude are correlated with evapotranspiration, was developed for humid conditions in the east-central United States, and its application to arid and semi-arid conditions will result in substantial underprediction of evapotranspiration [1]. Because of its relative simplicity, it has often been applied in areas for which it is not suited. It is important to select the prediction method that can make use of the available data and that can be corrected for local climatic conditions.

Measurement methods such as soil-water depletion or lysimeters may be used if time allows, or if pilot systems are considered [4]. Other methods, used mostly in research, are discussed in the ASCE publication [1].

F.3 Soil Investigations

Most renovation of wastewater takes place in the first 5 ft (1.5 m) of the soil profile. A rather accurate, quantitative knowledge of the properties of this section of the soil profile is necessary to design a land treatment system that will operate within the infiltration capacity of the soil. In many cases, prospective sites are located within areas where a complete soil survey has been conducted. When data from such surveys are available, the tasks involved in determining soil properties through field investigations, with the exception of soil infiltration rates, are reduced to confirming information presented in the survey report.

If detailed soil surveys are not available or more detailed information is needed for the areas of interest, more extensive field work will be necessary to define soil properties and their areal extent more accurately. This work may include detailed soil mapping to define boundaries of soil types within the area, field and laboratory analysis of soil to determine physical and chemical properties, and infiltration and permeability measurements to determine wastewater infiltration rates. Some guidance for conducting these investigations and for interpreting soil test results is presented in this section and in Appendix C. It is expected that, in most cases, a qualified soil scientist will conduct the actual field work.

F.3.1 Detailed Soil Mapping

Boundaries of soil classification units on soil maps are, of course, only approximate representations of the arrangement of units as they actually occur. Because of practical limitations of map scale and the number of field observations, soil mapping unit boundaries generally contain only 65 to 85% of the designated mapping unit. This limitation holds even for the detailed maps of standard soil surveys. Thus, a delineated area may contain a significant portion of soil that has properties different from those in the designated soil unit, but these inclusions are present in bodies too small to be delineated on the map. Knowing the soil properties within a small area may be very important in some cases, such as those involving the location of rapid infiltration basins. Therefore, it is recommended practice to field check soil survey maps to confirm the accuracy of the information provided and to define and locate any significant inclusions of other soil types that might affect the design of the system, such as intermittent clay lenses that could adversely affect drainage and percolation. If soil survey maps of a prospective area are not available, complete detailed soil mapping will be necessary. Detailed soil mapping or field checks of existing maps should be conducted by an experienced soil scientist or under the direction of the local office of the SCS.

The steps involved in detailed soil mapping are described briefly here. The purpose of the descriptions is not to provide a complete guide to soil mapping, but to familarize the user with the basic procedures. Texts that provide more complete information include those by Buol et al. [5] and Soil Survey Manual, compiled by the U.S. Department of Agriculture (USDA Handbook No. 18) [6].

When preparing a detailed soil map, the first step is to assemble all available information on soil in the area of the prospective site. From this information, it should be possible to determine which soil series are likely to be encountered during the mapping. Complete descriptions of possible soils series should be obtained for comparison in the field.

The second step is to prepare a base map covering the prospective site. Aerial photographs projected to a scale in the range of 1:6 000 to 1:24 000 serve as convenient field sheets on which the locations of boring sites may be recorded. The USGS has ortho photo quads available of many areas at a standard scale of 1:24 000. These may be used as base maps to save the expense of producing aerial photos specifically for the area.

The third step is to locate soil examination sites on a grid system ranging in dimensions from 660 to 1 320 ft (200 to 430 m). Observations are also made at sites other than grid points as necessary to define boundaries between soil bodies.

The fourth step is to make field examinations of the soil. Soil profile examination consists of removing soil samples in 3 to 5 in. (8 to 13 cm) increments using a barrel auger, which provides a disturbed soil core about 2 to 4 in. (5 to 10 cm) in diameter. Each profile horizon should be described according to the following properties: soil color by comparison with standard color charts; soil texture estimated by rubbing moist soil samples between the fingers; soil consistency; soil pH; presence of lime; and presence of clay films. These data should be sufficient to associate a given soil with a soil series known to be in the area. If a soil does not correspond to a known series, more extensive description will be necessary. Observation pits are desirable in lieu of auger holes to permit better description and sampling of the profile. Pits are usually excavated to a depth of 6 ft (2 m) with a backhoe.

The final step is to analyze a sample from each defined soil horizon in a laboratory to determine pertinent soil properties. Thus, only a fraction of the number of samples taken in the field during the fourth step would be analyzed in the laboratory. The major properties normally reported are listed in Table F-2.

TABLE F-2

TYPICAL PROPERTIES DETERMINED WHEN FORMULATING SOIL MAPS

Physical and hydraulic properties
Particle size distribution
% moisture at saturation
Permeability
Chemical properties
pH
Cation exchange capacity
% base saturation
Conductivity
% exchangeable sodium
% organic carbon
% nitrogen

F.3.2 Soil Sampling

Sampling of the surface soils is often needed to define trace element deficiencies, sodic or saline conditions, and nutrient and organic content for fertility. Sampling procedures are often the limiting factor in the accuracy of a soil investigation because of the size of the sample relative to the area represented by the sample. A 1 lb (0.5 kg) sample is normally collected to represent an area of 5 to 40 acres (2 to 16 ha), and 6 in. (15 cm) in depth. The sample represents, at most, one-billionth of the actual soil volume.

