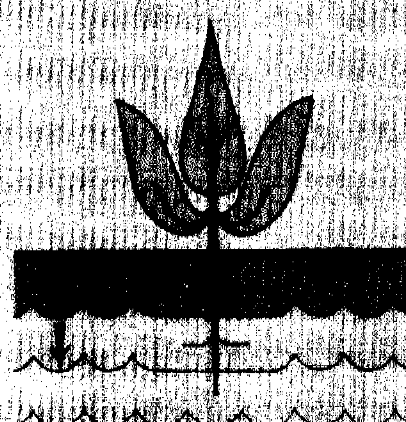
State of Knowledge in LAND TREATMENT OF WASTEWATER



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State of Knowledge in
LAND TREATMENT OF WASTEWATER

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H.L. McKim
Symposium Coordinator

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PREFACE

Many countries have been applying wastewater to the soil for centuries. In almost all instances the nutrients in the effluent from agricultural and domestic sources have been used for their fertilizer value. This alone can make the concept of land treatment of wastewater a cost-effective alternative to other forms of treatment of these waste products. Agricultural crops produced at land treatment sites are of high quality and can be used as an animal food source. When wastewater is applied to various forest species, yields may increase by a factor of two. This increase in yield may provide an alternative energy source to replace diminishing oil, gas and coal supplies.

In the United States several land treatment systems have been in operation since the turn of the century. In the past ten years, however, Federal legislation has not only given municipalities added incentive to utilize land treatment but it has also mandated that land treatment be one of the treatment alternatives that is considered prior to applying for federal construction grants. If land treatment is the selected method its percolating water must meet prescribed water quality standards and it must be the most cost-effective method for treating the wastewater.

The objectives of this Symposium are to summarize the state of knowledge of the practical aspects of the treatment of wastewater by land application and to identify the suitable approaches for the design of such land treatment systems. The topics to be included are: site selection considerations, case studies of national and international concern, health effects of land treatment systems, pretreatment considerations, uses of wastewaters in agricultural and forest systems, monitoring, modeling and design criteria.

The Symposium Proceedings are published in two volumes. Volume I contains the invited papers presented and discussed at the conference. Volume 2 contains shorter papers about on-going research that were selected from the responses received following a call for abstracts.

The program committee wishes to acknowledge Mr. Noel Urban, Chief, Urban Studies and Management Section, Engineering Division, OCE for his support of the Symposium. Special thanks is offered to Mr. Ben Yamashita and Ms. Mary Ann Austin of the Public Affairs Office, Mr. Stephen L. Bowen and Mrs. Donna R. Murphy of the Publications Section and Mrs. Donna Gerow of the Word Processing Center for their assistance in preparation and layout of the proceedings. In addition, the Symposium coordinator is particularly indebted to the program committee for their assistance in coordination and review of the proceedings, to Mrs. Sandra Messersmith who helped in the overall planning of the Symposium, to Ms. Nancy Dumont who arranged for the Russian translations, and to CRREL management for their support and cooperation in hosting the Symposium.

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CONTENTS

OVERVIEW

Land Treatment: Achieving Water Quality through Effective Recycling of Wastewater

John T. Rhett, Office of
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The United States Environmental Protection Agency (US EPA) manages an active program to use land treatment as an effective technology to recycle nutrients while improving water quality. This program for recycling wastewater nutrients is a rapidly growing program with great promise for the future. A rebirth of research interest some 20 years ago has had results of much greater impact than envisioned at the time. The results of this modest research effort were given strong support by the environmental movement. The impact of the environmental movement led to sweeping changes in Federal legislation that have brought land treatment to the forefront as an energy conserving technology which recycles nutrients while abating water pollution. The confidence with which people view land treatment is advancing rapidly through many avenues for information exchange. The US EPA program for using land treatment is reflected in this rapid gain in confidence. Land treatment may become a standard for recycling technology in the near future.

National management of the US EPA program for improving wastewater treatment is the major function of my office. Recycling and reclamation of wastewater by land treatment is a vital part of that program and my interest in this symposium is as intense as that of Mr. Thomas C. Jorling, EPA's Assistant Administrator for the Office of Water and Hazardous Materials. It is a coincidence that Mr. Jorling is delivering a talk on nutrient recycling by land treatment at a meeting underway in the USSR at this very time.

I know that Administrator Costle and Mr. Jorling will be extremely interested in the results of this conference because both are deeply committed to recycling and reclamation of nutrients by land treatment.

INTRODUCTION

In keeping with the symposium title "State of Knowledge in Land Treatment of Wastewater," I shall direct my remarks to how EPA, in operating an environmental construction program, views the state-of-the-art for land treatment. It has been my experience that the researcher, the regulator, and the user may diverge considerably in their assessment of the same data base. I think my point of view will stimulate some thought provoking questions in your minds today and I hope, generate some answers during the later presentations.

Given what we perceive as the "state of knowledge" of land treatment, I will be reviewing the symposium to answer the question, "Land treatment technology -- how can we improve its use?" Our interpretation leads us to conclude that the data base is strong. The recent historical record of land treatment is characterized by success stories. While we can always improve our knowledge and there is always one more question, we can build land treatment systems with confidence by using the knowledge that is available today.

It is this philosophical position that spearheads the US EPA thrust to encourage use of land treatment as a wastewater management alternative that improves water quality, is cost competitive, saves energy, and recycles nutrients.

BACKGROUND

The re-establishment of interest in land treatment as a wastewater management alternative came early in the environmental movement. Modest research programs in both the Federal and private sectors and at several universities date back 15 to 20 years. These research programs served as the catalyst for the intense research over the last 5 to 10 years. They also served as the basis for the strong emphasis placed on land treatment as a recycling option in the Federal legislation on water pollution control. In 1976, John Freshman(1) speaking for the Staff Director of the U.S. Senate Committee handling water pollution legislation stated, "...maybe people should be having conferences entitled 'Receiving Water as a Waste Treatment Alternative' or 'Aerate, Chlorinate, and Dump as a Waste Treatment Alternative' with land treatment -- since it makes the most sense -- as the standard." Mr. Freshman's tone in this statement is consistent with the tone of U.S. Federal legislation passed in 1972 and strengthened in 1977.

Although there are some differences in the use and definition of terms, there are three generally accepted methods of land treatment. The slow rate process (also called crop irrigation) couples wastewater management with recycling of nutrients in crop production. Rapid infiltration (also known as infiltration/percolation) emphasizes water reclamation rather than direct nutrient recycling. The product water from rapid infiltration may be reused for crop production, returned to surface waters, or allowed to recharge groundwaters. Overland flow also emphasizes water reclamation. Unlike rapid infiltration, however, the product water from overland flow is almost always discharged directly to surface waters. The terminology outlined has developed in the United States over the last decade. It has become the standard for the US EPA program on land treatment.

THE US EPA PROGRAM

The US EPA program on land treatment includes research groups at Ada, Oklahoma, and Cincinnati, Ohio, as well as the Construction Grants Program which is active across the Nation. Others will give presentations on the accomplishments of the research groups so I will concentrate on the use of overall research results in support of the Construction Grants Program.

THE CONSTRUCTION PROGRAM

Land treatment became a major consideration in program implementation as a result of Federal legislation passed in 1972. This legislation identified land treatment as a recycling alternative which should be given serious consideration in the national program to end pollution of streams and lakes. This legislation made it mandatory to consider land treatment for all projects starting on July 1, 1974. This legislative mandate had been anticipated for several years and an intensive program to assess the state of knowledge for design and operation of systems was nearing completion. Several reports on the state-of-the-art for research had been completed(2)(3)(4) and my staff had conducted a survey of operating systems in the United States(5) as well as gathering case history information from around the world. The results of these efforts led to the conclusion that there were dependable design options available for implementing land treatment as a recycling alternative as proposed in the Federal legislation. Program guidance was issued in July of 1974 and the US EPA embarked on an active program to use proven land treatment technologies to recycle wastewaters as a part of the national pollution abatement program. Additional technical reports were then prepared through my office to deal with costs(6)(7) and to provide a basis for evaluating and analyzing preliminary system designs(8).

It was also recognized that research would continue to substantially improve our design knowledge and that we would need periodic updating of the program guidance. Numerous research efforts initiated early in the 1970's were scheduled for completion with expected publication of results by 1975. These included efforts of the US Corps of Engineers (US COE) - led by the CRREL staff, efforts of the US Department of Agriculture (USDA), and many other sources as well as our own US EPA research efforts led by the Ada, Oklahoma, research team. In order to consolidate the research findings and issue them as quickly as possible, the US EPA took the lead for preparation of a design manual on land treatment.

An interagency workgroup co-chaired by the US EPA and US COE took on this task of preparing a design manual to be available in the fall of 1977. The workgroup did accomplish its assigned task and the design manual was issued in October 1977 as a joint effort of the US EPA, the US COE, and the USDA. The first printing of the manual(9) issued jointly by the three Agencies is entitled "Process Design Manual for Land Treatment of Municipal Wastewater." It is available from the US EPA Environmental Research Information Center at Cincinnati, Ohio. This manual is now established as the basic resource for those who conceive, plan, and design land treatment systems in the United States. It will serve as the US EPA standard for implementing land treatment as an alternative in the national program to abate water pollution while recycling nutrients and conserving energy. There will be a continuing effort to improve and update this manual as research opens new horizons and resolves issues which remain controversial. As the manager of an operating program, I extend, for myself and EPA, sincere appreciation to the research community for a job well done.

In response to the availability of detailed design information, the emphasis by President Carter on cost-effective, water conserving wastewater treatment facilities and the encouragement of Congress to consider wastewater reclamation and recycling by land treatment processes, on October 3, 1977 EPA Administrator Douglas Costle issued an Agency policy statement on land treatment of municipal wastewater(10). This policy has served to reinforce the attention given to wastewater treatment processes which renovate and reuse wastewater as well as recycle its organic matter and nutrients in a beneficial manner. The policy requires an applicant for Federal construction grant funds to provide "complete justification for the rejection of land treatment" if a method that encourages water conservation, wastewater reclamation and reuse is not selected after examining the alternative wastewater treatment technologies available to that applicant.

A look at the statistics concerning the implementation of land application projects through the Construction Grants Program indicates that we have made reasonable progress, but that clearly more can be accomplished in this area. Our information shows that to date at least 350 projects involving various forms of land application have been funded through the grants program. This represents approximately 15% of the total number of our projects which progressed to the point that a treatment alternative has been selected. In terms of dollars, however, this represents less than 5% of the funds obligated through the program. The comparatively small dollar amount is, in part, due to the fact that many of these projects have not reached the actual construction stage (i.e., the point where the greatest expenditure of funds occurs) and that most of the projects are relatively small in terms of wastewater volume treated. In any event, we most certainly expect to do better in the future.

FUTURE DIRECTIONS FOR THE CONSTRUCTION PROGRAM

New Federal legislation passed in December 1977 gives even greater impetus to land treatment as a recycling and reclamation alternative in the United States. In the words of the Congressional Conference Committee report which provides the legislative history in support of the Clean Water Act of 1977, EPA "has been provided all of the legislative tools to require the utilization of such innovative and alternative wastewater treatment processes and techniques." This legislation identifies land treatment as one of the best and most reliable technologies included in a group of technologies designated as innovative or alternative. These innovative/alternative technologies which emphasize recycling and energy conservation are to be given preference over conventional technologies in the US EPA program to abate water pollution.

There are greater than twelve new provisions of the Clean Water Act which provide incentives and requirements for the use of innovative and alternative technologies in the Construction Grants Program including the following:

- Innovative/alternative (I/A) projects may receive an additional 10% in Federal funds (i.e., the Federal share may be increased from the conventional 75% to 85% of the eligible project costs).

- The States are required to spend 2% of their allotment of construction grant funds on increasing the Federal share to 85% for I/A projects in fiscal years 1979 and 1980. In 1981, this "set-aside" increases to 3%.
- The Federal government may participate with full grant funding in I/A projects which are up to 15% more costly than the most cost-effective conventional alternative.
- I/A projects may receive 100% Federal grants for modification or replacement if they fail to meet their design criteria at a reasonable cost.
- I/A projects may receive 100% Federal grants (research and development grants as opposed to construction grants) for technical evaluations, training of personnel and information dissemination.
- The land used for storing wastewater prior to application to the land is eligible for grant funds. The land to which the wastewater was applied for actual treatment in a land treatment system was already an eligible cost.

We are presently in the process of promulgating construction grant regulations to implement these and other sections of the new law. The final regulations will be published in September of this year.

Obviously, the future holds even more promise for expanding use of land treatment processes in the United States. There is a strengthening sense among my staff that we are on the threshold of a revolution in the concept of wastewater treatment in the United States. There is growing optimism that alternatives which emphasize recycling will indeed become the standard in the near future.

Such optimism for effective recycling of wastewaters is not new. It has budded repeatedly over the centuries and several times in the last century. The recent history, the 19th and 20th centuries, of land treatment furnishes some timeless quotes to this effect: "The right way to dispose of town sewage is to apply it continuously on land and it is only by such application that the pollution of rivers can be avoided" -- Report of Royal Commission, England 1857; "...The most efficient purification of sewage can be attained by its application on land." Also "...on properly managed sewage farms the utilization of sewage is not prejudicial to health." -- George Rafter, U.S. Geological Report No. 1, 1897, and No. 3 1899.

Although the Muskegon, Michigan, system will be covered in detail as a design example, I want to cite it as one example of what the future may hold for nutrient recycling. This system, designed to serve a population equivalent to 430,000 people, recycles water and fertilizer nutrients to grow corn. Operating at a population load of 300,000 people in 1976 the 2,200 ha farm produced 144,000 hl (400,000 bu) of corn. Revenues from the sale of the crop offset much of the costs to operate all components of the wastewater management system. About 90 metric tons of phosphorus, 230 metric tons of nitrogen, and 250 metric tons of potassium were recycled by crop uptake as the wastewater was reclaimed.

While there have been obvious project benefits from the recovery and recycling of nutrients through crop production and revenues gained through the sale of these crops to help offset the system's overall operating costs, there are certainly important water quality benefits provided by this wastewater treatment system. Improvements have already been observed in the water quality of the lakes that had been receiving the wastewater discharges now passing through the land treatment system. And this marriage of water quality improvement and wastewater treatment with nutrient recovery and recycling has been achieved without adverse effects on the groundwater underlying the project site or surface waters into which the renovated water is discharged.

THE FUTURE OF LAND TREATMENT

The future role of land treatment in our Nation's water pollution control efforts looks brighter than ever, but continues to face many of the major impediments we have faced from the beginning. Overly restrictive state regulations, resistance by the consultants and engineers trained or experienced only in traditional sanitary engineering techniques, public acceptance issues and public health concerns remain to be more of a problem in the implementation of new projects than our ability to properly design and operate cost-effective and environmentally sound land treatment systems. We need continuing help from the research community in addressing these impediments in their future research efforts to compliment the efforts being made through the Construction Grants Program. Continued interagency coordination and cooperation with the university research community, as well as revitalized international exchanges, should play a major role in the success of our future efforts to advance land treatment practices.

CONCLUSION

I applaud the Corps for holding this conference and look forward to reviewing its results. You in this room here have been tremendous experimentally, but we are now ready for implementation. We need your help now. Our future success in advancing land treatment may well be measured by how fast we can get adequate information to the users.

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10. EPA Policy on Land Treatment of Municipal Wastewater. October 3, 1977, memorandum from Administrator Costle.

LEGISLATION

FEDERAL GUIDELINES FOR USE OF LAND TREATMENT OF WASTEWATER IN THE UNITED STATES

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This paper describes a number of major elements of the Environmental Protection Agency's land treatment program which collectively provide the basic framework for the Federal guidelines in this area. Discussed in the paper are the criteria for best practicable waste treatment technology, the EPA Administrator's October 3, 1977, policy statement on land treatment, a draft construction grants program requirements memorandum on evaluation of land treatment systems and the "Process Design Manual for Land Treatment of Municipal Wastewater."

INTRODUCTION

In preparing to write this paper, it was necessary to reflect upon the title - "Federal Guidelines for Use of Land Treatment of Wastewater in the United States" - as the initial step in deciding on the approach to take with the paper. One would think that determining the approach for such a paper would be easy for someone who represents a Federal regulatory agency like the Environmental Protection Agency (EPA). It would seem that all one has to do to write a paper on Federal guidelines in a certain area is pull together and cite the numerous Federal regulations on the subject.

This, of course, is not the case, particularly where EPA is concerned and particularly where land treatment is the subject. In addition to its regulatory responsibilities, EPA also operates in a great number of other areas, including a multi-billion dollar grant program for the construction of municipal wastewater treatment facilities and, as evidenced by a number of the other papers being presented at this conference, and extensive research and development programs covering a wide variety of subjects.

Likewise, the Clean Water Act, which is the major piece of Federal legislation covering wastewater practices in this country, does not specifically require EPA to develop regulations for the control of land treatment systems. It does, however, specifically require the Agency to encourage practices, such as land treatment, which provide for reclaiming of water and/or reclamation of potential sewage pollutants through the production of agriculture, silviculture or aquaculture. In fact, if one were to limit the coverage of this paper to Federal guidelines for land treatment of wastewater in the narrowest regulatory sense, the paper could be ended at this point.

The fact is that there is not a standard set of Federal rules which have been developed for the express purpose of regulating land treatment systems. This is not to say that EPA is not interested in seeing that land treatment systems are designed and operated so that they will also be compatible with the environmental protection goals of the Agency. EPA is of course vitally concerned with the implementation and management of land treatment systems in this country so that they will be built and operated in a manner which ensures protection of the environment. This objective must obviously be in harmony with the Agency's desire to achieve the reclamation, recycling and economic benefits which result from the use of land treatment systems.

Because of the nature of the EPA land treatment program, the "encouragement" activities cannot be easily separated from the "regulatory" activities. For this reason, the discussion of these activities in the remainder of this paper is interspersed. In keeping with the title of the paper, however, major emphasis is given to those pieces of the Federal program which are aimed at ensuring the proper management of land treatment systems from an environmental impacts point of view.

ELEMENTS OF THE EPA PROGRAM

Most of the identifiable pieces which collectively can be considered as the "Federal guidelines for the use of land treatment of wastewater" are linked in some way to the EPA Construction Grants Program. Section 201 of the Clean Water Act, which authorizes EPA to make grants to municipalities for the construction of wastewater treatment facilities, also requires that grant applicants evaluate "alternative waste management techniques...[which] provide for the application of best practicable waste treatment technology...." Section 201 further specifies that best practicable waste treatment technology (BPWTT) includes "reclaiming and recycling of water and confined disposal of pollutants so they will not migrate to cause water or other environmental pollution...." "Alternative Waste Management Techniques for Best Practicable Waste Treatment" was published by EPA pursuant to Section 304 of the Clean Water Act. Contained in this document are criteria for systems in three categories: treatment and discharge to navigable waters; land application and utilization practices; and wastewater reuse.

There are a number of other items related to the Agency's implementation of the Construction Grants Program which are important elements of the "Federal guidelines" for land treatment. These include: an extensive series of technical guidance documents most notable of which is the recently issued "Process Design Manual for Land Treatment of Municipal Wastewater"; an October 3, 1977 Administrator's policy statement on land treatment, and, a construction grants Program Requirements Memorandum (in draft form at the time of this writing) which provides guidance to implement the Policy Statement.

It is true that construction grant requirements are directly applicable only to the planning, design and construction of facilities which receive grant funding. Furthermore, the coverage of the rules and regulations under which the Construction Grants Program operates does not extend to the long-term operation of completed facilities, whether these facilities were assisted with grant funds or not. However, because of the magnitude of the Construction Grants Program (i.e., eventual expenditures likely to exceed $50 billion and greater than 12,000 projects), its impact on municipal wastewater treatment practices in this country obviously is considerable. Consequently, even though the Construction Grants Program elements listed above are not regulatory in the formal sense of the word, their impact is often comparable to that of regulations. Where state regulations or guidelines do not exist to control a particular practice, applicable construction grant guidelines have in some instances been adapted to fill this void. In other cases, existing state guidelines have been revised as necessary and appropriate in recognition of Federal requirements.

With respect to state land treatment guidelines, most states have some form of requirements available, although a significant number of states do not have any controls in place. Even in some states where land treatment guidelines had already been developed, revisions are being considered to provide more consistency with the Federal requirements issued through the Construction Grants Program. In all fairness, a number of states have also taken the position that their requirements for land treatment are justifiably different to account for local variations in the factors which affect land treatment systems.

The remainder of this paper discusses the major features of the above mentioned program elements which collectively comprise the Federal guidelines for land treatment.

BEST PRACTICABLE WASTE TREATMENT TECHNOLOGY

The Best Practicable Waste Treatment Technology (BPWTT) criteria published by EPA in October 1975 contain requirements for land treatment systems which for all intents and purposes have the impact of Federal land treatment regulations.

Municipal wastewater treatment plants are required by law to provide for the application of BPWTT by July 1, 1983.

The principal objective of the BPWTT criteria for land application and land utilization systems is to protect groundwater for drinking water purposes. The criteria describe three cases as follows:

CASE I: The groundwater can potentially be used for drinking water supply. In this case the groundwater resulting from the land application of wastewater, including the affected native groundwater, must meet the maximum contaminant levels for inorganic and organic chemicals specified in the National Interim Primary Drinking Water Regulations.

CASE II: The groundwater is presently being used for drinking water supply. In this case, the groundwater must meet the maximum microbiological contaminant levels specified in the National Interim Primary Drinking Water Regulations as well as the levels for chemicals specified in Case I.

CASE III: The groundwater has uses other than drinking water supply. In this case groundwater criteria are to be developed by the EPA Regional Administrator based on the present or potential use of the groundwater.

The National Interim Primary Drinking Water Standards for inorganic and organic chemical and microbiological contaminants presently include limits on arsenic, barium, cadmium, chromium, lead, mercury, nitrate, selenium, silver, fluoride, endrin, lindane, methoxychlor, toxaphene, chlorophenoxys (2,4-D and 2,4,5-TP Silvex) and coliform bacteria. Where a land treatment system results in a surface discharge (i.e., either effluent collected through underdrains or runoff from an overland flow system), the wastewater must meet the BPWTT standards for treatment and discharge. The minimum level of treatment required for municipal systems which treat and discharge to surface is secondary treatment as defined by EPA in terms of five-day BOD, suspended solids and pH. It is important to note that in any case, the point at which the wastewater is measured for compliance with the Federal BPWTT criteria is when it <u>leaves</u> the land treatment system (i.e., becomes part of the permanent groundwater or is surface discharged) and <u>not</u> at the point of application to the land.

EPA POLICY STATEMENT ON LAND TREATMENT OF MUNICIPAL WASTEWATER

On October 3, 1977, the Administrator of the Environmental Protection Agency issued a policy statement on land treatment of municipal wastewater. The policy statement highlighted the Agency's intent to "press vigorously for publicly owned treatment works to utilize land treatment processes to reclaim and recycle municipal wastewater." It also required that construction grant applicants who do not select methods that encourage water conservation, wastewater reclamation and reuse must provide a complete justification for the rejection of land treatment.

With respect to "guidelines" for land treatment systems, the October 3, 1977, statement emphasized the importance of ensuring that Federal, state and local requirements and regulations are imposed at proper points in the treatment system. To this end, the statement indicated that "Whenever states insist upon placing unnecessarily stringent preapplication treatment requirements upon land treatment, such as requiring EPA secondary effluent quality in all cases prior to application on the land, the unnecessary wastewater treatment facilities will not be funded by EPA." This is not to say that secondary treatment or an even higher level of treatment may not be necessary for some land application projects. The intent is to cause arbitrary preapplication treatment requirements, such as a minimum of secondary treatment or better in all cases, to be revised to reflect a level of preapplication treatment appropriate for the given situation. Just as secondary treatment may be appropriate in some instances, the equivalent of primary treatment or even raw wastewater may be sufficient in others. More discussion of preapplication treatment requirements is presented in the next section of this paper.

PROGRAM REQUIREMENTS MEMORANDUM FOR EVALUATION OF LAND TREATMENT ALTERNATIVES

In April 1978, the Agency issued a draft Program Requirements Memorandum (PRM) titled "Revision of Agency Guidance for Evaluation of Land Treatment Alternatives." PRM's are issued by the Agency to set forth policy for implementation of the Construction Grants Program. This particular PRM was developed to provide guidance in support of the October 1978 policy statement issued by the Administrator of EPA. It is expected that the PRM will be issued in final form by the time this conference is held in late August, 1978. For this reason, the following discussion should be read remembering that revisions prior to issuance are possible.

Again, while a guidance document such as this PRM is specifically developed for implementation of the EPA Construction Grants Program, it does constitute Agency policy. In this way it can be interpreted as having the effect of Federal guidelines, particularly because PRM's and other similar issuances are widely circulated for review and comment prior to publication in final form.

The PRM establishes the "Process Design Manual for Land Treatment of Municipal Wastewater" as the principal reference of technical information upon which review of construction grant projects involving land treatment will be based. The Design Manual is discussed briefly in the next section of this paper. The PRM also cites three planning and design factors which have historically limited the use of land treatment in this country. These factors are: (1) overly conservative and, consequently, costly design of slow rate (irrigation) systems; (2) failure to consider rapid infiltration as a proven and implementable land treatment alternative; and (3) requirements of a substantially higher and more costly level of preapplication treatment than is needed to protect health and ensure design performance.

The PRM also provides more specific guidance on four items related to the design of land treatment systems, plus cost. These four design-related factors are site selection, loading rates and land area, preapplication treatment, and environmental effects. Much of the information pertinent to these factors is included in the PRM by reference to other documents such as the Design Manual and the BPWTT publication.

An important point to re-emphasize is EPA's approach to the issuance of design criteria such as these. The Agency recognizes that no single value or even one set of values can be realistically applied to all locations considering the variability across the country in climate, geology, treatment needs and other factors affecting the design of land treatment systems. For this reason the EPA guidelines are varied to suit a number of possible situations and include ranges of values wherever possible. For example, the range of application rates indicated in the PRM varies from 0.6-6 M/yr for slow rate systems to 6-170 M/yr rapid infiltration systems.

This concept of flexibility is particularly important with respect to EPA's position on preapplication treatment requirements. The EPA statements on preapplication treatment have apparently been misconstrued by some who have interpreted them as indicating that the Agency will not support any project which requires secondary treatment prior to application to the land. This is not the case. What the Agency is saying is that the level of preapplication treatment must be suited to the particular situation. As previously noted, in some cases primary treated or even raw sewage may be acceptable for application to the land, while in other instances treatment beyond that provided by secondary may be necessary. Clearly an arbitrary requirement that all wastewater has to receive secondary treatment prior to land application is not consistent with the Agency's approach. The guidance on preapplication treatment included in the PRM ranges from simple screening or comminution for overland flow in isolated areas with no public access to extensive BOD and suspended solids control with disinfection for slow rate systems in public access areas such as parks and golf courses.

PROCESS DESIGN MANUAL FOR LAND TREATMENT OF MUNICIPAL WASTEWATER

The Design Manual is a comprehensive, state-of-the-art document prepared through the cooperative efforts of the Environmental Protection Agency, the U.S. Army Corps of Engineers and the U.S. Department of Agriculture. While the manual was developed for a number of purposes and a variety of audiences, one of its principal uses is as design guidance for Federally funded construction grant projects. In this way it serves as the technical basis for the entire framework of Federal guidelines for land treatment.

The manual presents a rational procedure for the planning and design of three types of land treatment systems. Chapters 1 and 2 introduce and define the land treatment systems referred to as slow rate, rapid infiltration, and overland flow. Chapters 3 through 5 cover the procedures for preliminary design, site evaluation, and detailed design. Chapter 6 is devoted to design of small systems where it may be advantageous to forego some of the detailed site evaluation required for design of large systems. Chapter 7 is a compilation of case histories, and Chapter 8 is a design example utilizing the procedures set forth in the manual. The appendices furnish indepth analyses of selected factors of special interest.

The manual is assembled in loose leaf form for the purpose of easy revision as new information becomes available. The state-of-the-art for some aspects of these land treatment technologies is changing rapidly. It is anticipated that the first update of the manual will occur in the fall of 1979.

SUMMARY

The intent of this paper was to present an overview of the Federal guidelines applicable to the land treatment of municipal wastewater. Because it would not be possible to describe in detail the individual features of such guidelines in a paper of this nature, the approach taken was to outline the major elements which provide the framework for the Federal criteria applied to land treatment systems. More specific information is contained in the documents and publications highlighted in the paper. Copies of "Alternative Waste Management Techniques for Best Practicable Waste Treatment" (EPA 430/9-75-013), the Administrator's Policy Statement of October 3, 1977, and the Program Requirements Memorandum on Evaluation of land treatment alternatives (when available) can be obtained by writing to Chief, Municipal Technology Branch (WH-547), EPA, Washington, D.C. 20460. The "Process Design Manual for Land Treatment of Municipal Wastewater" (EPA 625/1-77-008) can be obtained from the Environmental Research Information Center, U.S. Environmental Protection Agency, Cincinnati, Ohio 45268.

STATE GUIDELINES FOR THE USE OF LAND TREATMENT OF WASTEWATER

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Abstract

The history of land treatment in New York State and development of early State regulations is discussed.

Present State regulations and guidelines are presented with discussions as to their use in the implementation of land treatment systems. The regulations discussed include ground and surface water standards pertaining to land treatment or disposal and project review requirements including design guidelines. The use of New York State's permit system as a monitoring and management control system is also presented, including recommended operating requirements.

A need for multi-program involvement is demonstrated if land treatment opportunities are to be fully realized. The interrelationship of various State and non-State programs and institutions impacting on land treatment are developed. A multi-disciplined strategy is necessary for effective development of land treatment opportunities.

Both obstacles and opportunities in land treatment are discussed with emphasis on activities that are either needed to fully develop land treatment alternatives or that may present obstacles to such development.

Introduction

New York is a large diverse State with a population in excess of 18 million people and a land area of almost 50,000 square miles. Its population is concentrated in 10 standard metropolitan statistical areas which roughly form a band starting with Long Island and New York City up the Hudson River Valley to Albany, west through the Mohawk River Valley through Utica and on to Syracuse and then along the shores of Lake Ontario and Erie including the cities of Rochester, Niagara Falls and Buffalo.

New York is also geographically diverse, consisting of the Adirondack and Catskill Mountain terrains, the 155 mile Hudson tidal estuary, the central rolling farmlands and Finger Lake areas, the Great Lakes shore area, the Appalachian Plateau and the unique sandy Long Island area. Though the State is generally considered highly industrialized, more than 75% of its lands are agricultural and forest and bush woodlands.(1) This population and geographic distribution thus provides the opportunity for realizing varied effective land treatment opportunities.

The greatest potential for land treatment would be expected in the fruit and vegetable farming areas along the Great Lakes and in the Finger Lakes region and possibly in the lumber industry areas adjoining the Adirondack and Catskill Mountains and in the Southern Tier.

History of Land Treatment in New York State

The first municipal plant in New York State to use land treatment for wastewater effluent was the Village of Lake George in 1936. There are now approximately 25 municipalities discharging about 9 MGD to the groundwaters of New York State. These systems all use a rapid infiltration mode of discharge either through tile fields, leaching pits, or non-underdrained sand filters.[2] Our Department has extensively studied the Lake George Village system in relation to the trophic state of Lake George and the phosphorus removal capabilities of various soils. This system protects the lake from excessive phosphorus inputs by providing an effective land treatment system for the removal of nutrients from a secondary effluent.[3] Our research unit has also used this system as a historic base in its extensive studies on the phosphorus removal capabilities of various New York State soils. This research is directed towards providing a rational design based on soil classifications for the removal of phosphorus in septic tank-tile field systems.[4]

Spray irrigation as a viable means of industrial wastewater treatment evolved in New York State primarily because of favorable circumstances in the food processing industry. Their processing schedules occur primarily during late summer and early fall, which is the most favorable season for spray irrigation and the least favorable for surface discharges. Additionally, many of these food processors are located in the farming area of central New York State away from main watercourses and usually on the headwaters of minor tributaries which periodically become intermittent or completely dry in flow. These natural conditions demand high levels of treatment for short periods of time in order to protect the surface water quality. Spray irrigation, therefore, becomes very cost effective under these circumstances.[5] Over the last 10 to 20 years, 15 spray irrigation systems have been developed in New York State, applying a total of 4.7 MGD to 549 acres of land.

Early State Regulations and Guidelines

New York State groundwater classes and standards which were promulgated in May of 1967 essentially included a parametric list of drinking water standards which were applicable at the point of contact of the wastewater and ground surface. This effluent standard approach severely limited the development of cost effective spray irrigation systems, particularly systems containing non critical (non-toxic) organic material as found in some municipal and all food processing wastewaters.

Guidelines for the design review of spray irrigation systems have evolved in New York State over the last five (5) years along with the development of land disposal systems by the fruit and vegetable industry. These guidelines were initially conservative especially in the areas of unknown crop and soil renovative capability. They also were in part designed to prevent mis-management of spray systems which in the past have caused public nuisance complaints due to odors, runoff, ponding, etc. It should be noted that our guidelines are just that, "guidelines", and as such are subject to modification for any well developed specific project design.

Present Regulations and Guidelines

Several different types of regulations and guidelines guide our current activities in the implementation of land for waste treatment:

- Environmental Standards
  - Groundwater Standards
  - Surface Water Standards
- Project Review
  - Environmental Compatibility (NEPA/SEQR)
  - Wastewater Facilities Report
  - Design Guidelines
- Management Control
  - SPDES Permit
  - Operational Guidelines
  - Monitoring

Groundwater Standards:

In July of 1977, the State of New York completed public hearings on proposed revisions to its groundwater classifications and standards which are formally designated as Part 703 of Title 6 of the Official Compilation of Codes, Rules and Regulations of the State of New York. These revisions (6NYCRR 703.10(a)(3)) contain provisions for allowing the "potential renovative capabilities of a water management system employing land application technology and land utilization practices, provided for each specific proposal it has been demonstrated to the satisfaction of the Department that:

There shall be no actual or potential public health hazard; and
Applicable Water Quality Standards shall be met in the saturated zone; and
Applicable Water Quality Standards shall not be contravened in any adjacent waters of the State."

Our ground and surface water quality standards are found in Part 700 of 6 NYCRR and are promulgated pursuant to authority under Article 17 of the Environmental Conservation Law of the State of New York. The best usage class of generally all of New York's groundwaters are as a source of potable water supply. As such, quality standards are consistent with raw water quality standards promulgated in Sub-part 5-1 of 10 NYCRR by the New York State Department of Health, and applicable standards promulgated under the Safe Drinking Water Act (PL 93-523) in addition to quality standards promulgated by our Department in Part 703.

Surface water quality standards are promulgated according to best usage classifications with standards consistent with the propagation and protection of fish.

Project Review:

The New York State Environmental Quality Review Act (SEQR) and the National Environmental Policy Act (NEPA) require respectively an environmental impact statement and an environmental assessment summary as part of the construction grant process. The combined State and Federal environmental assessment identifies any major environmental constraints such as aesthetics, site locations, land usage, etc. The State (SEQR) procedure parallels the construction grant process and in addition to identifying major environmental consideration is used as a coordinating mechanism for all environmental permits.

The Environmental Conservation Law (Article 17) of the State of New York requires that plans for disposal systems be submitted for approval. The engineering review by the Department of Environmental Conservation is directed towards insuring construction and operation of wastewater facilities which can reasonably be expected to protect and maintain compliance with applicable State surface water and groundwater classifications and standards of quality and purity, applicable provisions of Article 17, applicable provisions of the Clean Water Act, prevention of public health hazards and minimization of nuisance problems.

The review is conducted in two (2) phases. The first phase involves the submission of an "Engineering Report" or "Wastewater Facilities Report" for the wastewater treatment facility followed by the second phase submission of "Plans and Specifications".

The Engineering Report must be a final and comprehensive description of the wastewater problems and the proposed solution, including applicable design criteria. The submission should include the manufacturing process descriptions and operations as they relate to wastewater generation or in the case of a municipal system, must be a properly developed facilities plan. A discussion of treatment objectives should be included. Facilities design, in this document, means adequate process sizing of wastewater treatment units and/or equipment to achieve acceptable effluent levels. The report should consist of narrative descriptions, engineering calculations, design criteria and process and equipment sizes.

The report review is conducted to evaluate whether: The design is directed at the required objectives; a proper basis for design has been established; acceptable engineering design principles have been utilized and the design has a reasonable expectation of meeting its objective. The scope of the engineering report should be sufficient to fulfill these review objectives.

The "Engineering Report" review of land application techniques by the Department of Environmental Conservation is guided in part by the guidelines in Figure 1. Again, it should be emphasized that these are review guidelines and subject to revision and modifications depending on specific site conditions and "State of the art" literature.

The plans and sepcifications include drawings and narrative descriptions of treatment systems as prepared for construction and descriptions of equipment as prepared for purchase. The review is conducted to determine whether the treatment facilities can be constructed, installed and operated in a manner consistent with the design and objectives presented in the approved engineering report.

Because land application by nature must be site specific and because a wide range of design possibilities is available, it is not well suited to standardized design guidelines. The few

New York State Department of Environmental Conservation's
Guidelines for the Design of Spray Irrigation Systems

Application Rates

1. Hydraulic - maximum application - 3 in/wk
2. Organic - maximum 500 lb BOD/ac/day (recent data indicates this value may be excessively conservative)
3. Inorganic - maximum salinity 1,000 mg/l

Groundwater

Class GA groundwater standards apply in saturated zone pursuant to Sec. 703.10 (a)(3) of 6NYCRR.

Spray Scheduling

1. Duration - 8 hours or less
2. Spray use/rest ratio - 1:4 or greater (maintain aerobic soil conditions)
3. Daylight spray only
4. Operation only from May 1 to November 30
5. No spraying during rainfall

Spray Field Layout

1. No overlapping wetted area
2. At least 200 feet separation to surface waters, dwelling and public roadways
3. Provide runoff collection and recirculation

Crop Cover Maintenance

1. Recommended crop - reed canarygrass
2. Crop harvesting - cut before going to seed, at least twice per season
3. Subsoiling - recommended seasonally

Pretreatment

1. Grit, stone and debris protection
2. Equalization and pH control if required
3. Storage - 2 weeks plus any flow generated in prohibited time period
4. Secondary treatment and disinfection for sanitary wastes - (Subject to reconsideration under Sec. 703.10 (a)(3) "Exceptions")

Monitoring

1. Flow measurement and water characteristics
2. Rainfall, runoff and groundwater elevation and quality

Figure 1

existing guidelines are generally limited in scope, dealing mainly with spray irrigation. In lieu of guidelines, the design must rely on a comprehensive understanding of the principles involved, site evaluation by specialists and ingenuity. A multidisciplinary approach to designing land application systems is necessary. This most desirably includes engineers, hydrologists, geologists, soil scientists and agronomists.

Management Control:

After conducting its engineering review of a proposed land application system, the Department of Environmental Conservation, drafts a proposed effluent discharge permit consistent with the Clean Water Act and the State Environmental Conservation Law. New York State has been delegated the National Pollutant Discharge Elimination System (NPDES) program. The State program in addition to including surface water discharges as required under the NPDES program also includes groundwater discharges such as land treatment systems.

The permit contains, for the proposal, management requirements that are designed to protect groundwater quality and prevent public nuisance conditions through maintenance of a healthy crop cover, adequate soil percolation capacity and aerobic crop and soil conditions. The permit requires operation of such systems in accordance with the parameters developed in its site specific design and accepted through DEC review procedures. The following items provide the main elements of a SPDES permit for a land treatment system employing spray irrigation:

General Management Provisions
Site Specific Design Parameters
- a. Maximum instantaneous application rate (in./hr.)
- b. Overall application rate (in./wk.) (includes rest period)

Monitoring Provisions
- a. Flow measurement and wastewater characteristics
- b. Application rates, rainfall and runoff
- c. Groundwater elevation and quality

Pollution Abatement Time Schedules (for)
- a. Operations and management manual
- b. Other problems that may occur

The following recommended permit conditions are designed to promote good management practices for spray irrigation sites:

Spray irrigation should be practiced only during the period May 1 to November 30.
Spray irrigation should be practiced only during daylight hours.
Spray irrigation should not be practiced during periods of measurable natural precipitation.
No area of the spray irrigation fields should be irrigated on two consecutive days.
Surface runoff or irrigated wastewater from the spray field shall not be permitted, nor shall surface runoff be permitted to enter the spray fields.
A viable cover crop shall be maintained on all spray irrigation fields.
Spray irrigation fields should be designed and operated to prevent surface accumulation of wastewaters.
The spray irrigation system shall at all times be operated in a manner acceptable to the Department.

These management provisions represent good technical practice. They have also evolved in part because of past abuses and problems and, if adhered to "in good faith", should assure public acceptance of spray irrigation systems as good neighbors. Again, it must be emphasized that, because of the public participation procedures of our water pollution control laws, we can expect objections from potential adjacent property owners of spray systems because of bad experiences from other systems.

It is our opinion that proper management control is so important for the continued viability and public acceptance of land treatment that we have followed up and required all of New York State's existing spray irrigation systems to develop and provide us with an operations and management manual for each system. This manual is intended to provide a definition of the proper operating conditions for each system, particularly for existing operations that have evolved over the years rather than being properly designed initially. The need for this follow-up is generally associated with the systems that have been built prior to the adoption of the SPDES permit system. The SPDES permit is used as the legal machanism to require submission of these manuals in accordance with a time schedule.

Our SPDES permits require monitoring of the applied wastewater and groundwater quality. Groundwater monitoring is by sampling wells so as to determine water flow and characteristics into and out of land disposal sites. The purpose of this groundwater monitoring is to provide assurance for the continued protection and maintenance of groundwater quality classifications and standards. Sampling frequency is generally related to the rate of groundwater movement and usually is monthly or quarterly. We generally require a minimum of three (3) off-field wells placed so as to insure interception of groundwater flow into and out of a land treatment site. The sampling wells are required to be constructed so as to obtain a representative sample. Casings must be sealed and capped to prevent the entrance of surface rain and irrigated waters. The critical sampling point is the top surface of the groundwater, therefore, screens or slits in the casing are needed to insure proper sample collection.

Failure to maintain permit conditions or groundwater quality subjects the owner of the land applications system to the enforcement provisions of Article 17 of the Environmental Conservation Law of New York State. These provisions are similar to the NPDES/SPDES point discharge enforcement programs. If violations or problems occur initial contact is made in a cooperative manner and then, if necessary, follow-up through our hearing procedures.

Multi-Program Involvement

Recent Federal legislation on the Clean Water Act and the Resource Conservation and Recovery Act both encourage the use of land for disposal and/or treatment. At this time, our Department intends to fully develop effective land treatment opportunities in New York State. Fortunately, we have within a single State Department, the Department of Environmental Conservation, the key divisions necessary for such an implementation. These program divisions are air resources, fish and wildlife, land resources and forest management, pure waters, and solid waste management.

The Division of Pure Waters, because of having responsibility for implementation of much of the Clean Water Act, is greatly involved with this implementation. Its activities in developing effective land treatment systems include:

1. Identification and implementation of municipal land treatment systems through (201 Grants) wastewater treatment facilities grants.
2. Contact with consulting engineers (conferences and guidance).
3. Implementation of toxic strategy and pretreatment programs which impact on acceptable wastewater characteristics.
4. Water research (phosphorus removal studies).
5. Development of ground and surface water quality standards.
6. Development of project design and review guidelines.

The Division of Land Resources and Forest Management has the following responsibilities in the implementation of land treatment systems:

1. Through water quality management plans (208 studies), the identification of potential land treatment and/or disposal sites for wastewaters and residual wastes.
2. The identification and use of State forest lands for land treatment and reuse.

The Division of Solid Waste Management impacts in the following program activities:

1. The implementation of the RCRA through development of secure landfills as co-disposal sites (refuse and municipal sludge).

The Division of Fish and Wildlife:

1. Development of freshwater wetland inventory and program (include wastewater disposal VS beneficial use of Freshwater Wetlands).
2. Development of ecological standards for the protection of fish and wildlife.

In addition, our Department is utilizing its manpower training and management, environmental education and regional operations resources to fully implement land treatment opportunities in New York State.

The following schematic in Figure 2 shows the multi-program involvement necessary for the development of a land treatment system. It illustrates how by having a relatively clear organization structure the many disciplines involved can be directed towards achieving this goal.

Use of Non-New York State Expertise

Our Department is only one of the many institutions that must be involved in order to develop land treatment opportunities.

Information for the education and use of professionals, public officials and communities is necessary for the proper development of land as a waste management alternative. Cornell University prepared a comprehensive modular course on land application of wastewater, including topics such as legal aspects, site evaluation, nitrogen removal and others. Assistance of this kind is necessary and should be supported. The engineering community in particular must have clear, concise technical principles for the design of these multi-disciplined systems.

Federal agency assistance is also used and is necessary. The EPA in the promulgation of laws, rules and regulations and in providing incentives plays an important role in the development of land treatment opportunities. This is particularly true regarding the additional grants available under Innovative and Alternative Technologies. The EPA policy statement making ineligible for construction grants unnecessary pretreatment facilities is important for the development of cost effective land treatment systems.

The U.S. Army Corps of Engineers, in cooperation with the EPA and Department of Agriculture, have provided a very timely Process Design Manual for Land Treatment of Municipal Wastewaters.

A multi-media education program is also necessary to reach the community, public officials and the engineering and scientific community. This can be accomplished through the education community, various professional conferences, public and trade journals and public relations news releases. We have held in our Department with the assistance of Cornell University and the Corps of Engineers an executive conference for the purposes of implementing land treatment opportunities in New York State.

Obstacles to Land Treatment

Land as a waste management alternative provides many opportunities, but it also has limitations and problems.