There is no standard sampling procedure that can be applied to all situations, but some basic guidelines can be given. The first step in any sampling procedure is to subdivide the area into homogeneous units. The criteria for homogeneity are somewhat subjective and may include visual differences in the soil or crop, known differences in past management, or other factors. Uniform areas should be subdivided further into sampling units ranging in size from 5 to 40 acres (2 to 8 ha), depending on the area of uniformity [7].

The second step is to establish a pattern such as a grid to denote sampling points. In general, samples should be composites of several subsamples from the area to minimize the influence of micro-variations in soil properties caused by plants, animals, and fertilization. However, compositing is not advisable when sampling for salinity testing, because there may be large variations in soluble salts over very small areas [8]. The number of subsamples necessary to represent the sample area adequately varies with the degree of accuracy desired, the test to be conducted, the type of soil, and previous management. No universally accepted number has been defined, but a minimum of 10, and preferably

20, subsamples has been suggested as a guideline [9]. A grid pattern should be used if fields are bare or if the status of the entire area is to be represented. If the identification of problem areas is the objective of the testing, and vegetation is present, the distribution of vegetation and its appearance may indicate affected areas and thus serve as a guide in selecting sampling sites. When sampling affected areas, adjacent unaffected areas should be sampled for comparison. These kinds of data will aid in defining the cause of the problem in the affected area. Nonrepresentative spots, such as manure spots and fertilizer spills, should be avoided when sampling.

The third step is to collect samples of the soil. Samples should be collected with a tool that will take a small enough equal volume of soil from each sampling site so that the composite sample will be of an appropriate size to process. A composite sample volume of 1 to 2 pt (0.5 to 1 L) is normally adequate. The most important consideration in selecting a sampling tool is that it provide uniform cores or slices of equal volume to the depth desired. The tool should be easy to clean and adaptable to dry, sandy soil as well as to relatively moist, clayey soil. Shovels, trowels, soil tubes, and augers are commonly used. When sampling for micronutrients or trace metals, tools containing the metals of interest should not be used. The depth of sampling depends on the analyses to be conducted, the crop to be grown, and previous knowledge of the soil profile. If a soil survey of the area has been conducted, sampling of the subsoil probably is not necessary for macronutrient studies. Sampling of the soil to a depth of at least 5 ft (1.5 m) is necessary to determine textural discontinuities or compacted layers that affect water penetration and root growth.

The final step is to preserve and transport the samples to the laboratory. Samples should be handled in a manner that will minimize any changes in properties. Samples for analysis of constituents subject to rapid changes, such as nitrogen, should be frozen or dried rapidly at low temperatures. Waterproof bags or containers should be used for salinity or boron samples to avoid salt absorption from moist samples. Any necessary mixing, splitting, and subdividing of the sample for individual tests should be done in the laboratory.

F.3.3 Soil Testing

The soils at prospective land treatment sites should be identified in terms of their physical, hydraulic, and chemical properties. The significance of physical and chemical soil properties as they relate to design and operation of land application systems is discussed in this section. Procedures for determining soil properties are also described; however, it is not the intent to provide complete analytical procedures. Basic concepts of soil properties are presented to provide the user with enough background information to make meaningful interpretation of test results. In addition, the ranges of the various soil properties that are normally encountered are described. Criteria for the selection of the type of land treatment process based on soil properties are presented and the effects of soil properties on system design and operation are discussed.

F.3.3.1 Physical Properties

Physical properties of soils that have important effects on system design and operation include texture, structure, bulk density, and porosity. The primary importance of these physical properties with regard to land treatment process design is their effect on soil hydraulic properties.

F.3.3.1.1 Texture

Determining soil textural class involves measuring the relative number of particles in the various particle size groups--gravel, sand, silt, and clay. These particle size groups are defined on the basis of a USDA scale, as shown previously in Figure 3-7. Once the particle size distribution is known, the textural name of the soil can be determined by the relative percentage of each group, as shown in the textural triangle (Figure 3-7). Terms commonly used to describe soil texture and their relationships to textural classes were listed in Table 3-6.

The textural group name is prefaced by "gravelly" when 20 to 50% of the material is of gravel size or "very gravelly" when more than 50% of the soil is of gravel size (2 to 76 mm diameter). The same proportions apply to coarser material such as cobbles (76 to 250 mm diameter) or stones (>250 mm diameter).

The most commonly used methods for determining size distribution of soil particles are sedimentation methods, in which large particles are classified by sieving and the settling rates of dispersed particles in viscous fluid are measured. The measured settling rates are related to particle size through Stokes' law. Various techniques are described in Taylor and Ashcroft [10]. When unknown soil series are present, particle size distribution is determined as part of a detailed soil mapping program.

F.3.3.1.2 Structure

Structure refers to the aggregation of individual soil particles into larger units (aggregates) with planes of weakness between them. The aggregates have properties unlike an equal mass of unaggregated primary soil particles. Soils that do not have aggregates with natural boundaries are considered to be structureless. Two forms of structureless conditions are recognized--single grain and massive. Single grain condition refers to soils (normally loose sands) in which primary soil particles do not adhere to one another and are easily distinguishable. Massive condition refers to soils in which primary soil particles adhere closely to one another but the mass lacks planes of weakness.