Toxic Substance Control:

The use of land for treatment or disposal must be undertaken carefully to prevent future environmental problems resulting from contaminants in surface and groundwaters and possibly in the food web. In order to prevent this, industrial pretreatment programs must be implemented through the Clean Water Act. The Toxic Substance Control Act must be effective. Secure and safe disposal sites must be developed under the Resource Conservation and Control Act. Our many environmental programs must be coordinated so as to track and control hazardous substances to a final, safe disposal sink. Toxic substance control through the land is an entire area of concern in itself particularly regarding contaminated sludges or dredging as in the case of PCB laden spoils from the Hudson River.(6)

Education:

The use of land as a treatment or disposal medium historically has not been emphasized in our universities. It is by nature a multi-disciplined approach and this fact in itself has been an obstacle because consulting engineers have not been trained in this manner nor are there conveniently available proper design principles. The disciplines involved in addition to engineering principles include the sciences and public health. Consulting engineers faced for the first time with

Multi-Institutional Involvement for Land Treatment Implementation

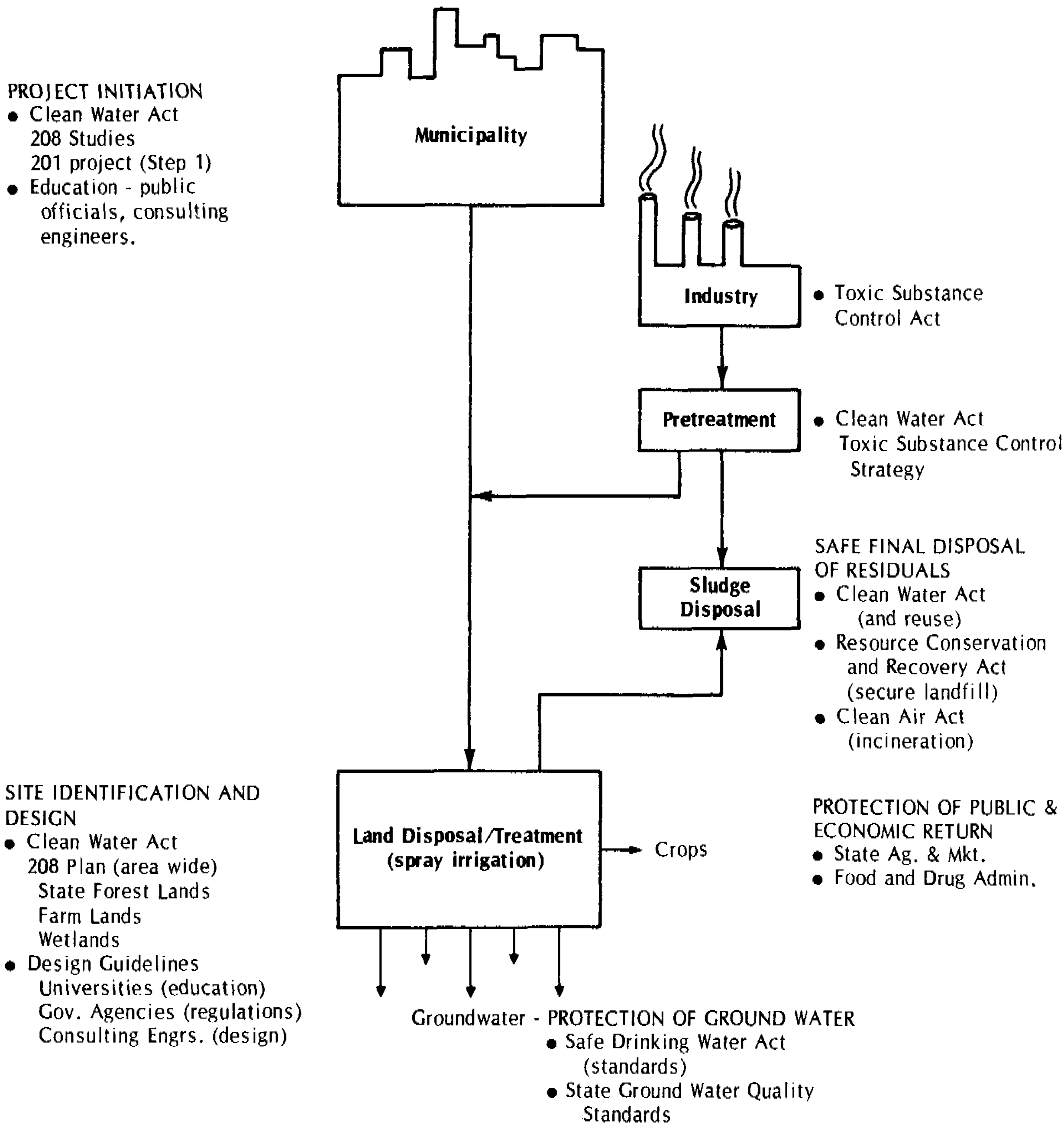

Figure 2

having to consider land management alternatives are bombarded with a seemingly exhaustive and sometimes contradicatory array of literature. Our Department has, for the last four years, maintained an annual land disposal literature update and in doing so considers for review an average of 150 separate publications per year. Professional manpower training programs for the design engineers are needed in a relatively short period of time.

Management:

At this time, much of our agency's contact with the public regarding existing systems has been due to nuisance conditions such as overt runoff, ponding and odors. The public participation procedures of our laws are such that should we be faced with a land management proposal we can expect objections from potential neighbors or communities due to bad experiences from other systems. In many cases, it is these issues which determine the fate of proposed systems and not the protection of our water resources. It is for these reasons that many of our design guideline items and permit conditions are directed towards encouraging proper system management.

Site Identification:

Many of the nation's water pollution control programs because of early State programs (e.g. New York State's Pure Waters Program begun in 1965), the Federal Water Pollution Control Act of 1972 and the 1977 Clean Water Act are in a mature accelerated phase and keyed into priority lists. Many large projects are currently under construction. For these reasons, it is important that all available resources be mobilized and directed towards the timely identification of any remaining potential land management opportunities. Agricultural and forest lands contiguous to future wastewater treatment projects must be identified and factored into the Step I phase of the 201 construction grant. Timing is critical and inter and intra agency cooperation is necessary to develop what remaining land management opportunities exist.

Opportunities in Land Treatment

The use of land as a waste management alternative besides providing an economic and desirable wastewater treatment alternative provides some unique opportunities.

Wetlands at first glance may seem to be areas in disarray because of a mixture of plants and animals and in the past have been filled, dredged and drained for urban and agricultural development. In reality, they are valuable resources providing fish and wildlife habitat, open space, flood control and maintenance of water quality. The use of land as a wastewater treatment alternative provides us a unique opportunity to help preserve, protect and even create wetlands. The wastewaters which so overburdened our streams during the drought in the early 60's also, in some instances provided the moisture for the maintenance of many wetlands. The Brookhaven National Laboratory developed and has been operating Marsh/Meadow/Pond systems which in effect are created wetlands. These are closed natural systems using a combination of land application techniques showing excellent energy and cost savings.

Some of our older,higher organically loaded spray irrigation systems have accumulated approximately 125 tons per acre of organics in the first foot of soil.[7] Opportunities exist in these systems for the improvement or creation of farmlands or wetlands by the use of land treatment techniques. It appears that for these acclimated systems, loadings can exceed 15,000 lbs./acre/day of organics with no apparent adverse effects on groundwater quality. These application rates are significant when compared to a generally accepted design limit of 600 lbs./acre/day for wastewater treatment lagoons.

The use of land treatment alternatives with imaginative leadership can be used as a mechanism for the preservation of open space in or adjacent to urban areas. Land treatment alternatives that are not cost effective because of the high value of land should be given reconsideration with other multiple objectives and purposes.

Though New York State's water pollution control program is mature and does not provide significant opportunities for large municipal spray irrigation systems because of systems either already built or under construction, other types of opportunities still exist. Seasonal wastewater spray irrigation provides a cost effective means of providing low flow protection and nutrient removal through land treatment, particularly in our small rural communities.

Effective implementation of our toxic control strategy and pretreatment programs will provide opportunities for the safe and cost effective land disposal of portions of the 4,900 metric tons per day of municipal wastewater treatment sludge expected to be generated in New York State by the year 2000. At this time, we are planning seven (7) potential co-disposal projects with a total capacity of 6,700 metric tons per day of refuse and sludge.(8) These projects appear to be the only reasonable cost effective alternative to the greater New York City metropolitan refuse and sludge disposal problems. Composting as well as energy recovery is being considered for these projects.

Our Department has made a strong commitment to fully develop effective land treatment systems in New York State. We have put together a multidisciplined strategy for its implementation and believe that only with an approach of this kind can we fully realize the opportunities that may exist.

Acknowledgment

The author wishes to thank staff members of the New York State Department of Environmental Conservation for their assistance in the preparation of this report and for their steady development of land treatment systems.

The assistance of Anthony Adamczyk of the Department's Division of Pure Waters is gratefully acknowledged.

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LEGISLATION

TREATMENT OF WASTEWATER IN BRITAIN, IMPLICATIONS FOR AGRICULTURAL LAND

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Land treatment was the general method of purifying urban sewage in Britain from about 1870. Because large areas of land were needed, costs were high, and offensive odours were produced, artificial methods of treatment began to be used around 1900. Many sewage farms continued for long but all had been abandoned by 1970. Modern artificial treatment gives a clear effluent discharged to rivers, and sewage sludge, which is rich in the plant nutrients N and P. The obstacle to using all sludge produced on farmland as manure is its content of heavy metals (zinc, copper, nickel and chromium) which are toxic to crops. Guidelines for using sewage sludge safely have been developed; they are based on metal contents of sludge and soil, on the crop, and on the efficient use of the N in sludge.

Intensive animal farming systems produce large amounts of semi-liquid slurries which, in UK, contain more N, P and K, and more BOD, than wastes from the human population. The only economical way of purifying these wastes, and making use of the plant nutrients they contain, is by land treatment, preferably on grass. Pollution of drainage water, and surface runoff, must be avoided. These requirements are met, and N, P and K are used efficiently, by using computer models to fit applications of wastes to the hydraulic capacity of soil, to the cycle of growth and use of the crop and to the land available.

LAND TREATMENT OF MUNICIPAL WASTES

Historical

Until 1847 there were legal prohibitions on the discharge from house-drains, water-closets or cesspools into British rivers. Such sewers as existed were only for removing surface water. The resulting pollution of subsoils, low areas of land, and wells, was accentuated by the increasing provision of piped water supplies and water closets. Conditions in large towns became so bad that discharge to rivers by sewers was legalised in 1847. This practice, largely adopted by the 1860s, however, gave no more than a breathing space and a Committee (1) reported in 1869 that it had led to "a greatly augmented pollution of rivers, which is now acknowledged to be an evil of national importance".

River Pollution was investigated by Royal Commissions from 1857. They quickly recommended that the right way to dispose of town sewage was to apply it to land but the practice was adopted in relatively few places. The British Association (1) reported in 1869 on 96 towns which had a system of water-borne sewage, only 15 used land treatment of their effluent and the Association recommended its adoption everywhere to benefit amenity, the health of the people and agriculture. Investigations made on land treatment reported included experiments with animals to measure the feeding value of crops grown and their safety, the quality of vegetables, and the health of people living on sewage farms.

The Public Health Act of 1875 forbade discharge from foul sewers into rivers (but it was many years before such discharges ceased completely). Sewage farms were established to serve all inland towns and many had lives of over 50 years. Writing in 1925 Kershaw (2) devoted considerable space in his book to land treatment, described as potentially the best way of producing high-class effluent. However, the process needed considerable areas of land and could not be hurried, particularly if irrigated crops were grown. The problems of finding sufficient land near large cities, were serious from the beginning. They were accentuated where less suitable soils had to be used, capable of dealing only with the effluent from 250-500 people on 1 hectare used for irrigation. Early difficulties were dealt with by partial treatment of sewage to settle solids by precipitation processes. Already in the last century development was begun of completely artificial processes which produced two products - sewage sludge and a clear effluent, and which used very little land. These processes began to be accepted in England in about 1896 and the Sewage Disposal Commission appointed in 1898 had the task of putting them on a scientific basis.

The recent period

Although Garner (3) writing in 1938 did not mention land treatment as an important process in Britain, some sewage farms remained in use after 1945. In 1970 Jeger (4) stated that all large-scale land treatment had been abandoned in Britain because of 1) large areas of land involved, 2) high cost, 3) smells and 4) the need to rest land between sewage applications if underground water supplies were not to be polluted. All sewage treatment (for four-fifths of the population) was by modern works, half using biological filters and half activated sludge processes, both originally developed in Britain. In 1970 23 million m^3 of water was used per day (430 l per person), about a third was used for domestic purposes. The total flow of sewage was 14.1 million m^3. As one third of British water supply is from rivers, treated sewage effluent is an important component. The minimum quality for treated effluent is 30 mg/l of suspended solids and 20 mg/l of BOD, assuming eight times dilution with river water, effluent discharged to rivers giving less dilution must be of a higher standard.

Half of the present sewage flow is from industry. The liquids range from wastes from food, drink, paper, leather and wool industries, which are easy to treat, to those from metal, engineering and chemical industries, which are difficult to treat as they upset biological processes. Synthetic detergents have increased the load of phosphorus and boron in sewage.

Sewage sludge on agricultural land

The only use of land in sewage treatment in Britain is now for disposal of sewage sludge. About 1 million tonnes of dry solids are produced annually containing 2.4% N, 1.3% P_2O_5, 0.3% K_2O. The total plant nutrients - 26,500 t of N, 14,000 t of P_2O_5 and 3,300 t of K_2O - are only about 2½% of the N, 3% of the P and 1% of the K used as fertilizer in UK. Although it is sensible to make use of these plant nutrients on farm land, only about half is used in this way; the remainder is dumped on land-fill sites, or at sea; a little is burned. Composting with town refuse has not led to products acceptable for use as manure. The main obstacle to using all sludge on farmland is that much of it is rich in heavy metals, toxic to crops and/or to man. Some sludges contain toxic chemicals or pathogens.

The agricultural value of the N, P and K in British sewage sludges, which has been partially dried, was investigated by Bunting (5). The organic matter was unimportant. The N and P were roughly as available as N and P in farmyard manure (FYM), the small amount of K was useless; on an equal weight basis FYM was superior to sewage sludge, largely because the former supplied much K. Some analyses are in Table 1.

Problems with heavy metals in sewage sludges were discussed by Webber (6). Poor growth was caused by toxic amounts of heavy metals, notably zinc, copper and nickel, occasionally chromium. He described the concept of maximum permissible applications based on "zinc equivalents" (on the basis that Cu is twice as toxic, and Ni eight times as toxic as Zn). He further assumed that the total loading should not exceed 250 mg of Zn-equivalent per kg of soil over a 30-year period. This is equivalent to 560 kg Zn equivalent/

Table 1. Analyses of Sewage Sludge and FYM (Bunting (5))

Percentages on fresh materials

	Dry matter	Organic matter	Ash	N	P_2O_5	K_2O
Raw sewage sludge	40	20	20	0.9	0.5	0.1
Digested sludge	52	23	29	1.4	1.1	0.2
Dried sludge	80	32	48	2.0	2.1	0.3
Farmyard manure	26	16	10	0.6	0.4	0.5

Table 2. Metals in Sewage Sludges and Soils

(after Berrow and Webber (7))

All data are for total concentrations in mg/kg

	Sewage sludges			Soils
	Range	Mean	Median	typical level
Ag	5-150	32	20	<1
B	15-1000	70	50	10
Cd	<60-1500	<200	-	0.1
Cr	40-8800	980	250	100
Cu	200-8000	970	800	20
Mn	150-2500	500	400	800
Ni	20-5300	510	80	50
Pb	120-3000	820	700	30
Sn	40-700	160	120	3
Zn	700-49000	4100	3000	80

Table 3. Guidelines for the Application of Liquid Digested Sewage Sludges. (DoE (8))

	Normal range of concentration*		Limit of recommended addition in sludge	Period of addition
	Soils	Liquid sludges		
	mg/kg*	mg/kg*	kg/ha	years
Zinc	10-300	1500-3000	560	30
Copper	2-100	600-800	280	30
Nickel	5-500	50-80	70	30
Chromium	5-500	100-400	1000	30
Cadmium	0.1-1.0	7-50	5	30+
Lead	2-200	200-700	1000	30+
Mercury	0.01-0.3	3-5	2	30+
Molybdenum	2	5	5	30+
Arsenic	0.1-40	7.5	10	30+
Selenium	0.2-0.5	5	5	30+
Boron	2-100	50	7(grass)	1
			5(arable)	1
Available nitrogen	-	20000-50000	525(grass)	1
			100(cereal)	1

* in dry solids

hectare or 19 kg/ha each year. In addition allowance must be made for zinc equivalents already present in the soil which could be extracted by conventional extractants. Table 2 from Berrow and Webber (7) gives concentrations of some metals in sewage sludges.

Guidelines for the use of sewage sludge on land were developed by a Working Party of the Department of the Environment (8). They concluded that the availabilities of N and P in dewatered or dried sludges were, respectively, one-third and one-half of the availability of these nutrients in inorganic fertilizers. They followed Coker (9) in saying that

85% of the N in liquid digested sludge was available. The guidelines for digested sludges are shown in Table 3. Their use should avoid the risk of damage to crops, animals or people, but they do require analyses of both sludges and soils to be available and this information is, at present, rare. No outbreaks of disease in man or animals have been traced to the use of sewage sludge. These guidelines represent the best advice at present available but they may need change as new information is acquired. Large dressings of liquid sludges applied in winter may pollute drainage with nitrate. Most heavy metals are less available in neutral than in acid soils and liming diminishes their toxicity; molybdenum and selenium are exceptions. Applications for nitrogen and boron are made on an annual basis, because they are soluble. The other elements are retained in soil and the amount applied in any one year should not exceed one-fifth of the total recommended for 30 years.

Application of liquid digested sewage sludge

Most sewage sludge applied to land has been partially dried. However Drew (10) proposed the direct use of liquid digested sludge on land to avoid drying and to save N; he stated that much inorganic N was lost by drainage and volatilisation during the drying process. Coker (9) reported tests on liquid digested sludges, applied by tanker, and supplying up to 132 kg N/ha in 70,000 litres/ha of liquid. The N in the liquid sludges was much more available than N in partially dried sludge; it was equivalent to 84% as much N in inorganic fertilizer for grass and was nearly as effective as fertilizer-N for barley. A dressing of 56,000 l/ha of liquid sludge supplied 4.8 kg/ha of zinc; this did not damage the grass crop. Coker (9) recommended that, to minimise damage from heavy metals, sludge should be spread thinly over a large area of farmland, and that dressings should not be applied frequently. DoE (8) report that application of liquid digested sludges to land has increased recently. Dry solids in the sludges do not exceed 6%. The sludges are applied direct by tanker, by rain gun or irrigation pipes. However DoE state that the cost of the tanker fleet can approach that of mechanical dewatering and disposal of drier sludge.

LAND TREATMENT OF ANIMAL WASTES

Historical

The traditional way of disposing of animal excreta is to mix them with straw and other organic bedding materials. The resulting mixture was then stacked to ferment and make farmyard manure, a relative dry material that could be spread on the land, supplying plant nutrients and organic matter to benefit soils and crops. In many European countries liquid manures were collected and used as important sources of plant nutrients. The "Gulle" system was common where a mixture of faeces, urine and litter was matured together and diluted with water. The resulting liquid was used for crops and grass. The traditional Swiss system was described by Gisiger (11). On all grass farms where the herbage was used on the farm for grazing or silage, over 80% of the N and P and 90% of the K in forage were returned to the land in liquid manures. Intensive farming (about 2½ cows/ha) was done to produce milk and meat with little need for fertilizer since plant nutrients were recirculated. On average a cow produced each day 43 litres of "Gulle" liquid manure containing 0.5% N, 0.09% P and 0.7% K. Two dressings a year were given to grass (after diluting with water) and they supplied about 130 kg N, 20 kg P and 200 kg K/ha.

In Britain the "Gulle" system was never used. In the Nineteenth Century some farmers collected urine and drainage in underground tanks and this liquid was applied to land in spring. With cheap fertilizer readily available in this century these installations were abandoned. Farmyard manure continued to be made and used but waste liquids often drained to watercourses and polluted them.

The recent period

Since 1950 the numbers of cattle in Britain have increased by 40%, pigs by 300% and poultry by 70%. Workers on farms have diminished by two-thirds but are paid much more so labour costs are much greater. Straw for animal bedding is scarce and expensive in grassland districts. Flocks and herds have become much larger. All these changes have resulted in intensive methods of livestock handling. Cattle are confined for half the year, pigs and poultry are permanently housed. Farmyard manure involves large costs in making and handling and is now much less common on British farms. Faeces and urine, plus some wash waters, from intensive animal systems are handled as liquid or semi liquid slurries. Recent leglislation made discharge of farm wastes into water-courses illegal, and the costs of disposal to public sewers (where available) became prohibitive. Farmers were therefore compelled to plan to dispose of slurries on land. Initially these wastes were regarded as a serious disposal problem, mainly because fertilizers were cheap, prices having risen little since 1955. Between 1972 and 1975 fertilizer prices trebled, largely as a result of the energy crisis, and farmers realised the value of the slurries as sources of plant nutrients. This had led to a re-appraisal in which the emphasis is on use, not disposal.

O'Callaghan *et al* (12) emphasise the problems created by increase in the number of specialist farmers with large numbers of livestock in small feeding areas, often using feeding stuffs bought into the farm, and sometimes with limited areas of land for disposal. Methods have been developed for partial or complete treatment of slurries by mechanical separation and fermentation to produce solid wastes that may be spread on land without offence from odour, plus liquid effluent suitable for discharge to a river; some processes produce methane for fuel. This paper is, however, solely concerned with land treatment of wastes.

The problem

Maximum efficiency in conversion of animal feed to product has required standardised feeding and management of intensively kept stock. There are therefore no large variations in the nature of the slurries produced; Table 4 gives typical data (from O'Callaghan *et al* (12)). The BOD of wastes varies from 15000 mg/l for cattle waste with 11% dry matter to 42000 mg/l for hen waste with 23% dry matter. The total animal wastes to be disposed of annually in UK are 57×10^9 litres containing 372000 t of N, 75000 t of P and 241000 t of K. Domestic sewage has BOD of 400 mg/l, with an annual load of 20 kg/person. The much higher BOD of animal wastes means that the problem of treating this waste is equivalent to that from a human population of 54 million - close to the population of UK. Clearly it is impossible to duplicate existing sewage facilities to treat animal wastes - the income from each animal is much too small to provide a margin for capital or running costs of sewage works ! There is no alternative to land treatment.

The value of these wastes must be realised; in terms of fertilizers (at 1978 prices for UK) they are worth \$182m for N, \$92m for P and \$68m for K, a total of about \$340m. The N in wastes is more than a third of that used in fertilizers, the P about half and the K about three-quarters of fertilizer K. Estimates of nutrients in human and animal sewage are compared in Table 5.

Constraints

Water pollution must be avoided. No effluent discharged to rivers may exceed 20 mg/l of BOD and 30 mg of suspended solids. Even where BOD is oxidized and suspended matter is removed in passage through soil, unless the N applied matches uptake by crops, drainage will contain excess nitrate that causes entrophication in water courses (leading to excessive biological growth) and gives undesirably high nitrate in drinking water. Drainage water passing

Table 4. Amounts of Livestock Wastes produced annually and their Contents of BOD and N, P and K

Animal	No.	Annual Production kg	BOD kg	Total N kg	Total P kg	Total K kg
Fattening pig	1	1660	36	10.5	2.7	3.2
Laying hens	100	4550	190	68	25	18.7
Young cattle*	1	6660	100	34	4	31
Cow*	1	11700	176	60	7.3	55

* housed for 5 months of the year

Table 5. Nutrients in Human and Livestock Sewage, (estimates from several sources)

	Human sewage			Livestock sewage
	Total content	Effluent (to rivers)	Sludge (to land)	Total quantity (to land)
	Thousands of tonnes			
N	151	100	26	372
P	64	50	14	75
K	59	50	3	241

to rivers or to deep aquifers must not contain more than 22.6 mg/l of NO_3-N, WHO's maximum 'acceptable' level.

Surfaces runoff must also be avoided since this conveys BOD plus N and P to streams with consequent damage to amenity and water quality. Excessive dressings may run off sloping land; heavy rain may cause runoff if it follows slurry application.

To satisfy these conditions manures treated as liquids must not be applied in UK in winter to soil that is frozen, or is at field capacity. Therefore storage must be arranged for slurry produced in winter. Dressings must also match the capacity of crops to use nutrients.

Odour is a problem in areas where people unconnected with farming live near to land on which wastes are used. O'Callaghan *et al* (12) consider that current research on treating slurry by mechanical and biological methods is most useful in lessening odour. Treated effluent may be spread on land without offence and separated solids may be transported for use on other farms or gardens.

Many of the problems of land treatment of domestic sewage that led to its being abandoned in UK in favour of artificial treatment have reappeared in exaggerated forms in the work done during the last 15 years (16). Although the volume of sewage wastes from animals are only a tenth of those from the human population the total BOD is as great. We have, overall, adequate land to accept the wastes. Problems are greatest where large animal enterprises are sited with limited land of heavy texture near urban districts in wet areas; they are least in rural areas and on permeable soils in drier climates.

Requirements for satisfactory treatment

1) The capacity of the soil to absorb the liquid must not be exceeded. The water applied must do no more than balance the moisture deficit on the day of application. Hydraulic loadings vary with soil type and rainfall; moisture deficits vary in summer from 180 mm in eastern England (dry area) to only 10 mm in western Wales (wet area). Spreading is not permissible in wet weather (when it may be impossible if tankers are used).

2) Odour problems may have to be avoided by pre-treatment or by injecting slurries into soil.

3) The plant nutrients applied should match the needs of the crop to be grown. Needs vary from 300 kg N, 30 kg P, 300 kg K/hectare for grass to 150 kg N, 20 kg P, 150 kg K/hectare for wheat; sugar beet and potatoes have intermediate needs. Intensively used grassland may receive up to 500 kg N/ha in a year. However the large amounts of K which accompany so much N are taken up "in luxury" by grass so depressing Mg intake and leading to a risk of hypomagnesaemia in grazing cattle. If grass contains more than 2% K in dry matter, a supplement of magnesium oxide must be fed.

Some losses of N by denitrification and volatilisation of ammonia may be expected, but this cause of inefficiency depends on local conditions of weather, soil and waste and cannot be forecast accurately.

4) Toxic effects must be avoided. Heavy dressings of slurry can damage all crops and grass may be checked in growth or killed. Copper is often added to pig diets to stimulate growth; the copper-rich slurries may damage sensitive crops, they may also injure sheep where these graze subsequently and ingest contaminated surface soil (sheep are particularly liable to copper poisoning). Palatability of herbage is also diminished by recent dressings of wastes.

5) The crop should be suitable for farming where animal wastes are regularly applied. Human food crops should

be monitored to avoid contamination by pathogens. Grass is the most suitable crop to receive slurries. It grows for all of the season when temperature is high enough, it takes up more N, P and K than other common British crops. Grassland carries wheeled traffic best for spreading. Several dressings may be applied in a year, this is not possible for arable crops.

Grazing is not the best way of using grassland that receives much slurry at regular intervals. Most of the N and nearly all P and K in grass eaten is returned in excreta of grazing stock - making further P and K from slurry unnecessary. The treated grass is best used for cutting and feeding fresh, or for silage, or hay. Unless there is a gap of 6 weeks between spreading and grazing, health of stock may suffer, and grass will be unpalatable.

The large amounts of N, and its organic origin (leading to slow release), make slurry unsuitable for growing high quality wheat and barley as crops are liable to lodge. Slurry is, however, suitable for producing maize for silage.

Management systems

The requirements for good management of wastes are: 1) that plant nutrients are fully used by crops to be grown; 2) water pollution is avoided. The first requirement is settled by knowledge of the needs of the crop and the use to be made of it, soil composition and concentrations of nutrients in the wastes to be applied. The second requirement is settled by comparing evapotranspiration and rainfall to calculate soil moisture deficit; this is called "permissible hydraulic loading rate". A model for long-term management of animal wastes was developed by O'Callaghan *et al* (12), Dodd *et al* (13) and Woods and O'Callaghan (14); the practical use of the computer program for this model was described recently by Parkes and O'Callaghan (15), see Appendix.

Much recent British work to develop management systems for wastes has been summarised by ARC (16) who report tests of systems where large amounts of animal wastes were spread. There was some leakage to the drains after an application of slurry, particularly when rain followed; this usually lasted about 12 hours and measurements on drainage water had returned to normal background after 36 hours. Less than 2% of very large dressings applied (310 t/ha) appeared in drainage, but grass was scorched, and some was killed where solids were deposited. Within a month the grass grew again and colonised the bare patches.

Bacteriological hazards

ARC (16) reports that as a result of applying much pig slurry during the summer months to grassland on sandy clay loam, organic pollution of drainage water in winter was not significant but 10^5 *E. coli*/litre were detected. In other experiments slurry from pigs fed on whey was spread at a rate equivalent to 292 mm slurry/year; large concentrations of organic and microbial pollutants were found in drainage for a few days after application. To minimise, and better to prevent, such pollution of drainage the hydraulic capacity of the soil must not be exceeded - as it was in these last experiments. Nevertheless drainage water from slurry-treated areas must be regarded as a potential source of human and animal pathogens.

Conclusions

If farmland is used as a dump for animal wastes at rates which exceed the moisture deficit at the time, and the capacity of the crop to use N, (and P and K) drainage water is liable to be polluted and plant nutrients will be wasted. The overall management of hydrological regions should include the production and use of animal wastes. Nutrient balances for regions should be calculated; a surplus of nutrients in an area should be solved by diverting wastes to other areas with a deficit, as Dodd *et al* (13) suggest. Plans for new farming enterprises, or changes to existing farms, should include arrangements for disposal of wastes. Intensive animal farming should not be started in areas which do not have access to sufficient land for spreading wastes.

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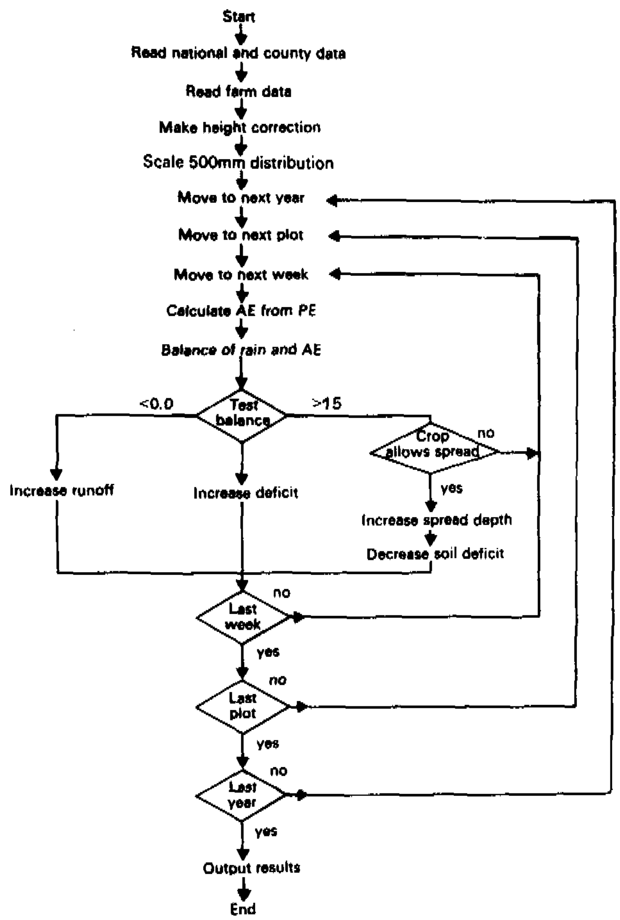
Appendix

LAND SPREADING MODEL

The land spreading model developed by O'Callaghan, Pollock and Dodd (17) was quoted by ARC (16) from which the figure below is taken. The flow diagram indicates the relevant steps in calculating the amount of liquid manure the farm are established. Starting with the land at field capacity in spring these data are compared with historical weekly rainfall to show either growing soil moisture deficit, or runoff if the soil becomes saturated. If the balance indicates a moisture deficit, spreading should only proceed if the negative balance is greater than a certain amount, application with a smaller



Simplified flow diagram for land spreading model.

that may be applied. National and local evapotranspiration data are published as averages for the summer months, as are average rainfall data. A matrix is available to relate potential evapotranspiration (PT) to actual evapotranspiration (AE); these estimates are corrected for the height of the farm above sea level. Total acreage of farm, numbers of stock, number and size of fields and cropping pattern are used in go/no go matrices to determine where spreading is permissible.

Weekly evapotranspiration data for deficit would be unsafe. O'Callaghan *et al* (17) assume the safe figure for deficit as 15 mm (0.6 in). With smaller values, no spreading is done; if the deficit exceeds 15 mm, 2.5 mm (0.1 in) of slurry is spread (equivalent to 25 t/ha) to reduce the deficit to 12.5 mm (0.5 in). These calculations are made for each field in turn until the whole farm has been covered.

COMPARISON OF HEALTH CONSIDERATIONS FOR LAND TREATMENT OF WASTEWATER

OVERVIEW --- HEALTH CONSIDERATIONS ASSOCIATED WITH LAND TREATMENT OF WASTEWATER SYSTEMS COMPARED WITH OTHER HUMAN ACTIVITIES

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ABSTRACT

Conceivable and potential health hazards arising from presence of toxic or carcinogenic chemicals and pathogenic microbial agents in wastewaters in connection with land disposal of such wastewaters are presented in broad terms; certain relationships of these hazards to those posed by occupational or environmental exposure to similar pollutants are discussed. The question is raised as to how great a hazard is really posed by the low concentrations of chemical contaminants or low numbers of microbial pathogenic organisms with which man comes in contact during the course of his normal household or recreational activities as compared to his occupational or environmental exposure to these same agents. It is questioned whether an aim of modern technology should be to achieve a truly zero-level or no threshold level of chemical or biological pollutants and whether cost-benefit ratios would not preclude or make infeasible the complete removal or destruction of such pollutants in wastewater, especially when such an ideal exceeds the purity of the natural environment to which man is well conditioned in the evolution of modern civilization.

TEXT

This session of the Symposium is concerned with consideration and assessment of potential public health problems that may arise from land treatment of wastewater systems as compared to similar potential hazards associated with other human activities. Such assessment must of necessity include the disease potential of microbiological as well as chemical components actually or potentially present in wastewaters. Inasmuch as subsequent papers will consider the biological and chemical aspects of the problem in some detail, we wish to direct our presentation towards a general overview of the field and to put into a broad, and perhaps somewhat different, philosophical perspective some of the speculations, hypotheses and conclusions of which you have undoubtedly heard much in other forums.

Man's pollution of his environment goes far back in time, and early on, in clan or tribal societies, it ostensibly presented no great problem. Tribes could simply move to new locations and leave their wastes to natural recycling processes. It was not until man abandoned his nomadic hunter-gatherer existence and adopted a communal life that accumulation and disposal of human excreta (and other man-generated wastes) created an increasing problem. For centuries and until comparatively recent times, household wastes and excreta were simply dumped into open drains which ran down streets and eventually collectively terminated in running streams or open bodies of water. Sewerage systems, in the modern sense, were a development of the late eighteenth century and did not antedate by very much the Industrial Revolution which added its own increment to the pollution problem via the chemical components of industrial wastes. Over the succeeding years, the tremendous technological advances which created for Western man the highest standards of living ever experienced, also added appreciably to pollution of his environment. Harmison (1) points out that as of today (1978), "an estimated 2 million recognized chemical compounds with more than 30,000 chemical substances are in commerce, and approximately 1,000 new ones are introduced each year". Such statistics buttress statements that we live in a sea of chemicals, and give rise, rightfully, to concern about the presence of toxic chemicals in our air, water and food.

The development of indoor sanitary facilities together with municipal sewerage systems served effectively to clean up the immediate household and communal environment but unfortunately gave rise to a parallel pollution of surface waters which were the most convenient sites for sewage disposal. The consequences were what hindsight would now lead us to expect, namely, a marked increase in waterborne infectious disease. This situation was subsequently to a large degree brought under effective control through the development and application of water purification methods. However, the environmental concerns stemming from a burgeoning population and increasing urbanization have brought a growing recognition that, even with conventional treatment, the discharge of human wastes into surface waters may be generating a new phase of unacceptable risks which necessitates consideration of alternative disposal methods. Land application is one of these.

It has been pointed out (2) that with the increase in the population over the coming years in the United States the increase in the demand for water by cities, industries and agriculture will not merely parallel, but will increase disproportionately to, the population increase. Bylinsky (2) has estimated that as of 1970 such water uses were approximately 1325 billion liters per day (BLD) (350 billion gallons per day (BGD))and that by 1980 that figure would reach 2370 BLD (600 BGD) and by the year 2000 would be in the vicinity of 3785 BLD (1000 BGD). As the latter figure conservatively exceeds the available supply of fresh water, estimated at 2460 BLD (650 BGD), it appears evident that the disparity between availability and demand will be resolved only through stringent conservation practices including the reuse of wastewaters.

Many regions of the world, although amply possessed of water resources, are actually short of fresh water; this results not from profligate use of water for the usual domestic, agricultural and industrial purposes but from the discharge, in large amounts, of polluted water into streams and lakes, thus rendering these recipient waters unuseable (3). Shuval (4), among others, points to the need to reuse wastewater as part of any conservation aim. The importance of water conservation and wastewater reuse is well illustrated by a recent news item (U.S. News and World Report, May 9, 1978) that in the arid West and Southwest of the United States there is intense competition among states, cities and agricultural interests for a diminishing water supply. The huge new water usage attending development of shale-oil production and coal-slurry pipelines can result only in a severe exacerbation of the prevailing water shortage problems. The use of wastewaters to supplement and/or replace use of fresh water for irrigation and for such industrial purposes is a logical step in water conservation, with the proviso, of course, that such useage does not present unacceptable risks to the public health.

Land application of wastewater for disposal and for agricultural purposes is no new development; it dates back to the late nineteenth century (4,5), occurring as a spinoff from the expanding development of sewage collection systems. By the early part of this century, this agricultural practice was adopted by numerous farms in both Europe and the United States (5), and especially in the arid Western and Southwestern U.S. (e.g., Wyoming, Colorado, California, Utah and Montana); aside from providing agricultural nutrients, the practice served for land disposal of domestic sewage. In California the use of raw sewage for irrigation of food crops has been prohibited since 1918. Regulations permitting irrigation of food crops eaten raw were first developed in 1933. In 1968, and again in 1975, these criteria were revised and comprehensive regulations were enacted allowing the use of reclaimed wastewaters under conditions which would not prove inimical to the public health. Over the succeeding years, the California Department of Health has looked upon treated domestic wastewater as a valuable resource and has encouraged its use, not only for crop irrigation but also for landscape irrigation (e.g., greenbelts, golf courses, parks, etc.), recreational impoundments and, in some cases, groundwater recharge. According to Crook (5) some $246.6 \times 10^6 m^3$ (200,000 acre-feet) of wastewater are reclaimed annually through planned operations, an amount which represents about 7% of the wastewater produced. In addition more than $616.5 \times 10^6 m^3$ (500,000 acre-feet) are recovered incidentally through return to streams and groundwater basins, thus contributing to the pool available for reuse. The amount of wastewater available for reclamation is estimated at $2,096.1 \times 10^6 m^3$ (1.7 million acre-feet) annually, equal to 5% of the current total water demand of the State.

This is a not inconsiderable resource, and water reclamation is increasingly looked upon as an important technic to meet future water needs. It thus becomes necessary to ascertain the presence, nature and magnitude of any potential hazards to the public health and to control any threatened hazards through administrative and technological means. The potentially hazardous contaminants found in wastewater provide, in effect, a mirror of the diverse health risks or problems which already exist in the community which gave rise to the wastewater. The extent and degree of potential hazards posed by chemical or biological contaminants in wastewater should be assessed against a relevant yardstick, such as exposures to similar substances or agents encountered during the course of other domestic or occupational activity as well as other environmental exposures.

With respect to toxic chemicals, their presence in wastewater is essentially a direct reflection of man's activities which generate industrial-production wastes, refinery wastes, disperse agricultural fertilizers and pesticides into the environment, etc. Each of these activities carries its own array of occupational hazards or environmental exposures which are, or need to be, a primary focus of health protection.

In addition to the potential hazards posed in land application of wastewater by the presence or accumulation of contaminants on the soil surface, there is the associated problem posed by their dispersion or dissemination downward through the soil by percolation, or laterally as other physical conditions determine. There are, thus, potential difficulties or hazards on several counts, including contamination of drinking water supplies.

For our purposes, chemical contaminants which may be present in wastewaters and which, following land application of such waters may conceivably appear eventually in groundwaters, are considered in two broad categories, viz., inorganic solutes and organic solutes.

Let us first consider the inorganic solutes. In its Summary Report on Drinking Water and Health (6) the Committee on Safe Drinking Water of the National Research Council, U.S. National Academy of Sciences, presents an extensive review and evaluation of potential health hazards referrable to the occurrence of 16 metals in drinking water. For 6 of these, a group comprising barium, cadmium, chromium, lead, mercury and silver, maximum allowable concentrations have been set in the Interim Primary Drinking Water Regulations.

Eight metals (chromium, cobalt, copper, magnesium, maganese, molybdenum, tin and zinc) are essential components of the human diet in trace amounts. On the other hand, any of these trace metals can be toxic for man if intake is excessive. As toxicity is manifested only with the intake of concentrations greater than those allowable or acceptable in drinking water, adverse health effects are generally associated with occupational exposures. Thus, in the case of nickel, epidemiological observations suggests that occupational exposure via the respiratory tract increases the risk of pulmonary and of naso-pharyngeal cancer. Long continued inhalatory exposure to beryllium-containing dusts may cause pulmonary sarcoidosis (there is no evidence that it produces pulmonary cancer). Similar comments can be made with respect to the toxicity of certain of the other trace metals reviewed in the NRC Report (6) when dietary intake is excessive (e.g., cadmium and Itai-Itai disease in Japan) or when there is prolonged occupational exposure to large concentrations. Man is not alone in introducing inorganic solutes into the environment; nature also plays a contributory role. Arrhenius (7) mentions an area in the USSR in which enzootic lung and stomach disease occurs in sheep ingesting river water containing unusually high naturally-occurring concentrations (0.2-2.0 mg/L) of boron; he also cites the natural occurrence of toxic concentrations of mercury, generally looked upon as a man-made contaminant, in an Indonesian river. These are perhaps extreme examples, but if recycling of water serves to superimpose additional amounts of trace elements on the concentrations derived from natural sources, the risk of adverse public health effects is sharpely escalated unless these contaminants are removed or, preferably, contained at the source.

When we come to the organic solutes, additional considerations enter into evaluation of toxicity and assessment of health effects. Practically, there are two approaches to assessment of the potential acute or chronic ill effects engendered by exposure to chemical contaminants in water, viz., (a) the epidemiologic and, (b) quantitation of toxicity in laboratory animals. In the case of non-carcinogenic substances, "toxic threshold" is defined as the largest dose which can be tolerated without production of ill effects. Lead poisoning provides an example of a disease where the individual can support a subthreshold or "non-zero" body burden without illness; it is only when the threshold level is exceeded that disease supervenes. In the case of carcinogenic substances, on the other hand, estimation of risk of exposure is based on the assumption that there is no threshold, i.e., the dose response relationship follows a straight line. Thus, under this assumption, if a given dose of an agent produces cancer in 1 out of 100 laboratory animals, or persons, reducing the dose 1,000-fold would still produce cancer in 1 of 100,000 subjects. Hence, this theory, holding that nothing more than a "zero-level" of a chemical contaminant is permissible in potable waters, calls for extremely rigorous safeguards to assure that land application of wastewater does not permit the slightest contamination of groundwater supplies.

Since some organic compounds which may find their way into groundwaters and drinking water supplies (e.g., the pesticide Kepone) might, in sufficient doses, result in chronic disease, experimental determination of threshold levels would seem relevant if attempts are to be made to hold the environmental concentrations below toxic levels. However, attention appears to focus primarily on those substances which have occupationally been implicated in cancer or suspected of being carcinogens. The corollary question then becomes whether these same substances in the low concentrations encountered in drinking waters can also lead to cancer in man. Extrapolation to man of findings and data from laboratory studies of carcinogenicity requires caution and an understanding of the limitations of the experimental approach. At this point a quotation from the National Research Council Report ((6) page 46) seems pertinent, to wit, "Because the bioassays that have been used to establish carcinogenicity of certain organic chemicals are conducted at doses which are hundreds to thousands of times greater than the levels at which these chemicals occur in water, the risks at these low levels must be obtained by extrapolation from higher doses. There is no hard evidence that low-level oral exposure to any of these chemicals produces cancer. An argument has been made that dose levels used to establish carcinogenicity (in animal experiments - Au.) are so high that they overwhelm normal detoxification or repair mechanisms or both, and produce cancer by some mechanism that does not operate under low dose conditions". Extrapolation may be misleading if no account is taken of animal species differences with respect to carcinogen activation, metabolic pathway, etc. Just how this problem should be dealt with is left to other speakers who will be discussing in greater depth later in this session the health effects of chemical pollutants arising from land application of wastewaters.

Similarly, since microbial pathogens will also be discussed at some length, our comments on the whole will be broad and general and concerned with the tenet that wastewater will contain significant numbers of infectious agents such as typhoid and shigella bacteria, wild poliomyelitis virus, hepatitis virus, etc., in direct relatiohship to the prevalence in the community of individuals infected with these agents.

Historically, attention to water as a source of human disease focused on the microbial pathogens, generally classifiable as either bacterial or viral.

Developing sanitary engineering practices, and especially the introduction of chlorination in the early part of this century, served to bring waterborne bacterial diseases essentially under full control. This does not mean that these diseases have been eradicated. The causal pathogens may still be encountered in sewage, their presence reflecting at any given time the microbial flora of the community. This becomes a central factor in the reclamation and use of reclaimed wastewaters, and obviously some degree of antimicrobial treatment is required before the effluents can be safely utilized. Logically, therefore, the quality standards that were developed for drinking water are now being applied to reclaimed wastewater.

The degree of treatment required increases directly in relation to the extent and nature of human exposure. This is illustrated in Figure 1 (kindly provided by Dr. James Crook, California Department of Health) which also summarizes the requirements established by California under the Wastewater Reclamation and Reuse Law of 1967 regulating the application of reclaimed wastewaters for specific purposes; thus, at the bottom of the treatment scale are primary effluents which may be employed without further treatment for the surface irrigation of orchards and vineyards and for non-food crops such as fodder, fiber and seed. The central portion of the figure shows the applications that require wastewater to be oxidized and disinfected by chlorination to the extent that the median MPN limit of total coliform organisms does not exceed 2.2 per 100 ml or 23 per 100 ml depending upon the use purpose, as shown. The requirement that reclaimed water for landscape irrigation must not exceed a median MPN limit of 23 total coliform organisms per 100 ml is directed to large landscape areas, e.g., golf courses, freeway median strips, where application can be controlled and public exposure is low. Similar expanses of landscape which lie both within an urban area and are intensively utilized by the public, e.g., parks and athletic fields, may pose a greater hazard to the user and hence more stringent standards are required. Thus, wastewater treated as shown in the lower section of Figure 1 results in a water of high quality by bacteriologic standards and which

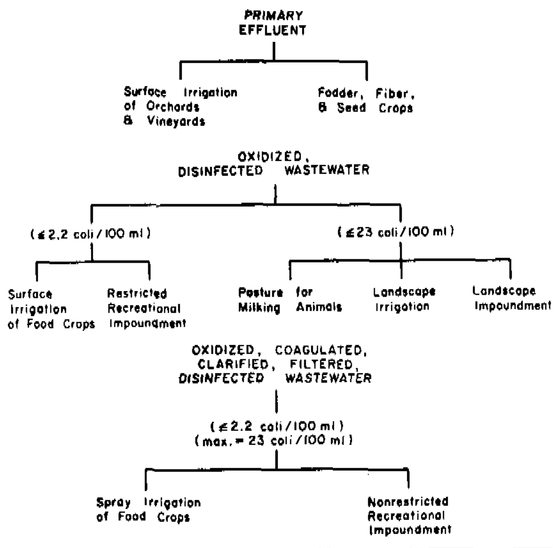

FIG. 1 - Wastewater Reclamation Quality Requirements

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should be safe for irrigation of food crops and for unrestricted recreational use.