Although soil structure is described in terms of size and shape of aggregates for purposes of classification, other factors associated with structure are more important from the standpoint of soil hydraulic properties and soil-plant relations. These factors include (1) the pore size distribution that results from aggregation; (2) the stability or resistance to disintegration of aggregates when wet and their ability to re-form on drying; and (3) the hardness of the aggregates. Organic matter in wastewaters often improves soil structure by serving as a binding agent for soil granules.

F.3.3.1.3 Bulk Density

Bulk density (ρ_b) of a soil is defined as the oven dry mass of soil per unit volume of undisturbed soil (e.g., g/cm^3). The volume thus includes void (air plus water) volume as well as particle volume. The bulk density of mineral soils can range from about 0.8 g/cm^3 for recently tilled soils to about 1.9 g/cm^3 for highly compacted soils. Plant growth ceases above a bulk density of 1.6 to 1.7 g/cm^3 due to mechanical impedance of root extension. Organic soils can have bulk density values as small as 0.2 g/cm^3. Bulk density is not a constant property, but tends to decrease as clay particles swell on wetting and to increase as the particles shrink on drying. When field measurements are not available, a commonly assumed value for bulk density of mineral soils is 1.33 g/cm^3.

Two basic methods are used to measure bulk density--gravimetric and gamma ray detection. The gravimetric method requires that an undisturbed soil core be obtained using a special core sampler. The core is dried to a constant mass, and the dry bulk density is determined by dividing the mass by the core volume. The gamma ray detection method involves the use of instruments, including a separate source probe and detector. The probe can be either placed on the surface to measure surface density or lowered into an access tube for depth measurements. The gamma ray equipment measures wet bulk density. Thus, the water content must be measured to calculate dry bulk density. This is normally done using a neutron moisture probe [10].

Bulk density is an important property because it is used to convert concentrations of soil constituents expressed on a weight or mass basis to concentrations expressed on a volume, area, or depth basis. For instance, soil moisture content of an incremental depth of the soil profile, when measured gravimetrically, is expressed in terms of g water/g soil. This value (θ_m) may be converted to volumetric water content (θ_v gm/cm^3) by multiplying by the bulk density. In irrigation work, θ_v is often expressed as an equivalent depth of water in an incremental depth of soil per unit area of soil (i.e., cm of water). A similar conversion is necessary for concentrations of other soil constituents such as nutrients and metals that are measured in the laboratory on a mass basis.

F.3.3.1.4 Porosity

Porosity is a measure of the total void space in a soil profile. If the soil bulk density (ρ_b) and the soil particle density (ρ_p) are known, porosity (E) may be calculated from the following equation.

$$E = 1 - \frac{\rho_b}{\rho_p} \qquad \text{(F-1)}$$

The average particle density (ρ_p) of most mineral soils is about 2.65 g/cm^3.

Porosity is of interest because it affects hydraulic properties of aquifers. These effects are discussed in Appendix C.

F.3.3.2 Chemical Properties

The chemical composition of the soil is the major factor affecting plant growth and a significant determining factor in the capacity of the soil to renovate wastewater. Thus, chemical properties should be determined prior to design to evaluate the capability of the soil to support plant growth and to renovate wastewater and should be monitored during operation to avoid detrimental changes in soil chemistry.

Because of the variable nature of soil, few standard procedures for chemical analysis of soil have been developed. Several references that describe analytical methods are available [11, 12, 13]. A complete discussion of analytical methods and interpretation of results for the purpose of evaluating the soil nutrient status is presented in Walsh and Beaton [14].

Important chemical soil properties affecting the design and operation of land treatment systems include: pH, cation exchange capacity, percent base saturation, exchangeable sodium percentage, salinity, plant nutrients, phosphorus adsorption, and trace elements. The significance of these properties as they relate to design and operation of systems is discussed. Methods of determination of chemical properties are also presented along with guidelines for the interpretation of test results.

F.3.3.2.1 pH

Soil pH is a very useful parameter because it is easy to determine and tells a great deal about the character of the soil. Soil pH may be determined in the field using portable pH meters with glass electrodes. Meters with electrode assemblies are used almost exclusively for laboratory determinations.

Soil is prepared for pH determination by making a soil-water paste. Various soil-water ratios are commonly used, including 1:1, 1:2, 1:5, 1:10, and a saturated paste. The use of a dilute salt solution has been suggested as a containing solution because the salt will mask small differences in the salt concentration of the soil solution that affect pH readings. A degree of accuracy of +0.2 pH is the best that can be expected from any method [15]. When interpreting or using pH data, it is important to know which method was used because of the influence of the procedure on test results.

Soils having a pH below 5.5 contain exchangeable aluminum and manganese ions; in soils having a pH below 5.0, these ions increase sharply in concentration. The aluminum ion is very toxic to plants, primarily affecting the roots. The manganese ion is less toxic and affects both plant tops and roots. At low pH, the other major soil cations (calcium, magnesium, potassium, and sodium) are comparatively low. Deficiencies of these and other plant nutrients, particularly phosphorus, may result from low pH conditions. Soils with a pH above 5.5 do not have appreciable exchangeable aluminum. Soils with a pH between 7.8 and 8.2 are likely to contain calcium carbonate, and soils with a pH above 8.5 are likely to contain sodium carbonate. High acid or alkali conditions can render a soil sterile and destroy soil structure. A summary of the effects of various pH ranges on crops is presented in Table F-3.