These standards require that the treatment process be consistently and reliably carried out, an objective not always achieved (8, 9) for various reasons. Absent or inadequate disinfection of wastewater can thus lead to a potential or actual health hazard. Microbial agents surviving the chlorination process would ostensibly give rise to potential health problems. We say ostensibly because in California, where over 85% (more than 209.6 x $10^6 m^3$ per year (170,000 acre-feet per year)) of the wastewater applied to land is used for non-food crop irrigation, no adverse health effects have been shown to occur (10). However, more precise information is needed with respect to possible infection hazards posed by exposure to aerosols generated by spray of treated wastewaters, and of possible contamination of groundwater supplies. The risk of infection through exposure to aerosols generated during spray irrigation with inadequately treated wastewater is well illustrated by the recent work of Shuval and his associates (11), who observed that the incidence of infectious hepatitis and of shigellosis, salmonellosis and typhoid fever were from two to four times higher in the communities using treated non-disinfected wastewater for spray irrigation as compared with those communities which did not practice any type of wastewater irrigation.

Two recent reviews (12, 13) on the health significance of microorganisms in aerosols from wastewater treatment processes and spray irrigation point out that the available evidence neither supports nor denies the existence of potential infection hazards (cf. 12) and that additional epidemiologic studies to resolve this point are desirable.

With respect to land application, bacterial pathogens (and perhaps viral pathogens) may survive in soil for a matter of several months (14) and conceivably could accumulate at the soil surface through repeated application of wastewaters over long periods of time. Normally, removal of both bacterial and viral agents occurs at, or within a few feet below, the soil surface (13, 14), but the nature and composition of the soil and geological formation may determine possible entrance into groundwaters. Removal of viruses is largely by adsorption to soil particles and is affected by soil concentration, pH, flow rates, etc., and the interplay of such factors may result in the elution of fully infective virus from the adsorbent and thus allow further carriage or dispersal of the virions. Along these lines, the field studies of Gilbert et al (15) showed that viruses were removed by soil filtration and did not move into groundwater. On the other hand, Wellings et al (16) found that viruses present in treated effluent discharged onto soil moved both vertically and laterally and appeared in nearby wells. (This interesting and important study also pointed to the fallibility of the coliform bacterial count as a measure of viral inactivation.)

This brings us to a final consideration, namely, if viruses do reach groundwater and, via this route, eventually occur in miniscule numbers in finished drinking water, what is their significance for human health? Since our experience and expertise fall primarily within microbiology rather than chemistry or toxicology, we shall use a microbial agent to exemplify the problem of assessing health effects, although the issues and philosophy are equally relevant to chemical contaminants.

The role of epidemiology has been, and still is, to sharpen our understanding of mechanisms of disease transmission so that intelligent distinctions can be made between agents which readily utilize the waterborne route of infection to maintain or to amplify their occurrence in the community, and those agents whose presence in water is of little or no epidemiologic significance in terms of disease transmission.

Thus, when poliomyelitis virus was first recognized in sewage in the early 1940's and about the same time was recovered from flies which had fed on infected feces, it was presumed that poliomyelitis was yet another enteric disease transmitted from feces via flies and polluted water. However, despite intensive study and epidemiologic investigation over the succeeding two decades, a period during which epidemics of the disease were occurring almost annually in the United States and Western Europe, virtually no clearcut events implicating water as a vehicle of transmission were documented (17). The viral sources for disease transmission and, incidentally, of virtually all the poliovirus in sewage, were the 100 or more persons who underwent unrecognized infections for every person with recognized clinical disease; thus, in a year in which there were 20,000 recognized cases of poliomyelitis, there were concurrently some 2 million or more persons freely mingling in the general population, excreting virus and thereby directly exposing their associates and other contacts.

Studies on the natural history of poliomyelitis have shown that the role of virus in polluted water, together with other environmental sources of fecal virus exposure in communities or regions with poor sanitation, was to bring about a protective immunologic conditioning of the population at an early age, so that clinical poliomyelitis was a rare occurrence. Thus, we have the irony that epidemic paralytic poliomyelitis did not appear in Western Europe and the United States until our improved levels of sanitation restricted poliovirus as a natural component of the microbiological flora to which infants are regularly exposed within the first few months of life. To control the disease in the United States, we now simulate the natural conditions of 100 years ago by feeding infants oral poliovirus, in the form of attenuated virus strains, during the first few months of life. Other examples of how immunizing doses of microbial agents protect man against overt disease are suggested by Melnick (18), who found many years ago that sewage workers "had the lowest amount of absenteeism among all the occupational groups studied. It appeared that sewage workers were regularly immunized by their exposure to small amounts of infected material" (see also (19, 20)). This parallels what is done in the laboratory, viz., animals are actively immunized against overt or lethal infection through the administration of small amounts of the agent. This leads us, as it did Melnick, to "wonder whether very tiny amounts of virus in water would be a public health hazard" (18) or simply another facet of man's innocuous, if not beneficial, microbiological environment.

What is the relevance of these comments to the present day health concerns in the use of wastewater? It is our philosophy that in struggling with the complexities of establishing goals or standards for treatment of recycled wastewater in order to provide reasonable assurance against health risks, we should be wary of insisting that nothing less is acceptable than the ultimate degrees of purity which modern technology can achieve. This technology is immensely costly and should be reserved for those situations in which there is a genuine and demonstrable need. Because modern technology permits measurement of such minute traces (e.g., one part in a trillion) of many chemicals and the detection of a single infective poliovirus particle in 378.5 liters or 100 gallons of water (1 to 3.3 trillion on a volume basis (21))* it does not necessarily follow that public health will be better protected or the public interest better served by embracing such sweeping goals as the complete removal of all viruses of human origin from any waters that man may contact or equivalent ideals such as "zero-levels" and "no-threshold levels" of toxic or carcinogenic substances. We have in these two situations what McLean (22) has so aptly stated, namely, "A conflict of decision between risks that are immeasurably small, but which cannot be proved absent, and a certainty of costs" and further "some groups.....seem to believe that every chemical banned represents a victory, and that the efficiency of a regulatory agency is judged by how many compounds it bans". The same might be said of viruses.

*This is "equivalent to detecting a submerged ping-pong ball in 20 billion gallons (or 75.6 billion liters -- Au.) of water" (21).

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COMPARISON OF HEALTH CONSIDERATIONS FOR LAND TREATMENT OF WASTEWATER

INFECTIOUS DISEASE POTENTIAL OF LAND APPLICATION OF WASTEWATER

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The question of health effects of land application of wastewater is examined in terms of (1) measurable risk and (2) the acceptability of such risk. A brief and selective review of the literature is combined with experience from the authors' laboratory to present what can be described as a personal position paper on this topic.

INTRODUCTION

When one is asked to speak or write of the health effects of such practices as land disposal of domestic wastewaters and sludges, the question implicitly (and often explicitly) is directed to the risk or safety of the practice. Lowrance (23) has written

> "that two very different activities are required for determining how safe things are: measuring risk, an objective but probablistic pursuit, and judging the acceptability of that risk (judging safety), a matter of personal and social value judgment."

This statement stresses the relativistic and judgmental nature of the task. We have focused first on the objective portion of determining the safety of land disposal and thus sought some measure of what biological agents may be distributed by land disposal, what the probable survival times of these agents are and what is their probable ultimate fate, i.e. decay, immobilization, ingestion.

PATHOGENS OF CONCERN

In a recent review of the health hazards associated with wastewater effluents and sludges, Akin et al. (1.a.) listed the organisms of major concern which *may* be present in raw sewage from U.S. communities (see Table 1).

Healy and Visvesvara (1.d.) concurred in the conclusion of Akin et al. that the major parasitic cause of water-associated cases of diarrhea in the U.S. at this time is the protozoan *Giardia lamblia*. From their 1977 paper is taken a listing (Table 2) of published cases of *Giardia* infection in which water was implicated. In the 1969 Aspen outbreak, organisms were not isolated from the town water supply, but *Giardia* cysts were recovered at two points where the sewage entered the treatment plant (water-sewer contamination had been found). In the 1977 Rome, NY outbreak of *Giardia* infection, organisms were isolated from the water supply, microscopic identification made and clinical illness produced experimentally in beagle puppies. Healy and Visvesvara suggested that the water used for drinking and bathing in many parts of the U.S. is a potential source of *Giardia*. Further, other *Giardia* species causing illness to man may be harbored by domestic and wild animals (beavers have been implicated). Wolfe (31), reviewing the epidemiology of *Giardia*, stated that he has recognized infections contracted in 97 different countries. Although tap water was implicated as the main source of infection in Leningrad, infection may also be initiated by the hand-food-mouth route (with the food handler an asymptomatic cyst passer) in the Soviet Union and other areas of increased risk, including Southeast and South Asia, West and Central Africa, Mexico, Korea, and western South America.

Healy and Visvesvara noted that free-living amoebae such as *Naegleria fowleri* and members of the genus *Acanthamoeba*

TABLE 1.

Major Organisms of Health Concern That May be Present in Sewage from U.S. Communities

ORGANISMS	DISEASE	RESERVOIR(S)
I. PROTOZOA and HELMINTHS		
Balantidium coli	Balantidiasis	Man, Swine
Entamoeba histolytica	Amebiasis	Man
Giardia lamblia	Giardiasis	Man, Domestic and wild animals?
Nematodes		
Ascaris lumbricoides	Ascariasis	Man, Swine?
Ancylostoma duodenale	Ancylostomiasis	Man
Necator americanus	Necatoriasis	Man
Ancylostoma braziliense	Cutaneous Larva Migrans	Cat
Ancylostoma caninum	Cutaneous Larva Migrans	Dog
Enterobius vermicularis	Enterobiasis	Man
Strongyloides stercoralis	Strongyloidiasis	Man, Dog
Toxocara cati	Visceral Larva Migrans	Carnivores
Toxocara canis	Visceral Larva Migrans	Carnivores
Trichuris trichiura	Trichuriasis	Man
Cestodes		
Taenia saginata	Taeniasis	Man
Taenia solium	Taeniasis	Man
Hymenolepis nana	Taeniasis	Man, Rat
Echinococcus granulosus	Unilocular Echinococcosis	Dog
Echinococcus multilocularis	Alveolar Hydatid Disease	Dog, Carnivore
II. BACTERIA		
Salmonellae (Approx. 1700 types)	Typhoid Fever Salmonellosis	Man, domestic and Wild Animals and Birds
Shigellae (4 spp.)	Shigellosis (bacillary dysentary)	Man
Escherichia coli (enteropathogenic types)	Gastroenteritis	Man, domestic animals
III. ENTERIC VIRUSES		
Enteroviruses (67 types)	Gastroenteritis, heart anomalies, meningitis, others	Man, possibly lower animals
Rotavirus	Gastroenteritis	Man, domestic animals
Parvovirus-like Agents (at least 2 types)	Gastroenteritis	Man
Hepatitis A virus	Infectious Hepatitis	Man, other primates
Adenoviruses (31 types)	Respiratory disease, conjunctivities, other	Man

TABLE 2.

Implicated Waterborne Giardiasis

Place	Source	Cases	Potential Source
Portland, Oregon, 1969	Water*	500	50,000
Aspen, Colorado, 1969	City Water*	298	7,000
Rome, New York, 1977	City Water*	350	50,000
Camus, Washington, 1977	City Water*	128	6,000
Berlin, New Hampshire, 1977	City Water	205	15,000
Estes Park, Colorado, 1977		9	42
Resort Lodge, Colorado, 1976		13	40
Unita Mts., Utah, 1976	Mountain Stream	34	7

* *G. lamblia* cysts recovered from water.

also cause disease, although relatively infrequently. They cited extensive evidence that pathogenic *Naegleria* can be isolated from almost every type of water system (Table 3). *Acanthamoeba* species have been isolated from sewage dumpsites from ocean sediment as well as in brackish waters. The authors stated that these protozoa often are present in source waters and after treatment as well as in wastewaters and sewage sludges. They considered them to be of increasing importance in any health effects assessment.

TABLE 3.

Isolation of Pathogenic Naegleria Sp.

Number Strains Isolated	Source	Country
1	Lake water	Virginia
1	Tap water	South Australia
1	Domestic water supply	South Australia
1	Swimming pool	Belgium
5	Sewage	India
2	Tap water & fire hydrant	South Australia
1	Sewage	U.S.S.R.
24	Thermal Canal	Belgium
1	Warm physiotherapy pool	England
2	5 Thermal lakes and rivers	Florida
22	Thermal water-factory	Belgium
1	Sewage	South Korea
4	14 Fresh water lakes, 1 Thermal lake	Florida

Although the prevalence of infection with parasitic helminths has decreased with modern sanitary practices, about 30,000 positive stool specimens were reported by state health laboratories in the U.S. in 1976 (the last year for which published CDC data were available at the time of writing). Akin et al. estimated that over a million persons harbor ascarids. In communities with immigrants from more tropical countries, whipworm (*Trichuris trichiura*) eggs may be concentrated in municipal sludges (17).

Hookworms, too, may present a problem in warm humid areas where the worm larvae can embryonate in the soil before penetrating the skin. This is an obvious hazard in climates where going barefoot is a common practice. Helminths of pets, whether roundworms or hookworms, also have the potential of causing a usually mild disease in humans where they have an incomplete developmental cycle.

In addition to those organisms listed in their table, Akin et al. noted that other bacteria such as *Clostridium*, *Mycobacterium* and *Yersinia* species may be present in sewage but are probably not at a concentration sufficient to cause disease via sewage exposure. Guentzel (1.c.) has reported the concentration of *Clostridium perfringens* and of *Mycobacterium* species in a non-chlorinated secondary effluent to be about 10^5/liter and 10^6/liter, respectively; no *Yersinia* species were isolated in his study. Parenthetically, it should be noted that Yersiniosis has been recognized much less frequently in the U.S. than in Northern Europe, Japan and Canada, although *Y. enterocolitica* has been implicated in school, hospital and familial outbreaks (6).

Of the more than 100 types of enteric viruses which may be isolated or observed in raw sewage, vaccine poliovirus strains are the most frequently recovered in demographically normal communities. Dominant types do change, reflecting the strains endemic in a community at any time.

An interesting picture of the range of virus species which may be present in unchlorinated secondary effluent may be obtained from a paper presented by Metcalf et al. (1.g.) on their studies of virus transport in a mariculture system. In addition to all three poliovirus types, 13 other potentially pathogenic enterovirus types were isolated and identified. In addition, a larger number of unidentified virus isolants were recovered. Metcalf et al. state that

> "the non poliovirus isolants were considered indicative of enteric virus pathogens excreted by infected humans, whether clinically ill or not, and representative of the opportunities presented for introduction. . .of more serious virus pathogens like infectious hepatitis virus."

Berg (1.b.), in his 1978 paper reminded us that the source of all these viruses is sewage. The use of poliovirus vaccination with its resultant heavy seeding of sewage, he believes, serves to mask the fluctuations in the levels of potentially pathogenic virus strains. Adenoviruses, for example, are primarily cold weather infectors, but are rarely sought specifically. In addition, hepatitis A virus and the rotaviruses implicated in human gastroenteritis are not readily detected in water.

Akin et al. cited Craum and McCabe's review (7) in support of their belief that direct sewage contamination through plumbing cross-connections or obvious failures in water treatment of sewage contaminated with source waters have been found to be at fault in most water-borne outbreaks in the United States.

WASTEWATER TREATMENT EFFICACY

Biological agents can be removed by available technology as Sproul (1.i.) has written. However, treatment is generally limited to conventional secondary treatment because of the costs involved in going beyond secondary treatment. In his paper, Sproul has given data on the efficacy of pathogen removal by conventional treatment (see Tables 4, 5). Chlorination can reduce the pathogen levels still further. However, many of the data cited for the efficacy of anaerobic digestion and of chlorination were carried out by adding free rather than solid- or particle-associated organisms to the water or sludge. Under these conditions, the organisms are readily accessible to the inactivating agents and processes. In real wastewater and sludge the free condition probably does not prevail and protection by particulate-association or solids-occlusion undoubtedly affects the rate and degree of inactivation.

Sanders and her co-workers (27) utilized solids-incorporated poliovirus in their laboratory study of the digestors and found that incorporation into solids afforded some protection to poliovirus I (see Figure 1). Even with solids incorporation, 99% loss of viral recoverability was obtained in five days. Nevertheless, Lund (24) has isolated poliovirus III from a municipal digestor operated at 50 C and Palfi (26) isolated up to 17 infectious indigenous viruses from 500 ml of sludge leaving a digestor with greater than 60 days retention time at 34 C. Moore and her co-workers (1.h.), in a field study of sludge handling, found a two $\log_{10}$ reduction in indigenous viruses isolated from a two-stage digestor-thickener with a 100-day retention time (see Table 6). The authors commented that this study emphasizes the limitations of laboratory modeling. Actual digestor operation is subject to overloading and shortcircuiting and total elimination of viruses is not achievable by this process. Low levels of enteroviruses probably will be introduced into soil systems by land application of municipal sludges.

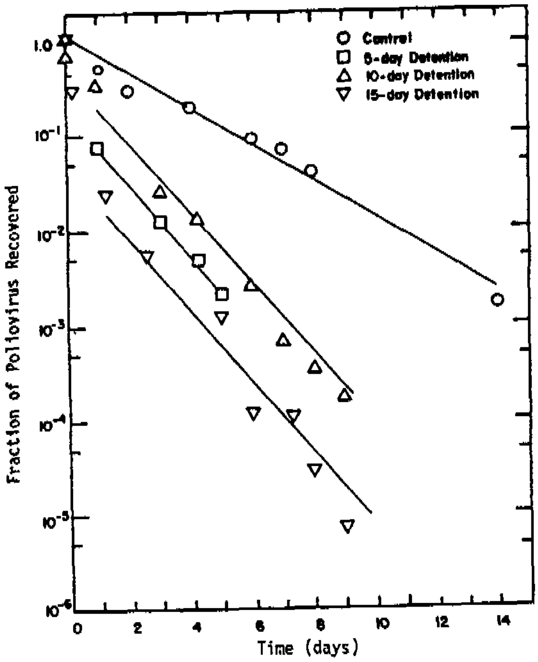


Figure 1. Poliovirus Recovery from Anaerobic Digesters (37 C)

TABLE 6.

Enterovirus Reductions by the Anaerobic Sludge Digested Process, East Pearl Treatment Plant, Boulder, Colorado

	Indigenous Enterovirus Reduction (%)*		
Sample	Digester (40 D)	Thickener (60 D)	Total
1	92	7	99
2	71	28	99
3	87	12	99

* Hela cell monolayers, direct plating of sludge eluates.

Foster and Engelbrecht (13) have estimated the level of pathogens applied to the soil in chlorinated, secondarily treated effluent assuming average efficacy of removal or disinfection by the treatment processes. These data are given in Table 7.

Moore and her co-workers (25) had shown earlier that viruses are sequestered in the secondary sludge in biological secondary treatment, confirming Lund's (24) observation on the virus-binding capacity of sludge. Recent data on the solids-association of viruses, reported by Moore and co-workers in 1978 are given below (Table 8). Unpublished data from Duboise, using genetically identifiable bacteria, suggested that they, too, are concentrated in the biomass. Hays (18) reported that ova and cysts are partitioned in similar fashion. If this is so, it can be seen that the potential for seeding the environment by land disposal of sludges is significantly greater even than that calculated for wastewaters.

TABLE 4.

Removal of Organisms by Conventional Treatment

Agent	Removal (%)	System
	PLAIN SEDIMENTATION	
Parasites		
Beef tapeworm eggs	58	Bench
E. histolytica cysts	0 to incomplete removal	Plant
Bacteria		
Tubercle bacilli	50	Plant
Coliform	27-96	Bench
Viruses		
Polio 1	0	Bench
	Inc to 69% removal	Plant
Polio 1, 2, 3	0-12	Plant
	TRICKLING FILTERATION	
Parasites		
Beef tapeworm eggs	30	Bench
E. histolytica cysts	Incomplete	Plant
	90-99.9	Bench
Bacteria		
Tubercle bacilli	45	Plant
S. typhi	72	Plant
Coliform	98	Plant
Ps. aeruginosa	+74 (increase)	Plant
Cl. perfringens	92	Plant
Viruses		
Coxsackie A9	94	Bench
Echovirus 12	83	Bench
Polio 1	85	Bench
Mixed (natural)	Inc. to 69	Plant
	ACTIVATED SLUDGE	
Parasites		
Beef tapeworm eggs	0	Bench
E. histolytica cysts	Incomplete	Plant
Bacteria		
E. typhosa	86-99	Bench
Cholera	96-100	Bench
Tubercle bacilli	90+	Bench
Coliform	97	Bench
Fecal streptococci	96	Bench
Viruses		
Coxsackie A9	96-99	Bench
Polio 1	79-94	Bench
Mixed (native)	53-71	Plant
Polio 1, 2, 3	76-90	Plant

TABLE 5.

Inactivation by Anaerobic Sludge Digestion

Agent	Inactivation	Temp.	Time(Days)	System
Parasites				
Beef tapeworm eggs	50+	85°C	180	Bench
Bacteria				
Tubercle bacilli	70-85	Not Given	35	Plant & Bench
Salmonella	25	Not Given	60-90	Plant
Viruses				
Polio 1	93	28°C	1	Bench
Polio 2	98	28°C	1	Bench
Polio 1, 2	99.999+	28°C	5	Bench
Coxsackie A9	98	35°C	1	Bench
Echovirus 11	54	35°C	1	Bench
Echovirus 11	98	35°C	2	Bench
Mixed	(15% of sample were positive)	33°C	40	Plant

TABLE 7.

Estimated Wastewater Pathogens Applied to Soil

Pathogen	Raw Wastewater	Number of Organisms per million gallons Primary Effluent	Secondary Effluent	Disinfection[a]	Organisms Applied per acre per day[b]
Salmonella	2×10^{10}	1×10^{10} (50%)[c]	5×10^{8} (95%)	5×10^{5}	3.9×10^{3}
Mycobacterium	2×10^{8}	1×10^{8} (50%)	1.5×10^{7} (85%)	1.5×10^{4}	1.2×10^{2}
E. histolytica	1.5×10^{7}	1.3×10^{7} (50%)	1.2×10^{7} (10%)	1.2×10^{4}	9.3×10^{1}
Helminth ova	2.5×10^{8}	2.5×10^{7} (50%)	5×10^{6} (80%)	5×10^{3}	3.9×10^{1}
Virus	4×10^{10}	2×10^{10} (50%)	2×10^{9} (90%)	2×10^{6}	1.6×10^{4}

[a] Conditions sufficient to yield a 99.9% kill.

[b] Applied at a rate of 2 inches per week.

[c] Estimated pathogen percentage removal efficiency of the treatment.

TABLE 8.

Enterovirus Distribution in Activated Sludge Aeration Basins

Treatment Plant Location	MLSS (mg/l)	Viruses Detected (pfu/l)* Liquid	Solids+	Solids Associated (%)
Austin, Texas	1800	90	950	93
	2100	40	600	95
	1700	70	450	91
Chicago, Illinois		40	202	83
		2	190	99
		5	279	98
Butte, Montana	1560	57	720	93
Portland, Oregon	2780	34	500	94

* Assayed as plaque-forming units (pfu) on Hela cells.

\+ Total pfu eluted from the collected solids of 1 liter mixed liquor sample.

SURVIVAL OF PATHOGENS IN SOIL

Land disposal makes it imperative that we be able to estimate the survival of microbiota in soil. Hays (18) has reviewed the survival of protozoa and helminths in treatment plants and in soil. Biological secondary treatment plus disinfection was reported to be ineffective in destroying parasites. Heavier eggs appear in the sludge as do most cysts. Only heat is effective in inactivating parasites. However, fully embryonated *Ascaris* eggs may be relatively resistant even to heating for 30 minutes at 60 C.

The survival of parasite ova and cysts in soil is influenced by some of the same factors which affect bacterial and viral survival. Of all the parasites mentioned, *Ascaris* ova are the most refractory to treatment and adverse conditions.

In a recent review, Gerba et al. (14) listed the factors affecting the survival of enteric bacteria in soil as moisture content, moisture holding capacity, temperature, pH, sunlight, organic matter, and antagonism from soil microflora. These parameters should be remembered when comparing microbial survival data among various studies. Early studies by Beard (4,5) demonstrated that *Salmonella typhosa* could be recovered from loam and peat soils for periods up to 85 days, while survival of this organism in drying sand was only 4-7 days. Additionally, *S. typhosa* may survive as long as two years at freezing temperatures. *Mycobacteria*, because of their high content of waxy substances, can survive even dry conditions for long periods of time. Greenberg and Kupka (16) in a review of available literature cited several times ranging from 150 days to 15 months for *Mycobacteria* in soil.

In two interesting papers, Bagley and Seidler (3) and Knittel et al. (21) have examined a large number of *Klebsiella pneumoniae* isolates of various origins for their ability to grow at elevated temperatures. They found that this characteristic was stable after prolonged growth. Further, their studies show colonization of the botanical environment by *Klebsiella* isolates of pathogenic origin. The authors suggest that aquatic environments polluted with botanical material may serve as potential reserviors for perpetuating the growth and spread of opportunistic *Klebsiella* pathogens that may ultimately colonize animals, humans and aquatic organisms.

Survival of viruses in soils are influenced by many of the same parameters described above, although at this time little direct evidence supports viral inactivation by antagonistic microorganisms. The effect of temperature and moisture on the survival of poliovirus I (Chat) is shown in Table 9. As expected, lower temperatures favor longer survival times. Observation of a one $\log_{10}$ loss of viral titer required approximately 3 months at 4 C, 1 month at 20 C, and less than one week at 30 C. Likewise, an optimal soil moisture content favors poliovirus survival in soil, while dessication results in a more rapid loss of virus recoverability. Bagdasaryan (2), working with a wide variety of human enteroviruses including polioviruses, Coxsackieviruses, and echoviruses, reported survival times ranging from 110 days to 170 days at a soil pH of 7.5 and a soil temperature of 3-10 C. Moore and her co-workers (l.h.) have presented field data showing recovery of indigenous enteric viruses from fields injected six months earlier with anaerobically digested sludge and then allowed to winter over.

of the core effluents. Changes in the ionic nature of percolate waters would be expected to have the same effect in field situations.

TABLE 9.

Effect of Temperature and Moisture on the Survival of Poliovirus I

Days	% Virus Recovered at Various Temperatures*			% Virus Recovered at Various Moisture Contents			
	4 C	20 C	30 C	25%	15%	Drying	
1	74	99	33	69	99	74	(13.1)
3	68	139	17	42	138	35	(10.9)
8	48	44	2	22	44	0.3	(6.2)
10	68	40	1	17	40	0.08	(5.5)
14	47	53	0.5	13	53	0.02	(4.6)
21	45	24	0.1	10	24	ND	
28	33	12	0.01	5	12	0.003	(4.6)
42	22	9	0.006	2	9	ND	
49	13	5	ND	1	5	0.002	(0.6)
80	12	0.7	ND	0.2	0.7	ND	
100	8	0.4	ND	0.07	0.4	ND	
134	5	0.2	ND	0.004	0.2	ND	

* 15% soil moisture content.

MOVEMENT OF PATHOGENS IN SOIL

Removal of bacteria from liquid percolating through a soil is due to both mechanical removal, i.e. straining or sieving at the soil surface, and adsorption to soil particulates. Studies in Rumania (recently reviewed by Gerba et al., 1975) using coliform bacteria labeled with radioactive phosphorus demonstrated that 92-97% of the bacteria were retained in the first centimeter of soil, while 3-5% were detected at depths between 1-5 cm. The direct relationship of coliform removal from percolating water to increasing cation concentration and decreasing pH are consistent with classical adsorption theory.

Bacterial movement through soils has been demonstrated at several field sites. Reporting from the available literature reports, Gerba et al. (14) noted coliform movements in a variety of soils for distances ranging from 3 feet to 1500 feet. Release and movement of microorganisms would be expected as physical assorption of particulates is a reversible phenomenon and, in part, ion-dependent. Duboise (1977, personal communication) has monitored the movement of a genetically-distinguishable coliform organism through soil cores during cyclic applications of secondary effluent followed by distilled water. The release and subsequent movement of this organism was consistent with decreasing conductivity

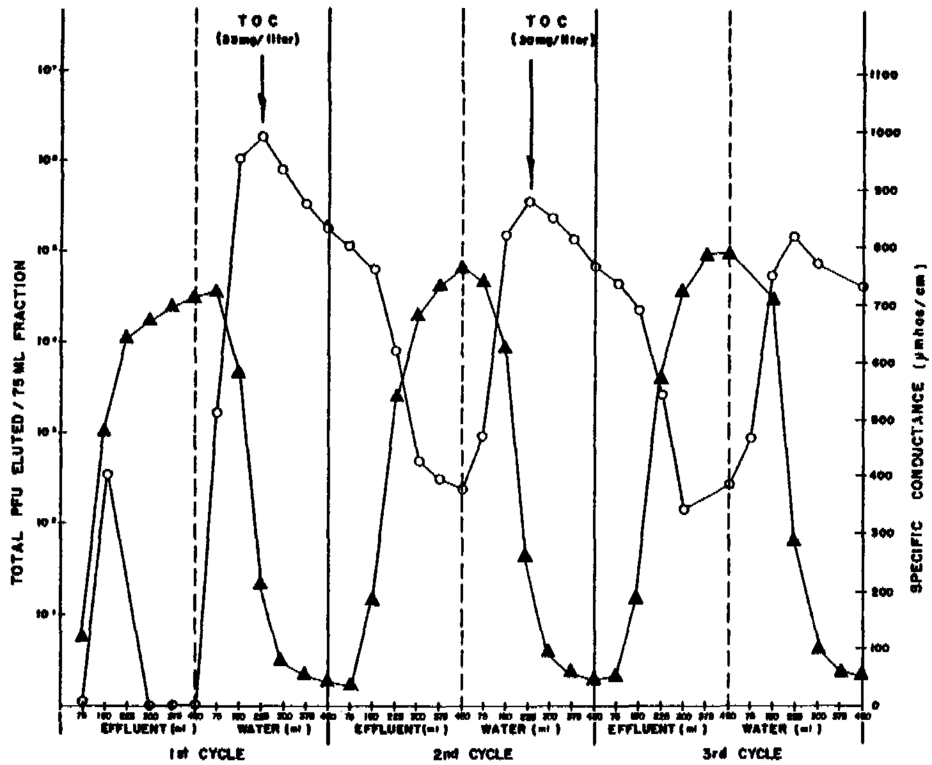
In like manner, the phenomenon of adsorption as a mechanism for the retention of viruses in soil systems was demonstrated by Drewry and Eliassen (10). The results they obtained using bacteriophage systems showed that virus adsorption followed typical Freundlich isotherms. In general, virus adsorption by soils increased with ion exchange capacity, clay content, organic carbon, and glycerol-retention capacity.

The movement of poliovirus I (Chat) through 20 cm length, non-sterile cores taken from a sandy forest soil was monitored using simulated cycles of effluent application and rainfall (12). Results, illustrated in Figure 2, show a burst of released virus detected in the core effluent as the specific conductance of the percolating water began to decrease. This pattern of movement inversely related to specific conductance, was repeated through three cycles with 22.4% of the total virus applied being recovered in the core effluents. Additionally, these authors found the capacity of surviving virions to migrate through the soil columns during an 84 day period (during which time the natural soil moisture was maintained) was unchanged. Similar movement of poliovirus in 250 cm columns packed with calcareous sand were reported by Lance et al. (22). While most of the virus inoculum applied to the column surface was adsorbed in the top 5 cm of soil, subsequent application of deionized water resulted in virus desorption and movement to a depth of 160 cm. In this study, drying for 1 day between viral application and flooding with deionized water reportedly prevented desorption (or enhanced viral inactivation).

two indicators of sewage pollution vary independently. Berg also argues from the widely-discussed Katzenelson, Buium, Shuval paper (20) on the risk of communicable

Figure 2.

Cyclic Elution of Poliovirus from Soil Cores



Both laboratory and field studies have yielded data consistent with prolonged survival and recovery of viruses, bacteria and helminth ova in soils where climatic conditions are appropriate to such survival. The subsequent movement of the applied pathogens through soil is a complex function of soil structure, its physicochemical characteristics and regional precipitation pattern.

Berg (1.b.) has stated that dependence upon the transitional fecal coliform levels as safety indicators in not adequate for judging virus levels. From his paper, Table 10 has been constructed. It can be seen that the levels of these

disease infection associated with wastewater irrigation that

> "[W]hat may be most significant is that in none of the affected kibbutzim had there been an awareness of higher incidence of illnesses therein, some of them quite severe, than in the kibbutzim that did not practice spray irrigation with wastewater."

He concludes that it is not acceptable to demand epidemiological proof of such transmission when we know that viruses are present at water intakes and that plants break down periodically.

TABLE 10.

Recovery of Viruses and Fecal Coliforms From Rivers at or Near Water Intakes

	Missouri River		Mississippi River					New York Bight									
Test No.	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
Viruses (PFU/380 liters)	38	6	5	6	2	2	6	14	11	4	28	5	235	2	2	32	1
Fecal Coliforms (CFU/100 ml)	137	0	--	400	1250	--	140	1300	13	172	330	17	4900	4	49	1300	8

This conclusion contrasts somewhat in tone with those of Llewellyn (l.e.) who said that

> "the state of scientific and of public health knowledge does not permit a statement of no risk to human health from . . . land application. . . . There is still little *evidence* (our evidence) of a major health hazard from land application . . . [t]he lack of data regarding disease associated with wastewater treatment and land application may reflect either the absence of a problem, the lack of intensive monitoring or the inadequacy of [the] present epidemiological methods to detect these problems. . . all three may be true."

Melnick (l.f.), too, questions whether against the background of the natural transmission of enteric viruses it is feasible to obtain data on the risk of *additional* exposure by contaminated water. In this he paraphrases Westwood and Sattar (30) who wrote

> ". . .extreme dilution of virus in the environment will reduce the probability of infection of individuals in an exposed population, but will not eliminate it. Since many viruses are stable for prolonged periods of time under external conditions, the effect of viral pollution of water must be a low-grade seeding of the population with the production of endemic foci of infection in the community.
>
> "It is suggested that the real question to be answered at this time is whether the importance of this seeding relative to other modes of teansmission in the community is sufficient to warrant major action. . ."

though he lacks their faith in the efficacy of standard epidemiological investigative techniques. Melnick argues that statistics on waterborne disease outbreaks are woefully inadequate. Further, he writes, our knowledge of waterborne viral disease has not been advanced much by conventional epidemiological studies. This (citing Goldfield, 1976) is because a substantial portion of the exposed population would likely possess prior immunity and simply suffer, at most, silent transient reinfection. The larger portion of those not previously infected would not develop illness as the inapparent infection disease ratios in the enterovirus infection are quite high (even in hepatitis A; Dienstag et al., 1978). And of the small number developing illness, so varied a clinical picture would be present as to be unrecognizable as related to a single etiologic agent.

In another form these arguments appeared again when Hetrick (19) wrote concerning the burgeoning problem of human viruses in estuarine waters. He cited the large number of infectious conditions which can be produced by viruses and stated that

> ". . .Although it is generally acknowledged that bathing in polluted seawater poses a threat of disease, there are no reports tracing an outbreak of a viral disease to such a water. However, this lack of evidence is probably a false security, as many infections of individuals could be occurring and not be reported or, if reported, are not associated with exposure to polluted water. Indeed, the following facts indicate that the presence of very low levels of enteric and hepatitis viruses in surface waters are of public health importance. First, the minimum infective dose of enteroviruses for human beings is very low; e.g. only 1 to 2 plaque-forming units of poliomyelitis is needed to infect a human being, which is in marked contrast to waterborne bacterial diseases, such as typhoid fever or cholera where large numbers of organisms need to be ingested before infection results. Second, the great majority of enterovirus infections (>90%) are sub-clinical in nature, but individuals with such infections can transmit the virus to their contacts. . . . Third, enteric viruses produce a range of clinical diseases with widely varying incubation periods, which further complicates epidemiological identification of a waterborne source of viral infection."

The data presented above and the references cited all make clear the measurable survival of significant numbers of helminthic and protozoal parasites, bacteria and viruses through all the conventional secondary wastewater treatment processes, including chlorine disinfection. However, such treated effluents have been discharged to surface waters and both the effluents and sludges used in intensive agricultural practice for many years.

Schaerff (28) has reported on nearly 80 years of such experience in Braunschweig in Central Germany. Three thousand hectares now are irrigated by the Braunsschweig Sewage Utilization Association. Irrigation is combined with percolation basins to handle the volume. The sludge is stabilized by six months digestion in ponds and is plowed under by association farmers. Cereals, potatoes, sugar beets and asparagus are the principal crops. The *economically* successful cultivation of sugar beets, barley and wheat on the area's light sandy soil is possible because of the use of wastewater irrigation. Schaerff wrote that the long digestion period in the ponds effectively destroys known pathogens which settled with the solids. No evidence of ill health has been seen in persons working with the wastewater irrigation system. No contamination of groundwater has been evidenced by the wells in the irrigation and percolation basins area. Finally, the water quality of the Oker River which drains the Braunschweig area has shown improvement downstream of Braunschweig.

In Munich, Süß and his co-workers at the Bavarian Agricultural and Botanical Institute have studied the effects on agricultural yields of sludge utilization, an area practice of long-standing. At present, they are irradiating the sludge from a plant serving about one-fourth of the city's population. They report that 300 kilorads from a Co^{60} source improves dewaterability and reduces the extremely high level of *Salmonella* found in the region's domestic sludges. Again, no adverse health effects of the selective agricultural use of properly treated sludge is seen (29).

Currently, in California, there are over 200 planned reclamation projects using reclaimed water for crop and landscape irrigation, ornamental and recreational impoundments, groundwater recharge, industrial purposes, and fire protection. Planned reclamation returns about 7% of the wastewater produced (8). About 90% of the wastewater reclaimed is used for crop irrigation, much of it for fodder, fiber and seed crops. The criteria applied to water for such uses are less stringent than those required for water to be used for food crop irrigation. The most restrictive standards are applied for use in irrigation of urban parks, playgrounds and athletic fields.

SUMMARY AND CONCLUSIONS

This paper has shown that current treatment chain design appropriately utilized is probably adequate to remove much of the potential public health hazard due to biological cause from wastewater effluents. Available data suggest that rather than inactivation, what occurs during treatment is sequestering of the microbiota into the secondary biomass. The problem then becomes one of sludge handling.

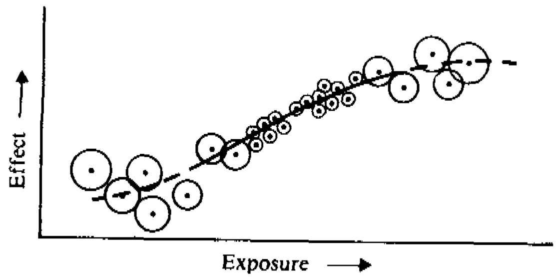
Anaerobic digestion is reasonably effective in inactivating most pathogens, the precise degree of efficacy reported from the laboratory depends upon the indicator organism used and the conditions of the experiment. In actual practice, short-circuiting and mechanical defects in mixing make such anaerobic digesters less efficient in the field than they are in the laboratory.

The restraint on land disposal which is operative despite all that which has been cited is the implicit desire for absolute safety, for risk-free land application. To return to Lowrence (23) briefly:

> *"A thing is safe if its risks are judged to be acceptable.*
>
> "By its preciseness and connotative power this definition contrasts sharply with simplistic dictionary definitions that have 'safe' meaning something like 'free from risk.' Nothing can be absolutely free of risk. One can't think of anything that isn't, under some circumstances, able to cause harm. There are degrees of risk, and consequently there are degrees of safety.
>
> "Notice that this definition emphasizes the relativity and judgmental nature of the concept of safety. It also implies that two very different activities are required for determining how safe things are: *measuring risk*, an objective but probabilistic pursuit; and *judging the acceptability of that risk (judging safety)*, a matter of personal and social value judgment."

We must recognize then, that in laboratory and field studies we have been measuring risk. Using appropriate treatment methodology brings us to the area of uncertainty shown in Figure 3 for low exposure effects. Judging the acceptability of that risk (judging safety) is a matter for social value judgment, an issue quite appropriate to the political arena. It is also an issue which will be decided differently in different countries, depending on direct economic benefits, on the level of technological development and---this is of enormous importance---on the present level of pathogens extant in the local environment and indigenous to that country's population.

Figure 3.
Generalized exposure-effect curve showing uncertainly at high and low exposure: diameter of circles indicates degree of certainty about data points.



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COMPARISON OF HEALTH CONSIDERATIONS FOR LAND TREATMENT OF WASTEWATER

TOXIC CHEMICALS ASSOCIATED WITH LAND TREATMENT OF WASTEWATER[1/]

A. C. Chang and A. L. Page[2/]

ABSTRACT

Land application of domestic wastewater has been practiced around the world since the advent of the wastewater treatment technology. In recent years, because of the need to conserve water and restrictions imposed on the waste disposal in surface waters, the interest of land oriented wastewater treatment systems has again attracted widespread attention in the U.S. The more recently conceived concepts of land treatment are not intended to replace the conventional wastewater purification processes, but to further the treatment of wastewater by removing additional impurities in a natural or artificially constructed soil mantle. Through the tertiary treatment, the discharged wastewater could be effectively utilized for recharging natural water systems (surface or underground) or irrigation, two widely accepted uses of reclaimed water. Numerous investigations have been conducted to determine the fate of disease causing microorganisms, nitrogen, and phosphorus in the wastewater during the course of a land treatment. As to the existence of potentially toxic chemicals in the wastewater and their possible pathways in soils, little is known at the present time. Depending on the nature of the land treatment system, hazardous substances in wastewater may pose a present or potentially future long-term danger to human health or other living organisms because they are toxic, non-degradable, or persistent in the natural environment. Some substances may also be biologically magnified or otherwise cause elemental cumulative effects. This presentation systematically reviews the fate of potentially hazardous substances during soil treatment of wastewater effluents.

The physical, chemical and microbiological mechanisms of wastewater treatment systems are capable of removing the majority of the toxic agents. In the treated wastewater effluents, the concentration of hazardous substances are usually low. Based upon existing data and water quality criteria, the hazardous nature of wastewater effluents is characterized and evaluated. Physical and chemical mechanisms of removing hazardous chemicals from wastewater during land treatment are reviewed. It appears that trace metal elements may accumulate to become a potential hazard during land treatment,

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especially for wastewater originating from industrial waste discharge. The concentrations of organic substances in wastewater are very low making the task of identifying their origin difficult. Further studies are necessary to determine the role of stable organic substances in land treatment systems.

INTRODUCTION

Land application of domestic wastewater has been practiced around the world since the advent of wastewater treatment technology. Most of these practices were abandoned for rapid urbanization, advancement in wastewater treatment technology and increasing concerns over the potential public health hazards. In recent years, the interests of land oriented wastewater treatment concepts again attracted widespread attention in the U.S. Part of the reason, at least, lies on the nation's commitment in achieving the ultimate goal of water pollution control through physically, chemically and biologically balanced natural water systems. The single goal ecological oriented approach to clean water resulted in a management strategy that would ban all waterborne waste disposal practices wherever possible through the use of closed-cycled water reuse technology and land disposal systems.[1] The more recently conceived concepts of wastewater land treatment are not intended to circumvent the conventional wastewater purification processes, but to further the degree of wastewater treatment by removing additional impurities in a natural or artificially constructed soil mantle.[2,3,4]

Numerous investigations have been conducted to determine the fate of disease causing microorganisms, nitrogen and phosphorus compounds, and biodegradable organic matter in the wastewater during the course of a land treatment.[5,6,7] As to the existence of potentially hazardous chemicals in the wastewater and their possible pathway in land treatment, hazardous substances may pose a present or potentially future long-term danger to human health and other living organisms because they are toxic, and possibly non-degradable or persistant in the natural environment. Some substances may also be biologically magnified or otherwise present elemental cumulative effects. Understanding wastewater borne toxic substances' behavior in soils is essential to the effectiveness of wastewater land treatment.

DOMESTIC WASTEWATER AND ITS TREATMENT

Any beneficial use of water always results in quality degradations. If the increments of contaminants in the water interfere with the subsequent use of this water or degrade the quality of the receiving water body, the wastewater requires treatment. Wastewater purification processes are essential in preventing the continuing degradation of water quality in a multiple-purpose, multiple-use water resource management system. Wastewaters of urban environment contain five general categories of impurities that seriously affect its quality. They are:

(1) biochemical-chemical oxygen demanding organic material,
(2) disease causing microorganisms,
(3) essential plant nutrients,
(4) toxic chemical substances, and
(5) dissolved minerals.

A conventional wastewater treatment system consists of the physical separation of suspended solids at the primary treatment stage, followed by bio-adsorption and flocculation of dissolved biodegradable organic substances and a final disinfection process before discharge. This type of treatment process is effective in removing the turbidity causing suspended solids, oxygen demanding organic substances and bacteria of fecal origin from the wastewater (Table 1). The removal of nitrogenous and phosphorus compounds in a typical wastewater treatment plant, by and large, is accidental and not by design. As to the increment of chemical constituents that contribute to the total dissolved minerals of the water no significant change of its concentration in the effluent water is expected. Since the water quality changes during a treatment were not evaluated by specific compounds, little information could be extrapolated to its effectiveness of removing potentially hazardous compounds. The efficient removal of suspended solids and BOD (biochemical oxygen demand) in a treatment in no way indicates the corresponding reduction of toxic chemicals.

Organic Chemicals of Wastewater

The presence of organic matter in raw and treated wastewater is highly variable and dependent upon the source of wastewater and the degree of treatment. The organic matter content of wastewaters is expressed in terms of measured parameters such as BOD (biochemical oxygen demand), COD (chemical oxygen demand) or TOC (total organic carbon). These parameters bear little indication to the chemical nature and toxicity of wastewater organics. Attempts to characterize the organic matter in wastewaters in the past have not met with complete success.[8,9,10] Compounds to be identified are numerous and complete isolation and identification of each compound has not yet been achieved. In the raw wastewater entering a treatment plant, organic compounds consisted primarily of carbohydrates, fatty acids and esters, proteins and amino acids and anionic surface-active agents (detergents). After the secondary treatment, organic matter content of wastewater is greatly reduced. The more refractory and high molecular weight organics such as fulvic acid, humic acid, and hymathomelanic acid become the major organic constituents of treated wastewater. Many of these substances are naturally occuring and there is no indication that they would be toxic to the subsequent users of the water. Much of these high molecular weight organic substances generally constituted the "color" and "taste" associated with the treated water. The nature of the unaccounted fraction (usually 20-50% of the total organic matter) remains to be characterized.