TABLE F-3

EFFECTS OF VARIOUS pH RANGES ON CROPS

pH range	Effect
Below 4.2	Too acid for most crops
4.2-5.5	Suitable for acid-tolerant crops
5.5-8.4	Suitable for most crops
Above 8.4	Too alkaline for most crops (indicates a probable sodium problem)

Acid soil conditions (low pH) can be corrected in many cases by the addition of calcium carbonate (lime) to the soil. Alkaline soil conditions (high pH) can be corrected by the addition of acidifying agents.

F.3.3.2.2 Cation Exchange Capacity

The cation exchange capacity (CEC) is the quantity of exchangeable cations that a soil is able to adsorb. The adsorption occurs as a result of the attraction of the positively charged cations by negative charges that exist on the surface of clay minerals, hydrous aluminum and iron oxides, and organic matter. The major cations held on the exchange include calcium, magnesium, potassium, sodium, aluminum, hydrogen, and ammonium. Also involved in ion exchange to a small extent are micronutrients such manganese, iron, and zinc. The CEC is a measure of the chemical reactivity of the soil and is generally an indication of the effectiveness of the soil in adsorbing cationic contaminants from wastewater such as the heavy metals.

It is apparent from the nature of the exchange sites in soil that soils with large amounts of clay and organic matter will have higher exchange capacities than sandy soils low in organic matter. It should be noted, however, that the negative charges associated with the hydrous metal oxides and organic matter are the result of hydrogen ion dissociation from OH and COOH groups and are thus pH-dependent. Therefore, the measured CEC will vary with the pH of the solution used to run the test. The CEC will increase with pH in direct proportion to the amount of pH-dependent charged particles in the soil.

Several methods are used to measure CEC. All of them involve displacement of the adsorbed cations from the exchange sites by a concentrated salt solution and analysis of the extract for the displaced cations. The important difference in the methods is the pH at which the soil solution is held. When comparing or using data on CEC, it is important to be aware of how the test was conducted. When dealing with acid soils (pH <6.5), it is recommended that the CEC be calculated from the sum of the exchangeable cations (Al^{3+}, Mn^{2+}, Na^+, K^+, Ca^{2+}, Mg^{2+}) determined from individual analysis to avoid the effects of pH dependent charge. Determination of the individual exchangeable cations will allow calculation of percent base saturation and exchangeable sodium percentage as described in the following sections. A discussion of CEC determinations is presented in Tisdale and Nelson [16].

Although CEC generally increases with clay content, the actual value depends largely on the type of clay mineral present. The CEC of expanding clays such as montmorillonite and vermiculite is 5 to 10 times greater than nonexpanding clays such as illite and kaolinite. However, as mentioned previously, the CEC of a particular type of soil is not a fixed property, but varies directly with soil pH. Typical ranges of CEC for various soil types are given in Table F-4.

TABLE F-4

TYPICAL RANGES OF CATION EXCHANGE CAPACITY OF VARIOUS TYPES OF SOILS

Soil type	Range of CEC, meq/100 g
Sandy soils	1 to 10
Silt loams	12 to 20
Clay and organic soils	Over 20

F.3.3.2.3 Percent Base Saturation

The percentage of total CEC occupied by calcium, magnesium, potassium, and sodium is an important property known as percent base saturation. The term is a misnomer because these cations are not actually basic, but the term is still commonly used. In general, the availability of the basic cations to plants increases with the degree of base saturation. The pH also increases as percent base saturation increases. A satisfactory balance of exchangeable cations occupying the CEC is given in Table F-5. If sodium occupies 10% or more of the exchange capacity sites, it can cause significant permeability problems in fine-textured soils.

TABLE F-5

SATISFACTORY BALANCE OF EXCHANGEABLE CATIONS OCCUPYING THE CATION EXCHANGE CAPACITY

Cation	% of total
Calcium	60 to 70
Magnesium	20 to 35
Potassium	5 to 10
Sodium	Under 5

F.3.3.2.4 Exchangeable Sodium Percentage

Soils containing excessive exchangeable sodium are termed "sodic" soils. In the past, the terms "alkali" and "black alkali" were used. A soil is considered sodic when the percentage of the total CEC occupied by sodium, the exchangeable sodium percentage (ESP), exceeds 15%. These levels of sodium cause clay particles to disperse in the soil because of the chemical nature of the sodium ion. The dispersed clay particles cause low soil permeability, poor soil aeration, and difficulty in seedling emergence. The level of ESP at which these problems are encountered depends on the soil texture. Fine-textured soil may be affected at an ESP above 10%, but coarse-textured soil may not be damaged until the ESP reaches about 20%.

Sodic conditions may be corrected by addition of chemicals containing soluble calcium to displace the sodium followed by leaching of the sodium from the profile. Use of amendments to correct sodic conditions is covered in Section 5.7.2.

The procedure for the direct analysis of ESP may be found in references [11, 12, 13]. The ESP may also be determined with a considerable degree of reliability using an indirect method. This involves the analysis of the saturation extract of the soil for calcium, magnesium, and sodium; calculation of the SAR, as defined in the following discussion; and determination of the ESP using the nomograph presented as Figure F-1.

The degree to which sodium will be adsorbed by a soil from water when brought into equilibrium with it can be estimated by the SAR:

$$SAR = \frac{Na^{+}}{\sqrt{(Ca^{++} + Mg^{++})/2}} \qquad (F\text{-}2)$$

where sodium (Na), calcium (Ca), and magnesium (Mg) concentrations are in milliequivalents per litre.

A modified SAR equation developed by the U.S. Salinity Laboratory can be used to adjust the calculated SAR for the added effects of (1) the precipitation or dissolution of calcium in soils, and (2) the content of carbonate (CO_3) and bicarbonate (HCO_3) alkalinity in the water. The adjusted SAR formula and required factors are presented in Table F-6.

Adjusted SAR values of water exceeding 9.0 can cause permeability problems in clay-type soils. High SAR is more damaging to shrinking-swelling clay soils (montmorillonite) than to nonswelling types (illite-vermiculite and kaolinite) [17]. Permeability problems related to a high SAR of irrigation water can be corrected by the addition of gypsum followed by leaching.

FIGURE F-1

NOMOGRAPH FOR DETERMINING THE SAR VALUE OF IRRIGATION WATER AND FOR ESTIMATING THE CORRESPONDING ESP VALUE OF A SOIL THAT IS AT EQUILIBRIUM WITH THE WATER [13]

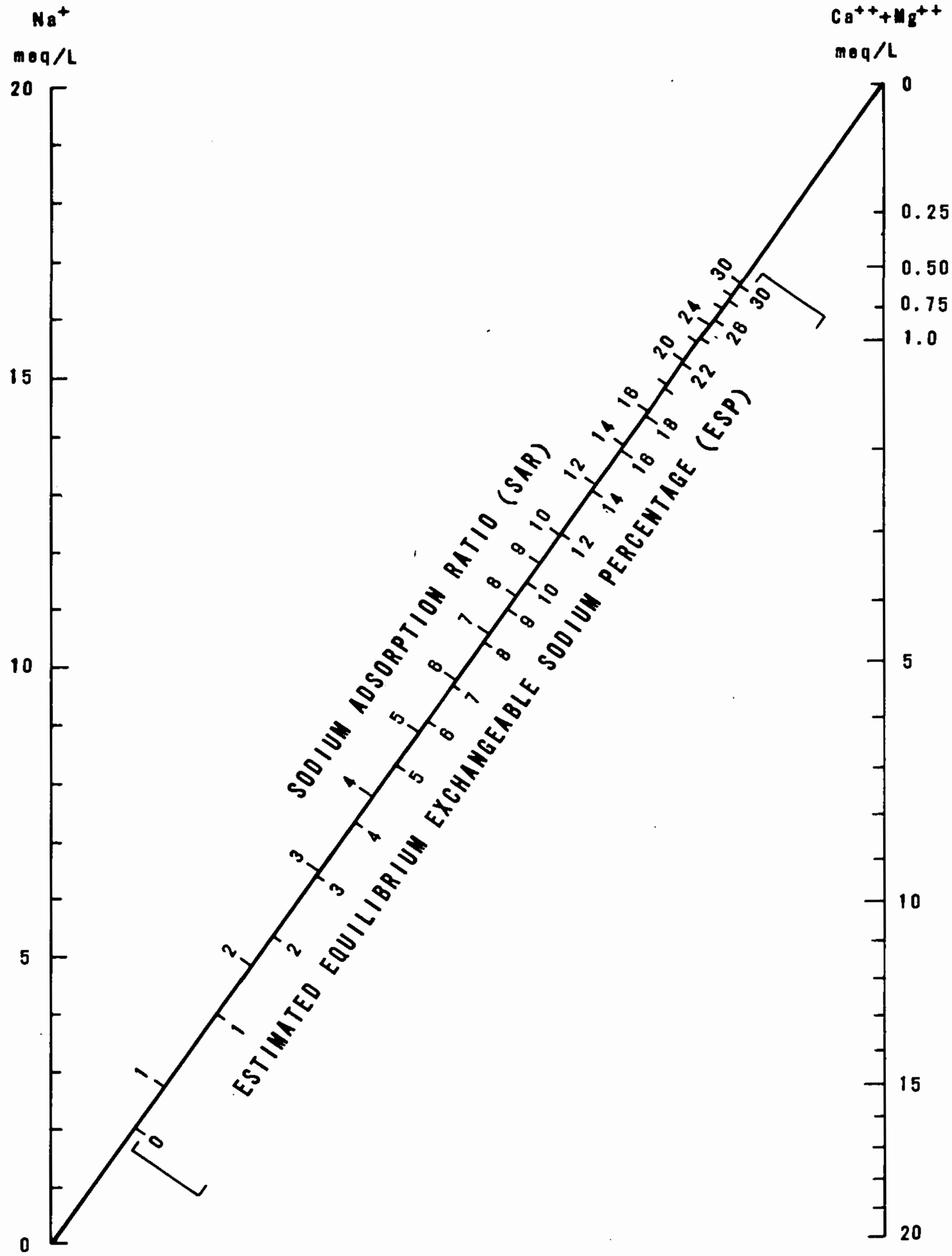

TABLE F-6

FACTORS FOR COMPUTING THE ADJUSTED SAR [17]