With the assistance of more sophisticated analytical instruments and advanced techniques, efforts have been made to identify stable organic compounds that entered the wastewater collection system and survived the treatment processes. Because of their extremely low concentrations in water (generally less than 100 μg/l), the procedures of isolation and identification are tedious and time consuming and the list of identified compounds is long.[11,12] Chlorination of treated wastewater effluents brought about many chlorinated organic compounds (chloro-benzoic acids, chlorophenol, etc.) that are more persistent in natural systems.[13] Although many of these organic compounds are naturally occuring and they present more of a problem in terms of taste, odor, and color rather than toxicity.[14,15] There are also compounds considered potentially hazardous, because they are persistent, in the environment. More importantly, their chemical and biochemical reactions in soils are not known. The low concentration in water would undoubtedly affect the chemical reaction kinetics.

Besides treated wastewater effluents, organic compounds are also found in natural waters and public drinking waters. Very little is known concerning the public health impact of trace amounts of biologically resistant and potentially hazardous organic chemicals in drinking water. There is little evidence to link the presence of these substances in water to any human disease. The hazard of exposing human subjects to long-term low level contact and the probability of biological accumulation in other organisms should not be overlooked.[16] For this reason, the concentration of organics in the wastewater should be reduced to as low as possible in any wastewater treatment. The national Academy of Science, Safe Drinking Water Committee has identified more than 100 organic compounds whose presence in water are considered undesirable because they have been found in detectable concentration in water, and there are data to suggest toxicity to man and test animals.[17]

Inorganic Chemicals of Wastewater

Unlike the vaguely-defined and still little understood toxic trace organic constituents in wastewaters, the toxicological significance and hazardous nature of inorganic constituents of wastewater are well documented.[18] Generally, hazardous inorganic chemical constituents of wastewater are the trace elements. Geochemically, trace elements are a group of otherwise unrelated chemical elements that are present in natural system in extremely low concentrations. Some of these elements are essential for the growth of living organisms. When present in high enough concentrations, essential trace elements may become toxic. The list of potentially hazardous trace elements is long and probably increasing. Boron, cadmium, chromium, copper, nickel, mercury, arsenic, molybdenum, vanadium, cobalt, silver, lead, fluorine, aluminum, iron, manganese, titanium and zinc all have to be considered potentially dangerous to humans or to plants at elevated concentrations.

Many of these elements are widely used in industrial processing and in manufacturing of consumer goods. For this reason, small amounts of trace elements are always present in wastewater. Excessively high concentrations of any trace element in wastewater always indicates industrial waste discharge into the collection system. During a wastewater treatment, trace elements could be removed from wastewater, but the removal efficiencies are not always consistent (Table 2). If the concentration in the wastewater is high or the removal in a wastewater treatment plant is inadequate, additional removal becomes necessary.

Table 3 summarizes the concentration range of selected trace elements in the treated wastewater effluents.[18] These concentrations are also compared with the established water quality criteria for public water supply and for crop irrigation use. Properly treated, most wastewater could meet the recommended trace element limitations for both uses. Among elements likely to exceed the upper limits, lead and cadmium are closely related to industrial wastes. The main source of boron is the widespread use of boron containing household detergents. Except for boron, the use of wastewater on fine textured soils with neutral or alkaline pH minimizes toxic effects of trace elements to plants.

REMOVAL OF TOXIC CHEMICALS BY LAND TREATMENT

A conventional wastewater treatment system is designed to remove gross contaminants of wastewater (i.e., suspended solids, biochemical oxygen demanding organic material, bacteria of fecal origin, etc.). To safeguard the biological treatment processes, toxic chemicals in high concentrations are also prevented from entering the treatment system. Certain degrees of toxic chemical removal during wastewater treatment is also expected. However, the removal of any toxic chemical substances is not integrated into the operation of a domestic wastewater treatment process. For this reason, the removal efficiency of these substances in wastewater treatment is not consistent. Like the first and second stage wastewater treatment, the performance evaluation of land treatment of wastewater has also been focused on the effectiveness of soil in removing gross contaminants of water. With the large number of potentially hazardous chemical substances that may be present in the treated wastewater, their impact on a land treatment system is difficult to assess.

Mechanisms in soils that may interact to affect the fate of toxic chemicals in the water to be treated are summarized in Table 4. The effectiveness of each outlined pathway in removing toxic substances from passage water would depend on the properties of the soil and the hydraulic characteristics of the land treatment system. Based on methods of water application, land treatment of wastewater may be categorized into three basic systems:
(1) Irrigation,
(2) infiltration-percolation, and
(3) overland flow.

Irrigation is the most commonly used land treatment system, involving the application of treated wastewater on land for further treatment and for meeting the water demand of growing plants. The applied water undergoes treatment by means of physical, chemical and biological pathways in soils. It not only avoids the direct surface discharge of wastewater, but also provides for water conservation.
In infiltration-percolation system, wastewaters are applied at a relative high rate in confined basins. Because of the expected high hydraulic loading in these systems, coarser textured soils are always used. Under such operating conditions, removal of contaminants from water is predominantly physical straining and filtration. The biological barrier formed near the soil surface may also be effective in chemical adsorption

and biological decomposition. The overland flow system resembles a biological treatment process in which the treated wastewater is applied on the top end of a sloped field and allowed to flow overland through a vegetated surface to a collection ditch at the lower end. As the water flows, physical, chemical and biological reactions take place at the water-soil interface. At the present, existing information does not permit any thorough review on the fate of toxic chemicals under each of the land treatment options. A general overview, however, is presented in the following sections.

Fate of organic chemicals in soils

There is ample evidence to indicate that a relatively large number of soluble organic substances may survive the conventional wastewater treatment process and are present in the treated effluent at concentrations in the $\mu g/l$ range. These compounds involve both toxic and non-toxic organic substances. Because of their extremely low concentration, the source of these substances are difficult to trace and they are less susceptible to microbial decomposition. They enter into waters through natural processes, through waste disposals or result from microbial synthesis or degradation.

In wastewater effluents, most of the stable organic substances appear to be present in polymeric forms and to some extent resemble the soil humus material. If we view soil mantle as a chromatographic column in which adsorption and desorption are rather active, the trace amount of stable organic substances in the percolating water may be adsorbed onto the soil surface.[19] The molecular weights as well as the chemical structure would determine eventually the extent of adsorption under a specific hydraulic loading. The adsorption phenomena of soils could bring the organic substance concentration at the soil surface to a higher concentration and therefore make them more susceptible to microbial decomposition. Simple organic compounds such as aliphatic and phenolic acids, amino acids, and sugars are readily decomposible by soil microorganisms. Substances with branched chains and higher molecular weight are more resistant to biodegradation and require longer periods of time to decompose in soils. Their immobilization by soil adsorption in the biologically active zone is essential to their inactivation.

Pesticide residues constituted a significant fraction of the presently identified potentially toxic organic substances of the water and wastewater. The behavior and decomposition pathway of many pesticides in soils have been studied for many years.[20] In terms of adsorption and mobility of these organic substances, both soil and pesticide characteristics influence the outcome. The affinity of pesticides for soil minerals and organic matter generally decrease in the series of organic matter > vermiculite > montmorillonite > illite > chlorite > kaolinite. Adsorption of organic pesticides by soil also characteristically increases as the concentration of functional groups such as primary, secondary and tertiary amines, amides, carboxyl, and phenolic increase. The presence of organic substances usually increase the adsorption affinity. The pesticides can also be effectively bonded with iron and aluminum oxides at proper soil pH ranges. Both laboratory and field experiments indicate that most pesticide residues remain in surface soils with only a few migrating to depths of 30-60 cm in soil.[21] Because of the rather strong adsorption by soil particles, there is little evidence that pesticide chemicals would move with percolating water to any significant depth during land treatment. Observations made at a few long term ground water recharge operation sites also seemed to support this reasoning; pesticide residues found in the recharged water were well within the concentration normally found in natural waters.[22] Organic substances other than pesticide residues in the wastewater also appear to be adsorbed onto the soil.[23] Although the ability of soil to absorb these compounds is less than that of activated carbon, soils serve as an effective filter to remove them from leaching water. The immobilized organic substance in soils are subsequently subject to chemical or photochemical decomposition, volatilization and microbial decomposition.

Fate of trace elements in soils

Studies of trace elements in treated wastewater effluents indicate that they are present in the wastewater effluent principally as finely divided suspended solids.[24] These fine suspended solids do not settle out easily in the secondary settling process. Through the physical straining at the surface and filtration through the porous soil media, land treatment should effectively remove this fraction of the trace metal elements from the applied water. The removal of soluble trace metal elements of the water, however, would depend largely upon the type and the extent of chemical reaction that may take place in the soil. The amount and the type of clay minerals and soil organic matter are important factors in determining the nature of the chemical reactions. Water percolating through low organic matter, coarse textured soils may exhibit little change in trace element concentration of the applied water. On the other hand, soils with a high percentage of clay and organic matter facilitate quite effective removal of trace elements from the water.

In general, elements which occur in solution as anions or neutral molecules may pass through soils more readily than elements that are present as cations. Elements such as As, Se, F, B and Mo in soils often occur as anions or neutral molecules. Metallic elements (Cd, Cu, Cr, Pb, Hg, Ag and Zn), on the other hand, are most commonly found in water and in soils as cations. All trace elements concerned may also react to form organo-metallic complexes in water and in soil. The chemistry or organo-metallic complex in the heterogeneous natural system is extremely complicated. Currently information is insufficient to assess the impact of organo-metallic complexes in land treatment system. Besides the apparent soil mineral and ion species effect, there are also many other factors that would influence the chemical reactions of the soil. Each of these elements should be treated individually. The probable chemical reaction of a few elements in soils are briefly outlined below:

As. In solution As is commonly present in the trivalent state ($HAsO_2$ and AsO_2^-) or in pentavalent state (H_3AsO_4, $H_2AsO_4^-$, $HAsO_4^=$ and $AsO_4^{\equiv}$). In the pH range of most soils and waters, arsenic in solution is most likely to be present as a mono-valent or divalent anion. Although the chemical reactions of As in soils are not completely understood, it is thought to react with iron, aluminum and/or calcium to form rather insoluble compounds. It appears that in acid soils, the solubility is controlled by the solubility of iron and aluminum arsenites and arsenates. In neutral and calcareous soils, the solubility of calcium arsenate is thought to control concentrations of dissolved arsenic. The solubility of As in soils usually increases as the pH of the soil increases.

Se. The chemistry of Se resembles that of the sulfur. Inorganic selenium occurs in solution as either $SeO_4^=$ or $SeO_3^=$. In acid soils, these anions most likely react with iron and aluminum oxides to form the rather insoluble selenites ($SeO_3^=$). Selenium also becomes more soluble in neutral and calcareous soils.

Cd and Zn. Chemical reactions of Cd and Zn are commonly similar in natural systems. They occur in solution as divalent cations and sometimes as ion pairs. The ion pairs may be cationic, anionic or neutral and have a general formula of $[M_aX_b]^{2a-yb}$ where M represents the divalent metal ion, X represents the anion, y its valence, and a and b the numbers of cations and anions in the complex, respectively. Ion pairs of Zn and Cd may include anions such as OH^-, Cl^-, $CO_3^=$, and $SO_4^=$. Although ion pairs are known to occur under some conditions, their signification to a land treatment are difficult to evaluate. More commonly, Zn and Cd in water are sufficiently attenuated in soils by precipitation, coprecipitation, and adsorption involving clay minerals, organic matter amorphous oxides. As long as soil pH is greater than 4.5, Zn of water is not likely to exceed recommended limits of water quality criteria.

Cu. Like Cd and Zn, copper can exist as ion pairs or free divalent cations in solutions. Under reducing conditions, divalent cupric ion is reduced to the monovalent cuprious ion. In slightly acid, neutral, or calcareous soils, the solubility of inorganic Cu seldom exceeds 1 mg/l due to the formation of quite insoluble organic and inorganic complexes. In most soils, the solubility of Cu is less than that predicted from solubility products of carbonates, indicating other mechanisms (adsorption or other insoluble compounds) may be controlling the solubility of Cu in soils.

Cr. Chromium may occur in solution as Cr^{++}, Cr^{+++}, CrO_3^{---}, $CrO_4^{=}$ and $Cr_2O_7^{=}$. The Cr^{++} ion is unstable and in natural systems rapidly oxidized into Cr^{+++}. Hexavalent chromium ion in soil is more mobile and toxic to plants than trivalent chromium. Recent studies have demonstrated the rapid oxidation of hexavalent chromium to trivalent chromium in the presence of organic matter.[25,26] In turn, the trivalent chromium would form insoluble oxides, leaving little chromium available for plants or leaching.

Pb. Lead may occur in solution as divalent or tetravalent cations. The solubility of Pb in natural system is extremely low due to the formation of sparsely soluble or insoluble $PbSO_4$, $PbCO_3$, $Pb(OH)_2$ and lead phosphate. Lead present in wastewater is probably largely in the form of fine suspended solids and should be effectively removed from wastewater in any form of land treatment.

Hg. The concentration of Hg in soil solution is principally determined by the ionic adsorption on soil minerals, and organic matter. In soils, inorganic and organic compounds may be reduced to elemental Hg and subject to volatilization.

From the brief review of the chemistry of selected toxic trace elements in soils, it is obvious that most soils possess the capacity to immobilize trace elements through chemical adsorption, complexation and precipitation reactions. Since wastewater effluent to be applied on land is usually low in trace elements, for most beneficial use of the water the added removal by soils would further safeguard the use of these waters. Except for boron, laboratory[27] and field experiments[28] both indicate effective trace metal removal by soils or soil-plant systems. Since the trace elements in wastewater effluents are chemically immobilized in the soil, it is essential that the chemical properties of soils that facilitate the immobilization of trace elements be maintained. A change of the chemical properties of soil could release large amounts of potentially hazardous trace metal elements.

CONCLUSIONS

Land treatment of wastewater has been extensively studied in recent years. To wastewater treatment authorities, land treatment offers an alternative to the direct surface discharge of treated wastewater. It may also facilitate conservation of water resources. Among all constituents of concern in a land treatment system, toxic chemicals (i.e., stable organic substances and trace metal elements) are the least understood. Little attention has been focused on the impact of these potentially hazardous substances on a disposal system or on reuse of the water. Existing information tends to indicate that toxic chemicals in the treated wastewater are in low enough concentrations not to interfere with land treatment operation. The concentration of several known toxic substances in treated wastewater were also within the limit recommended for most beneficial use of water. On the other hand, soils also seem to possess the ability to attenuate the input of toxic chemicals through various mechanisms. With existing data it is difficult to project the long term effectiveness and the consistency of soil mantle to immobilize and detoxify hazardous substances in wastewater. Therefore, for long term, full scale land treatment operations, it is essential that the fate of toxic chemicals during the land treatment not be overlooked.

Table 1. Typical treatment of municipal wastewater*

Parameter	Concentration Influent Wastewater	Secondary Effluent	Removal Efficiency
	-------- mg/l	---------	---- % ---
Biochemical Oxygen Demand	200	20	90
Chemical Oxygen Demand	400	80	80
Suspended Solids	200	20	90
Nitrogen	20	18	10
Phosphorus	10	9	10

*Environmental Pollution Control Alternatives: Municipal Wastewater, U. S. EPA Technology Transfer, EPA-625/5-76-012.

Table 2. Trace Metal Removal in Wastewater Treatment*

Element	Removal by Secondary Treatment (%)
Cd	20-45
Cr	40-80
Cu	0-70
Hg	20-75
Ni	15-40
Pb	50-90
Zn	35-80

*Environmental Pollution Control Alternatives: Municipal Wastewater, U. S. EPA Technology Transfer, EPA 625/5-76-012.

Table 3. Concentration of Trace Elements in Treated Wastewater vs. Water Quality Criteria

Element	Wastewater Effluent		Water Quality Criteria		
				Irrigation	
	Range	Median	Public Water Supply	Continuous Use	Short term Use
	mg/l				
As	<0.005-0.023	<0.005	0.1	0.1	2.0
B	0.3 -2.5	0.7	---	0.75	2.0
Cd	<0.005-0.22	<0.005	0.01	0.01	0.05
Cr	<0.001-0.1	0.001	0.05	0.1	1.0
Cu	0.006-0.053	0.018	1.0	0.2	5.0
Hg	<0.0002-0.001	0.0002	0.002	---	---
Mo	0.001-0.018	0.007	---	0.01	0.15
Ni	0.003-0.60	0.004	---	0.20	2.0
Pb	0.003-0.35	0.008	0.05	5.0	10.0
Se	---	---	---	0.02	0.02
Zn	0.004-0.35	0.04	0.05	2.0	10.0

Table 4. Soil Reactions in Removing Toxic Chemicals

Mode of Reaction	Reaction Pathway	Active Component in Soil	Affected Constituents
Chemical	Cation exchange	Clay minerals, organic matter	Toxic metal elements
	Surface adsorption	Clay minerals	Organic substances, toxic metals
	Precipitation	Oxides of Fe, Al, carbonates hydroxides	Toxic metals
	Complexation	Organic matter	Toxic metals
Physical	Surface straining	Surface soil	Suspended solids
	Absorption	Soil particles	Suspended solids
	Sedimentation	Soil pore space	Suspended solids
	Volatilization	Soil surface, pore space	Volatile organic and inorganic constituents, microbial products
Microbial	Decomposition	Soil microorganism	Organic compounds
	Methylation	Soil microorganism	Selected trace metal elements
	Biomagnification	Soil microorganism	Selected trace metal elements pesticides residues

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COMPARISON OF HEALTH CONSIDERATIONS FOR LAND TREATMENT OF WASTEWATER

USE OF WASTEWATER ON LAND - FOOD CHAIN CONCERNS

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ABSTRACT

Disposal of wastewater and effluents on land provides many potential benefits, but carries with it the risk of contaminating human and animal food.

Among the chemical contaminants present in wastewater, heavy metals, including cadmium and lead, and persistent organic compounds are of most concern. Because of dilution and distribution to larger land areas, wastewater must be considered less hazardous than sludge when used at typical application rates, but there are exceptions. Microbiological concerns apply to both wastewater and sludge and the degree of concern depends upon the pretreatment, use practices, and the type of crop receiving the wastewater and/or sludge.

For these reasons, we recommend that consideration be given to the possible adverse effects of the application of wastewater to food crops before this practice is instigated.

INTRODUCTION

Many variables determine the benefits and risks of the practice of applying wastewater to land. The origin of the wastewater is of primary importance. Municipal wastewater ordinarily contains liquid domestic sewage, but may also frequently contain varying amounts of industrial effluents and street runoff. Domestic sewage is a known source of pathogens for humans. Industrial wastes contain a variety of metals, particularly cadmium, lead, and mercury, and refractory organic compounds including polychlorinated biphenyls (PCBs). Street runoff is a source of lead from automotive exhausts and also of some other metals such as zinc and cadmium.

Before its application to land, wastewater may or may not have undergone treatment, which may range from only minimal sedimentation or primary treatment, to the more customary biological processes such as the activated sludge, trickling filter, or aerated lagoon systems that are used in the United States. In general, the more advanced the treatment, the greater the reduction in the levels of many of these chemical and microbiological contaminants in the effluents which are to be applied to land and/or crops. Many of the contaminants, will, however, remain in sewage sludges which, when applied to land, may present a more severe problem than the use of wastewater.

Sewage and effluents usually contain substantial concentrations of soluble nitrogen compounds and phosphates. To prevent ground and surface water contamination, use of vegetation on the land is desirable or necessary. In many instances, grasses are used which sometimes provide food for grazing animals. A more economically attractive solution is the planting of food crops to make use of the fertilizer, water, and organic humus contents; however, planting crops, in turn, causes concern because of the problems of chemical and microbiological contamination of the food supply.

NATURE OF CONTAMINANTS

Metals

The diet in the United States and other industrial countries is high in some toxic metals. The elements of primary concern are cadmium and lead. Table I gives the maximum tolerable daily intakes for these metals proposed by the World Health Organization (WHO) (1), compared with the daily intake of a teen-age male, as estimated by the Food and Drug Administration (FDA) (2). These estimates are based on the results obtained in FDA surveys of the lead and cadmium content of about 50 food commodities and on data obtained in FDA's Total Diet, or "Market Basket" study (3). Depending on the study used and its interpretation, estimates of cadmium intakes range from 30 to 70 µg/day, and are thus at or near the tolerable daily intake proposed by WHO. For this reason, FDA has stated that new practices which would significantly increase the cadmium levels in foods should not be instituted. In the case of lead, we are especially concerned about infants, since they eat more food per pound of body weight and absorb a higher proportion of the lead they ingest than do adults.

Numerous studies have been conducted in recent years on both effluents and sludges containing metals (4,5,6,7,8,9,10). In most instances, the use of sewage sludges is of more concern than the use of wastewater unless the wastewater is heavily contaminated and application rates are high. Figure 1 demonstrates this fact graphically for cadmium. Treated wastewaters usually contain cadmium at less than 20 or at the most 30 parts per billion (ppb) (8), while sludges contain this element in the parts per million (ppm) range, often exceeding 50-100 ppm or more. Although much more wastewater than sludge is applied to a given land area, this difference is not large enough to make up for the higher concentrations of cadmium in sludges as compared with those in wastewater. The end result is that cadmium and other metals are usually applied at higher rates to a given land area when sludge is used rather than wastewater.

Limits allowed for cadmium in drinking water are 10 ppb, based on the drinking water regulations of the Environmental Protection Agency (EPA). In 1972, the National Academy of Sciences proposed an irrigation water standard of only 5 ppb of cadmium because of the accumulation potential of this element. As shown in Fig. 1, use of wastewater with these concentrations on land would not result in cadmium additions of over 0.5 kg/ha/year at realistic application rates of up to 400 cm/year (about 3 inches per week). This rate of 0.5 kg of cadmium applied annually would be the maximum permitted after 1986 in the regulations on the use of wastes on land proposed by EPA (11) and supported by FDA. However, higher cadmium levels in wastewaters of industrial origin would, in the long run, present a much greater health problem. Sewage sludges, even those having moderate cadmium levels, would have to be used at low application rates on land to avoid crop contamination problems. FDA recommends that sludge containing more than 25 ppm should not be used on land on which certain food crops are grown and that maximum annual and cumulative limits be established. The same applies to highly contaminated wastewater, taking the difference in application rates into account.

On secondary treatment, a varying proportion of the amounts of cadmium present in raw wastewater is removed with the sewage sludge; the balance remains in the effluent. The ratios are dependent on the level of cadmium initially present and many other factors: generally, the higher the initial level, the greater the percentage that will remain in sludge (12). Most of the cadmium from low-cadmium wastewater will pass into the effluent in soluble form.

Organic Contaminants

The presence of organic contaminants is a matter of considerable concern, partly because of the lack of sufficient research and study and partly because many of these contaminants are toxic and potentially carcinogenic, teratogenic, or mutagenic to humans.