Concentration Ca+Mg+Na meq/L	Column 1 $p(K_2'-K_2')$	Concentration Ca+Mg meq/L	Column 2 p(Ca+Mg)	Concentration CO_3+HCO_3 meq/L	Column 3 pAlk
0.5 →	2.11	0.05 →	4.60	0.05 →	4.30
0.7	2.12	0.10	4.30	0.10	4.00
0.9	2.13	0.15	4.12	0.15	3.82
1.2	2.14	0.2	4.00	0.20	3.70
1.6	2.15	0.25	3.90	0.25	3.60
1.9	2.16	0.32	3.80	0.31	3.51
2.4	2.17	0.39	3.70	0.40	3.40
2.8	2.18	0.50	3.60	0.50	3.30
3.3	2.19	0.63	3.50	0.63	3.20
3.9	2.20	0.79	3.40	0.79	3.10
4.5	2.21	1.00	3.30	0.99	3.00
5.1	2.22	1.25	3.20	1.25	2.90
5.8	2.23	1.58	3.10	1.57	2.80
6.6	2.24	1.98	3.00	1.98	2.70
7.4	2.25	2.49	2.90	2.49	2.60
8.3	2.26	3.14	2.80	3.13	2.50
9.2	2.27	3.90	2.70	4.0	2.40
11	2.28	4.97	2.60	5.0	2.30
13	2.30	6.30	2.50	6.3	2.20
15	2.32	7.90	2.40	7.9	2.10
18	2.34	10.00	2.30	9.9	2.00
22	2.36	12.50	2.20	12.5	1.90
25	2.38	15.80	2.10	15.7	1.80
29	2.40	19.80	2.00	19.8	1.70
34	2.42				
39	2.44				
45	2.46				
51	2.48				
59	2.50				
67	2.52				
76	2.54				

$$\text{adj SAR} = \frac{Na}{\sqrt{\frac{Ca+Mg}{2}}}\left[1 + (8.4 - pHc)\right]$$

$$pHc = (pK_2'-pK_2') + p(Ca+MG) + pAlk$$

$$= \text{Column 1} + \text{Column 2} + \text{Column 3}$$

F.3.3.2.5 Salinity

Soluble salts accumulate in the root zone of soils when leaching is inadequate to move them deeper into the soil profile. Inadequate leaching may be due to low rainfall in natural soils, insufficient irrigation of irrigated soils, or poor drainage conditions. In arid regions where annual evaporation is substantially in excess of precipitation, salts will accumulate in nearly all soils unless leaching is practiced.

The salinity level of a soil is usually measured on the basis of the electrical conductivity (EC_e) of an extract solution from a saturated soil. The procedure for analysis involves preparation of a saturated paste, followed by vacuum extraction and determination of the EC_e as described in the standard references [11, 12, 13]. Saline soils are defined as those yielding an EC_e value greater than 4 000 micromhos/cm at 25°C. Soils exhibiting both saline and sodic conditions are referred to as saline-sodic soils.

Salts in the soil solution will restrict crop growth at various concentrations depending on the plant. Approximate EC_e ranges at which crop growth is affected are given in Table F-7 for different levels of crop salt-sensitivity. The salt tolerance levels of individual field and forage crops are presented in Section 5.6. Leaching requirements to maintain an EC_e level in the root zone suitable for full growth are also discussed in Section 5.6.

TABLE F-7

SALINITY LEVELS AT WHICH CROP GROWTH IS RESTRICTED

Salinity range, EC_e, micromhos/cm at 25°C	Effect
<2 000	No salinity problems
2 000-4 000	Restricts growth of very salt-sensitive crops
4 000-8 000	Restricts growth of many crops
8 000-16 000	Restricts growth of all but salt-tolerant crops
>16 000	Only a few salt-tolerant crops produce satisfactory yields

Saline soils may be reclaimed by leaching; however, management of the leachate is often required to protect groundwater quality. The U.S. Department of Agriculture's Handbook 60 [13] deals with the diagnosis and improvement of such soils for agricultural purposes. This reference can be used as a practical guide for managing saline and saline-sodic soil conditions, especially in arid and semiarid regions.

F.3.3.2.6 Plant Nutrients

The essential plant nutrients in addition to carbon, hydrogen, and oxygen, include nitrogen (N), phosphorus (P), potassium (K), sulfur (S), magnesium (Mg), calcium (Ca), iron (Fe), zinc (Zn), copper (Cu), manganese (Mn), molybdenum (Mo), chloride (Cl), and boron (B). Sodium (Na) and cobalt (Co) are necessary for some plants. The elements N, P, K, S, Mg, and Ca are designated as macronutrients because they are required in relatively larger quantities. The remaining elements are referred to as micronutrients. Deficiencies in any will adversely affect plant growth. The amount of each nutrient in the soil that is considered adequate for plant growth depends not only on the plant but also on the test method used to measure the nutrient.

The objectives of soil testing for plant nutrients are to assess the fertility status of a soil and to develop a fertilizer recommendation for production of a particular crop. In most cases, the testing involves N, P, and K since deficiencies in these nutrients are most common. Deficiencies of S, Zn, Fe, and B occur in a few soils, but deficiencies of other nutrients (Mo, Cu, Co, Ca, Mg, Mn, Na, and Cl) are relatively rare.

On the basis of commonly used test methods, the University of California Agricultural Extension Service has developed a summary of adequate levels of the more deficient nutrients for some selected crops. This summary is presented in Table F-8. Critical values for nitrogen are not included because there are no well accepted methods for determining available N. This is due primarily to the fact that N availability depends on decomposition of organic matter which is affected by temperature and moisture conditions, hence seasonal variations in N may be large. The chemistry of N in the soil is discussed in detail in Appendix A. Prior to the design of systems in which crop production is considered, it is recommended that the fertility status of the soil be evaluated.

A soil testing program to assess soil fertility should be conducted by a reputable commercial laboratory, preferably one that offers a complete service of sampling, testing, interpretation, and fertilizer recommendations. All of the analytic procedures involve extraction of the nutrient with water, a salt solution, an acid solution, or a chelating agent. Selection of the procedure is an important decision because there must be a known relationship or correlation between the test result and crop response if the test is to provide meaningful data. Test data in themselves are not worthwhile unless they are interpreted correctly. Selection of the most suitable test procedure and interpretation of test results must be based on a good deal of background information, such as the significant chemical forms of the available nutrients in the soils in question, the relative productive capacity of the particular soil for the various crops, and the response of different

TABLE F-8

APPROXIMATE CRITICAL LEVELS OF NUTRIENTS
FOR SELECTED CROPS IN CALIFORNIA

Nutrient	Approximate critical range, ppm	Test method
Phosphorus		
Range and pasture	10	0.5 M $NaHCO_3$ extraction at pH 8.5
Field crops and warm season vegetables	5-9	
Cool season vegetables	12-20	
Potassium		
Grain and alfalfa	45-55	1.0 N ammonium acetate extraction at pH 7.0
Cotton	55-65	
Potatoes	90-110	
Zinc	0.4-0.6	DPTA extraction

crops to various rates and methods of fertilization. A complete discussion of these and other factors is not within the scope of this manual, but may be found in Tisdale and Nelson [16] and in Walsh and Beaton [14]. Plant tissue testing may be used to diagnose plant nutrient deficiencies or toxicities. The information from plant testing is often obtained only after it is too late to take corrective action. Information on the use of plant testing may be found in reference [14].

F.3.3.2.7 Phosphorus Adsorption

For rapid infiltration or slow rate systems where phosphorus removal is important, the phosphorus adsorption test can be conducted. To conduct an adsorption test, about 10 g of soil is placed in containers containing known concentrations of phosphorus in solution. After periodic shaking, the solution is analyzed for phosphorus and the difference in concentrations from the initial is attributed to adsorption. Procedures are presented by Enfield and Bledsoe [18].

Tofflemire and Chen reported on 5 day phosphorus adsorption in sandy soils and found that the range was from 2.8 to 278 mg/100 g of phosphorus with an average of 38 [19]. Enfield and Bledsoe conducted adsorption tests of up to 4 months and found that the 4 month retention was 1.5 to 3.0 times the 5 day retention [18]. Tofflemire and Chen concluded that total phosphate retention in an actual system will be at least 2 to 5 times the estimate based on the 5 day adsorption test.

F.3.3.2.8 Trace Elements

A few plant nutrients can reach levels in the soil that are toxic to plants (phytotoxic). Those of concern include B, Zn, Cu, and Mn. There are also alien or nonnutrient contaminants present in wastewater that will accumulate in soils and can be phytotoxic or toxic to consumers of plants containing the elements. The major elements include arsenic (As) and the metals cadmium (Cd), lead (Pb), nickel (Ni), mercury (Hg), and chromium (Cr).

Excess B occurs in scattered areas in arid and semiarid regions. It is frequently associated with saline soils but most often results from use of high-boron irrigation waters. As in the case of soluble salts, B concentrations in soils may vary greatly over short distances. Thus, similar sampling precautions should be observed. Toxic levels of B for plants of varying sensitivity have been well established. Critical levels of B in the saturated extract are given in Table F-9. A list of crops according to their B tolerance is presented in Table 5-34.

TABLE F-9

CRITICAL PHYTOTOXIC LEVELS
OF BORON IN SOILS

Value in saturation extract, ppm	Effects on crops
Below 0.5	Satisfactory for all crops
0.5 to 1	Sensitive crops may show visible injury
1 to 5	Semitolerant crops may show visible injury
5 to 10	Tolerant crops may show visible injury

Two procedures can be used when analyzing soils for metals--partial extraction and full decomposition. Partial extraction measures metal content that is available for uptake by plants. The most promising method of this type is extraction with the chelating agent DPTA followed by atomic absorption determination of the metals in the extract. This test is described by Viets and Lindsay [20] and by Brown and DeBoer [21].

While extraction methods are useful for assessing fertility status, they have distinct disadvantages when used to monitor changes in soil composition resulting from wastewater application. Total decomposition and solubilization of all metal by hot acid digestion is the preferred method for monitoring purposes, primarily because it will probably yield the most reproducible final results. However, total metal analysis is time consuming and expensive. A compromise method is extraction with hot, concentrated acid. A further discussion of the relative merits of these methods is presented in Appendix E.

An important point about extraction methods for metal analysis, particularly when developing data for comparisons, is that the analytical procedures must be exactly the same in every detail each time the test is conducted. Comparison of data developed using different extraction methods or solutions can be misleading.

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APPENDIX G

GLOSSARY OF TERMS
CONVERSION FACTORS
AUTHORS INDEX
SUBJECT INDEX

GLOSSARY OF TERMS

acre-foot--A liquid measure of a volume equal to covering a 1 acre area to 1 foot of depth.

aerosol--A suspension of colloidal solid or liquid particles in air or gas, having small diameters ranging from 0.01 to 50 microns.

aquiclude--A geologic formation which, although porous and capable of absorbing water slowly, will not transmit it rapidly enough to furnish an appreciable supply for a well or spring.

available moisture--The part of the water in the soil that can be taken up by plants at rates significant to their growth; the moisture content of the soil in excess of the ultimate wilting point.

available nutrient--That portion of any element or compound in the soil that can be readily absorbed and assimilated by growing plants. ("available" should not be confused with exchangeable.)

evapotranspiration--The combined loss of water from a given area and during a specified period of time, by evaporation from the soil surface, snow, or intercepted precipitation, and by the transpiration and building of tissue by plants.

field area--The "wetted area" where treatment occurs in a land application system.

field capacity--(field moisture capacity)--The moisture content of soil in the field 2 or 3 days after having been saturated and after free drainage has practically ceased; the quantity of water held in a soil by capillary action after the gravitational or free water has been allowed to drain; expressed as moisture percentage, dry weight basis.

fragipan--A loamy, dense, brittle subsurface horizon that is very low in organic matter and clay but is rich in silt or very fine sand. The layer is seemingly cemented and slowly or very slowly permeable.

horizon (soil)--A layer of soil, approximately parallel to the soil surface, with distinct characteristics produced by soil-forming processes.

infiltrometer--A device by which the rate and amount of water infiltration into the soil is determined (cylinder, sprinkler, or basin flooding).

lysimeter--A device for measuring percolating and leaching losses from a column of soil. Also a device for collecting soil water in the field.

micronutrient--A chemical element necessary in only small trace amounts (less than 1 mg/L) for microorganism and plant growth. Essential micronutrients are boron, chloride, copper, iron, manganese, molybdenum, and zinc.

mineralization--The conversion of a compound from an organic form to an inorganic form as a result of microbial decomposition.

sodic soil--A soil that contains sufficient sodium to interfere with the growth of most crop plants, and in which the exchangeable sodium percentage is 15 or more.

soil water--That water present in the soil pores in an unsaturated (aeration) zone above the groundwater table. Such water may either be lost by evapotranspiration or percolation to the groundwater table.

tensiometer--A devise used to measure the negative pressure (or tension) with which water is held in the soil; a porous, permeable ceramic cup connected through a tube to a manometer or vacuum gage.

till--Deposits of glacial drift laid down in place as the glacier melts, consisting of a heterogeneous mass of rock flour, clay, sand, pebbles, cobbles, and boulders intermingled in any proportion; the agricultural cultivation of fields.

tilth--The physical condition of a soil as related to its ease of cultivation. Good tilth is associated with high noncapillary porosity and stable, granular structure, and low impedance to seedling emergence and root penetration.

transpiration--The net quantity of water absorbed through plant roots that is used directly in building plant tissue, or given off to the atmosphere as a vapor from the leaves and stems of living plants.

volatilization--The evaporation or changing of a substance from liquid to vapor.

wilting point--The minimum quantity of water in a given soil necessary to maintain plant growth. When the quantity of moisture falls below this, the leaves begin to drop and shrivel up.

CONVERSION FACTORS
U.S. Customary to SI (Metric)

U.S. customary unit			SI	
Name	Abbreviation	Multiplier	Symbol	Name
acre	acre	0.405	ha	hectare
acre-foot	acre-ft	1 234	m^3	cubic metre
cubic foot	ft^3	28.32 0.0283	L m^3	litre cubic metre
cubic feet per second	ft^3/s	28.32	L/s	litres per second
degrees Fahrenheit	°F	0.555(°F-32)	°C	degrees Celsius
feet per second	ft/s	0.305	m/s	metres per second
foot (feet)	ft	0.305	m	metre(s)
gallon(s)	gal	3.785	L	litre(s)
gallons per acre per day	gal/acre·d	9.353	L/ha·d	litres per hectare per day
gallons per day	gal/d	4.381×10^{-5}	L/s	litres per second
gallons per minute	gal/min	0.0631	L/s	litres per second
horsepower	hp	0.746	kW	kilowatt
inch(es)	in.	2.54	cm	centimetre(s)
inches per hour	in./h	2.54	cm/h	centimetres per hour
mile	mi	1.609	km	kilometre
miles per hour	mi/h	0.45	m/s	metres per second
million gallons	Mgal	3.785 3 785.0	ML m^3	megalitres (litres $\times 10^6$) cubic metres
million gallons per acre	Mgal/acre	8 353	m^3/ha	cubic metres per hectare
million gallons per day	Mgal/d	43.8 0.044	L/s m^3/s	litres per second cubic metres per second
parts per million	ppm	1.0	mg/L	milligrams per litre
pound(s)	lb	0.454 453.6	kg g	kilogram(s) gram(s)
pounds per acre per day	lb/acre·d	1.12	kg/ha·d	kilograms per hectare per day
pounds per square inch	lb/in.^{2}	0.069 0.69	kg/cm^2 N/cm^2	kilograms per square centimetre Newtons per square centimetre
square foot	ft^2	0.0929	m^2	square metre
square inch	in.^{2}	6.452	cm^2	square centimetre
square mile	mi^2	2.590 259.0	km^2 ha	square kilometre hectare
ton (short)	ton (short)	0.907	Mg (or t)	megagram (metric tonne)
tons per acre	tons/acre	2 240 2.24	kg/ha Mg/ha	kilograms per hectare megagrams per hectare

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