Wastewater contains a wide variety of organic substances. Some of the natural compounds may or may not be of concern, but the "man-made" variety is of particular interest. This category includes many of the thousands of industrial chemicals; some examples are insecticides, herbicides, and the

TABLE I

Tolerances and Dietary Intakes of Cadmium and Lead

	CADMIUM	LEAD
WHO/FAO (1972), Provisional Tolerances (Adult), Converted to ug/Person/Day	57-71	429
Dietary Intake, U.S. Teen-Age Male, ug/Person/Day	30-70	254

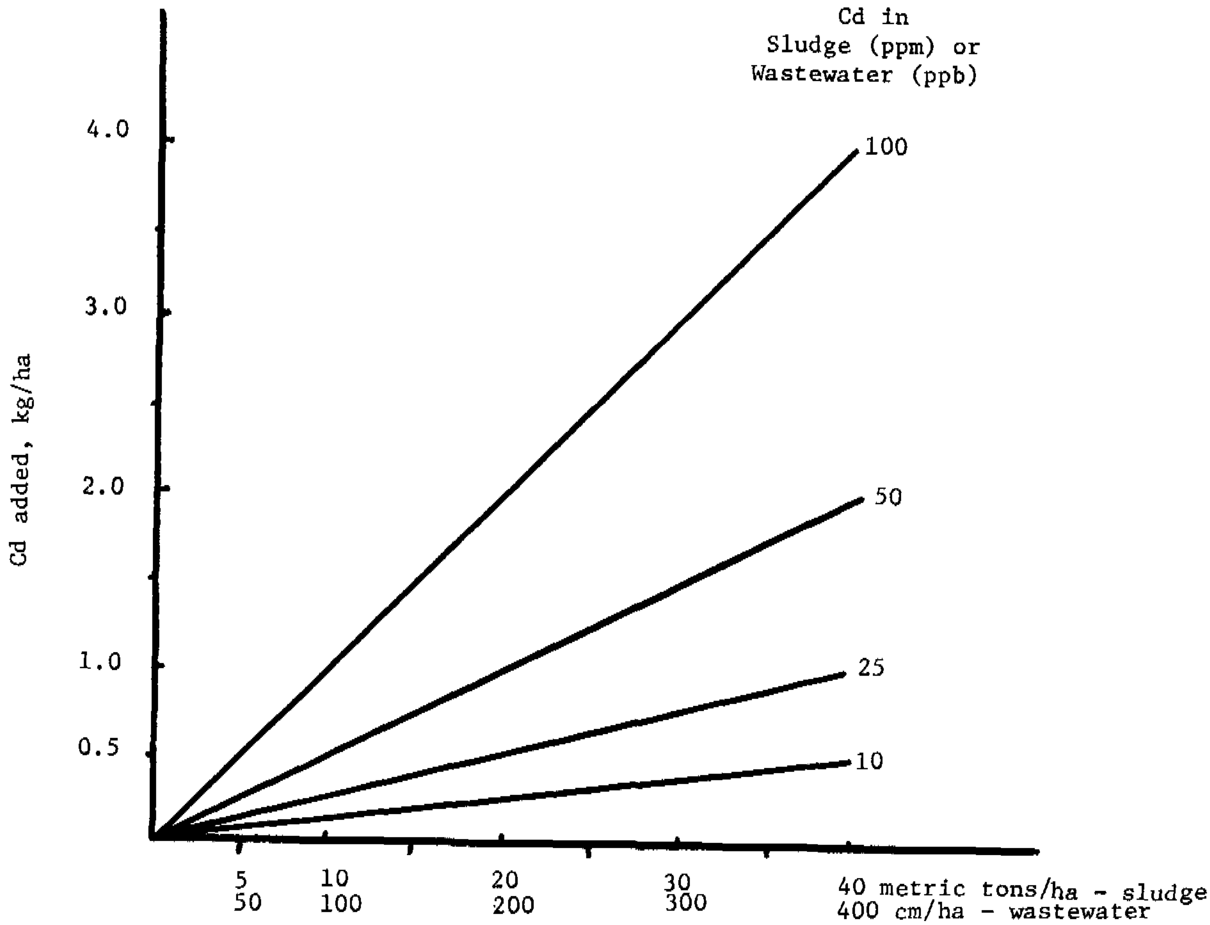

Fig. 1 - Annual Additions of Cadmium to Land (calculated)

persistent compounds such as PCBs and chlorinated hydrocarbons. Based on the information being developed under the Toxic Substances Control Act, more than 30,000 different commercial chemical products, mostly organic, are being manufactured in the United States. It appears probable that many, if not most, of these chemicals can be found in wastewaters together with all their by-products, metabolites, oxidation products, etc. (13).

Sewage treatment is often combined with chlorination of the effluent prior to land application. This procedure is obviously desirable microbiologically, since the treatment may reduce the number of pathogenic microorganisms present. However, chlorination also results in the formation of a wide range of chlorine- and other halogen-containing derivatives which can then be found in the wastewater and often in drinking water (14).

Chlorination of natural organic as well as industrial substances has been well documented. An example is the surprising finding that chlorination of humic substances in water yields chloroform and other lower halocarbons. In addition to those chemical substances that have been discovered in wastewater in recent investigations, large numbers of more hydrophilic compounds are believed to be present. These latter compounds are very difficult to detect with current analytical methods.

Because wastewater thus can contain a wide range of known and suspected organic contaminants, risks involved must be carefully considered before its application to land.

Microbiological Contaminants

A review of the potential microbiological problems associated with the application of sewage containing domestic wastewater to land has been prepared by Larkin et al., and appears in Volume 2 of the Proceedings of this Symposium. Essentially, an assessment of the hazards involves consideration of the kinds of microbial and parasitic pathogens in domestic wastewater, the probability of their survival during any treatment the wastewater might receive, the manner and schedule of application of wastewater to the crop, the processing the crop receives before consumption, and, finally, the infectious dose of the pathogen in question.

When these potential problems are considered, it is apparent that there is a good probability of human pathogens reaching the crop through application of domestic wastewater, and the persistence of some of these pathogens is such that they could remain viable until the food is either processed or consumed. Because of this possibility, we believe that wastewater containing domestic sewage should not be applied to crops that are normally eaten raw. Furthermore, because of the persistence of some parasites, land that has received wastewater containing domestic sewage should not be used to grow crops normally eaten raw for 3 years after the last application of wastewater. In addition, unless it is free of pathogenic microorganisms, domestic wastewater should not be used on crops that enter the home or restaurant in the raw state (even when the food is to be cooked before consumption) because of the potential for cross-contamination.

MECHANISMS OF POTENTIAL FOOD CONTAMINATION

Given the presence at significant levels of some or all of the contaminants discussed above, what are the ways in which they may enter the human and animal food chain?

Wastewater application methods are important. Various types of overhead sprays are usually employed where crops are planted, but other techniques include ridge and furrow irrigation and overland flow. Some of these latter methods may not be especially adapted to the planting of crops, at least in some parts of the country.

Physical Contamination

Chaney and Lloyd (15) have shown that sewage sludges are effectively retained when applied to vegetation by spraying, and that they are not well removed by rain. Because of the more dilute nature (and lower viscosity) of wastewater and effluents, lower contamination levels on vegetation would be expected. However, evaporation and repeated application may provide sufficient buildup of contaminants to be a cause for concern. Contaminants physically retained on crops may be ingested by animals, or possibly directly by humans, if improper practices are followed (16).

Uptake from Soil

Much research has been conducted on the uptake of metals such as cadmium, lead, zinc, nickel, and copper from soils. In recent years, such research has accelerated because of concerns about the use of sewage sludges on land (10). As discussed above, wastewaters would provide lower concentrations of such metals. However, some metals may also be more readily available from wastewaters, since they are mostly present in dissolved form. For this reason, additional studies with realistic and exaggerated concentrations should probably be conducted to determine metal uptakes.

The uptake of organic compounds such as pesticides and PCBs from soils into plants has also been studied, but on a more limited scale. Some pesticides are taken up, but others are not or are taken up only at low levels (17, 18). The pesticides heptachlor, dieldrin, and chlordane are taken up by soybean plants from soil, translocated to the seed, and stored in the oil, but at low levels (19). PCBs, on the other hand, are not carried from soil to the top of plants if direct vapor transmission is prevented by a barrier over the soil (20). Uptake of PCBs by root crops, however, has been demonstrated under field conditions (21).

In addition to the uptake of organics and metals, two research groups and possibly a third have shown that virus uptake through the root system of plants is possible. This would result in internal contamination of the plant with viruses (22).

Foliar Uptake

The subject of foliar uptake and translocation of sewage-borne contaminants by plants and food crops is an area of considerable interest, but few if any direct research studies have been conducted. The development and evaluation of commercial herbicides, however, has prompted extensive studies on the relationship between chemical structure, physical properties, and the behavior of these chemicals in and on plants. Although no simple relationship was found between the variables involved, it is apparent that many chemicals are effectively taken up from leaves, and that others are also taken up via roots or shoots. Some compounds may be retained in the area to which they are applied or may be readily translocated to other parts of the plant. An example is the application of diquat, a bipyridylium, to beet leaves: 33% of the amount applied is retained within 30 seconds and can not be washed off (23). Some herbicides, such as 2,4-D, are notorious for their ability to migrate to most parts of a plant when applied to either the foliage or soil.

As in many processes of this type, the concentration of the particular contaminant in the liquids or wastes applied is of importance in foliar uptake and very low concentrations do not appear to be a major cause for concern. A possible exception here would be extraction or transport phenomena by which such substances would be selectively removed and retained by the cuticle on crop and plant leaves.

In sewage treatment processes, many organic substances are partly metabolized, modified, and broken down. These alteration products are usually more water-soluble than the original chemical. Normally, such substances may be of less concern environmentally than the more fat-soluble materials because of decreased biomagnification potential through the food chain. However, when such metabolites or degradation products are applied to plants, absorption, retention, and/or distribution to edible portions may result.

Direct Uptake by Food Animals

Cattle and other animals ingest soil as well as plants while grazing. Chemical contaminants, especially metals and PCBs, may build up in selected tissues of such animals when sewage sludge is used on land and pastures (24). Although wastewaters may present a lesser problem, those from highly contaminated sources may result in the buildup of contaminants in the surface layer of the soil similar to that observed with sludges.

There have been sporadic reports of animal disease from grazing on land treated with wastewater or sludge. As an example, a recent case in Pittsylvania County in Virginia involved human infestation with tapeworms from cattle that had pastured on land treated with digested sludge (25).

Aerosol Drifting

Spray application of wastes often results in the drifting of aerosolized materials for relatively long distances. Drifting may result in the contamination of crops on neighboring fields, which may not have been selected for the application of that particular waste (of concern would be, for instance, crops to be eaten raw such as lettuce and strawberries). Primary hazards here would be microbiological, though chemical contamination by foliar uptake, retention, or other mechanisms should be considered.

CONCLUSION

Use of wastewater on land may or may not be a safe procedure. Much depends on the origin of the wastewater, the treatment process used, the crops planted, and above all on the quality of the planning and care. All these factors will determine whether such practices are the desirable recycling of a resource or a public health liability.

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PUBLIC, INSTITUTIONAL AND TECHNICAL ACCEPTABILITY

OVERVIEW OF PUBLIC, INSTITUTIONAL AND TECHNICAL ACCEPTABILITY OF LAND TREATMENT OF WASTEWATER

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ABSTRACT

Land treatment of sewage has been called a "Paper Tiger" in the words of a sanitary engineer. It might more appropriately be called a "Caged Tiger" about to break out of its cage in view of the substantial support for the concept reflected in legislation from the Congress, pronouncements of policy positions from top administrative levels of the U.S. Environmental Protection Agency and actions of citizens groups. A substantial history of opposition from the sanitary engineering profession whose training predisposes engineers towards conventional treatment processes is the major problem the concept faces. That attitude tends to prevail in many of the Regional Offices of EPA and State environmental agencies. But changing National goals and needs have focused attention on recycling concepts in recent years. The ability of recycling systems to provide clean water at low cost in a sound manner while accomplishing other desirable objectives should mean that land treatment of sewage will gain wide acceptance in the next few years.

Introduction

Chairman Walker, ladies and gentlemen, I am pleased to have the opportunity to participate in this International Symposium on Land Treatment of Wastewater. I have spoken on the subject of land treatment in several parts of the Country over the past few years, but never in a more attractive setting. A friend in Washington who attended Dartmouth said I would like the territory and I do. It has only the drawback of needing to be leveled somewhat to accomodate center pivot irrigation rigs. Well, sometimes trade-offs are necessary.

I'm not really that much of an advocate of land treatment. I don't think it is a panacea. I doubt if Barrow, Alaska, and a few mountainside communities in Switzerland can use this approach. Of course, I'm not <u>sure</u> they couldn't.

There is an interesting psychology about land treatment. It needs to be checked out with Freud's disciples some day, but there is just one city in this Country that I visit quite often where I don't feel guilty about using the restroom facilities. That is Muskegon, Michigan. Any place else I wonder who is down stream. At Muskegon I wonder how many bushels of corn I've just produced.

In the Metropolitan Washington area we worry about who is upstream. And we've been forced to worry even harder about upstream the past few years because some local and state officials have wanted to build a 228,000-cu m/day (60 mgd) AWT facility above our drinking water intakes on the Potomac and prove that past worries about drinking treated effluent have been unfounded. But you know many top level EPA officials live in the Washington area. And maybe they don't believe all those glowing reports that come from Technology Transfer about AWT. So at least for the time being that AWT system is dead; killed by Russell Train and buried under Doug Costle, so to speak. More about that later.

I'm not supposed to talk about AWT. My talk title is "Overview of Public, Institutional and Technical Acceptability of Land Treatment of Wastewater." That covers a good bit of territory. Land treatment itself covers so little territory that the Water Pollution Control Federation Journal in January called land treatment a "Paper Tiger." I would prefer to call it a "Caged Tiger" and the reasons will become apparent as I proceed.

Problems with sanitary engineers

Let me talk about institutional acceptability of land treatment first, and begin with the sanitary engineering profession. Part of what I am going to give you will be personal impressions as I have worked with the subject for the past eight and a half years, since first becoming exposed to the subject in the fall of 1969 at Muskegon.

Most of the sanitary engineers that I have dealt with in that period, with certain notable exceptions, suffer from what I have chosen to call acute myopia. They were brought up on slide rules, calculators, plumbing systems and reinforced concrete. They were taught how to arrange pipes and design activated sludge and trickling filter sewage treatment plants. They were taught that rivers have assimilative capacity and are meant to be used as part of the sewage treatment process.[1]

I believe it is reasonable to say that sanitary engineers are strongly influenced by public health officials many of whom are also sanitary engineers. The activated sludge and trickling filter systems they design were first developed to solve health and esthetic problems.[2] They reduce BOD, thus reducing oders, and make it cheaper to chlorinate effectively, to reduce pathogen levels. They reduce nitrogen levels[3] inadvertently, but just to make certain too many problems are not solved they produce substantial biomass in the form of sludge to give operators another source of headaches.

Sanitary engineers in EPA, state agencies and the consulting engineering profession like these systems that provide what is called conventional secondary treatment. They believe they are the "Best Practicable Treatment Technology" to meet the "Best Practicable Treatment" goal of Public Law 92-500 for July 1, 1977. They don't produce clean water, but they are easy to build. They don't produce clean water, but they are not too expensive. They don't produce clean water, but the engineers have the plans ready after they design the first one so it is just a little bigger here or a little smaller there on the blueprints, an adjustment here and a minor change there for each additional facility. They break down and are subject to erratic performance, but engineers know how to design them.[4] They don't produce clean water, but it is easy to locate them down hill where they collect sewage cheaply and can discharge to a river, usually in an industrial area or a low income area which has grown accustomed to dirty water over the years. Most sanitary engineers seem to believe such systems are just fine. And they certainly don't want to be bothered much with land treatment even though required by NEPA and PL 92-500 to consider viable alternatives to conventional systems.

Sanitary engineers have long been reluctant to embrace land treatment as a viable alternative to conventional processes.

Occasionally the attitude of the profession shows up in print. In November of 1972 in a letter[5] to the Mayor of Miami, the Florida Department of Pollution Control said among other things about land treatment, "EPA has not yet approved it as a means of disposal of sewage effluent. Also, we of the State Pollution Board, will not approve any plan that uses recycling---. Therefore, without approval and without funds recycling is not a workable way of disposing of sewage effluent." And later in the letter, "The idea of land disposal in South Florida is patently rediculous."

EPA found its own interesting excuses which are described in a letter to Congressman Guy Vander Jagt in 1974 after the Congressman had challenged the agency on issues related to South Florida.[6] EPA's letter said, "Large scale utilization of conventional (sic) land disposal systems in Southeast Florida were (sic) held not to be practical, due to the large area of land required (approximately 8,100 hectares [20,000 acres] per 380,000-cu m/day [100 mgd] for crop irrigation), the infiltration of saline waters into the wastewater collection system, and the secondary ecological stresses caused by storage of excess water during wet seasons and the disposal of excess reclaimed irrigation waters."

My impression at the time was that the State Agency was justifying its failure to give serious consideration to land treatment, and the Atlanta Regional Office of EPA was justifying its ignorance of the subject of land treatment. It was fascinating that Tallahassee, Florida, was successfully irrigating crop land with treated effluent at the time and had been doing so since 1966.[7] Also fascinating at the time was an article in the Christian Science Monitor describing the oxidation of organic soils in South Florida and an observation that "Using massive amounts of treated sewage and garbage from the nearby Florida Gold Coast, the soil could be rebuilt."[8]

There are signs that the Atlanta Regional Office of EPA is changing its attitude in that it has approved for construction a 76,000-cu m/day (20 mgd) forest irrigation system for Clayton County, Georgia. The system will re-charge a drinking water supply aquifer.

In Muskegon County, Michigan, (When I worked for Congressman Vander Jagt who represents the County) we ran into many obstacles including one represented by a statement early in the step 1 process attributed to two members of the Michigan Water Resources Commission that the Commission should, "Give them enough rope to hang themselves, but not enough to hang us, too,"[9] referring to approval of the construction plans for the Muskegon County land treatment system. To paraphrase Winston Churchill, ---some rope --- some hanging!

Another interesting observation about Muskegon was made in the Water Pollution Control Federation Journal article of July, 1973, referred to earlier, when the author said, "Professional engineers and agronomists warned that the land disposal plan was extremely costly, might cause irreparable ecological damage, could cause hygenic hazards, and in all likelihood, would never create the agricultural economy its promoters claimed it would. But the colorless facts, statistics, and cost estimates were no match for flamboyant academic theories, claims, and predictions."[10] One of those "colorless" cost estimates was prepared by consulting engineers for the City of Muskegon as a competitive alternative to the County system. They said the 160,000-cu m/day (42 mgd) County system would cost $72 million when in fact it cost $45 million. Michigan State University's Agriculture Department predicted crop income at $10 per acre.[11] And, of course, the system has produced about 140,800 hl (400,000 bushels) of corn on 2,000 ha (5,000 acres) each of the past two years with net income of more than $500,000 each of those years from corn sales reducing operating costs of the system by roughly one-third. Those operating costs are far lower than the costs for other systems which do not provide the high quality water Muskegon produces.[12]

Consulting engineers for the city of Owosso in Michigan designed a physical-chemical system for the city that will cost 30% more than the cost of land treatment. The consultant's estimates indicated land treatment to be substantially more costly, but an EPA review found the estimates too low for the physical-chemical system and too high for land treatment.[13] Unfortunately, EPA believes the construction grant process has proceeded too far to require the city to abide by EPA cost-effectiveness requirements, or so they tell me.

Local officials along the Gulf Coast of Mississippi are currently opposing a plan for sewage treatment proposed by a team of state, federal and private "experts" which would discharge effluent into brackish coastal bays thus threatening marine life. The Governor of Mississippi's coordinator for water quality planning is quoted as saying "Land disposal would require a 200 ha (500 acre) lagoon of pure sewage large enough to sail a yacht on" and "another lagoon would be required for resettling, and 2,000 ha (5,000 acres) would be necessary to pump water onto, surrounded by a 4,000 ha (10,000 acre) buffer zone."[14] Incredible.

Michigan changes

I believe State Government in Michigan has substantially improved its posture in the past several years with respect to land treatment. There is not much evidence to suggest that the sanitary engineering profession in the state has changed, but Dr. Howard Tanner, new as the Director of the Michigan Department of Natural Resources in 1975, is a forceful advocate of resource recycling. In a speech to the Michigan Water Pollution Control Association in 1976, Dr. Tanner said, "I am convinced we must take a fundamentally different direction -- in our water pollution control efforts --. We must commit ourselves to and follow the principle that wastes of all kinds must be recovered and reused -- recycled -- if we are going to survive and thrive over the long haul." Tanner also said, "The real problem lies in fitting this concept into our economic, social and cultural patterns. Solving that problem will be an extremely difficult job, calling on all the ingenuity we can bring to bear. It is a job of information and education, of innovation and demonstration. It will take many years to accomplish, the going will be slow and difficult, and there will be many obstacles and setbacks. But it's a job we must do regardless." In his concluding remarks he said the most important responsibility of State Government in water pollution control was to provide "leadership in development of the new recycling technology I have been talking about."[15]

Congress continues to lead

Perhaps most exciting in recent months has been the attitude of the Congress in its mid-course correction of PL92-500 in which the encouragement of resource recycling has been greatly strengthened. The action, in which the House-Senate Conference on amendments to the Act adopted virtually intact a bill introduced by Congressman Don Clausen of California, is virtually unprecedented.

Clausen's bill added a substantial carrot to accompany EPA requirements for the consideration of land treatment by providing 85% Federal funding for construction of land treatment systems versus 75% for conventional processes, by providing a 15% cost-effectiveness preference for land treatment and by making land used for storage of wastewater an eligible cost.

Fortunately, at least at higher levels of EPA, there is also a firm recognition of the need for at least some limitation on the luxury, or call it extravagance or wastefullness, in certain kinds of convenience. Bottle deposit legislation to encourage recycling is not making progress in the Congress because of powerful entrenched interests although it has gained acceptance in some states in recent years and months. On the other hand, the recycling of nutrients in sewage has been strongly encouraged by both the Congress and the Administrators of EPA in Washington.

In 1972 Public Law 92-500 was enacted and contained the first language providing strong encouragement for the recycling of nutrients in wastewater, in the amendments offered by Congressman Vander Jagt. But the EPA Regional Administrators, the states and consulting engineers essentially ignored that language. In response to admonishments from Vander Jagt, John Quarles, then Deputy Administrator of EPA, on November 1, 1974, issued a memorandum to the Regional Administrators calling on them to "do a better job in assuring that land treatment is given full and adequate consideration as a possible method for municipal sewage disposal in projects funded with Federal grants."

Also during those several years following enactment of PL92-500, Belford Seabrook, in EPA's Office of Water Program Operations, was able to stir EPA to fund a number of land treatment oriented studies which have helped to provide essential technical background information for the implementation of the law.

In 1977, with the appointment of Douglas Costle as Administrator of EPA and Tom Jorling as Assistant Administrator for Water and Hazardous Materials, a stronger policy in support of land treatment evolved at EPA. In testimony before the Senate Subcommittee on Environmental Pollution on June 30, 1977, Jorling said, "Regarding the future of the program EPA sees two primary themes in pursuing its mission of water pollution control: (1) to control toxic pollutants and (2) to encourage the conservation, reuse and recycling of both water resources and the valuable elements contained in the various streams currently being discharged to the nation's water."

On October 3, 1977, Mr. Costle issued a memorandum to Assistant and Regional Administrators in which he said, "<u>Therefore, the Agency will press vigorously for publicly owned treatment works to utilize land treatment processes to reclaim and recycle municipal wastewater</u>." He also said, "<u>If a method that encourages water conservation, waste water reclamation and reuse is not recommended, the applicant should be required to provide complete justification for the rejection of land treatment</u>."

So it is very apparent that the highest levels of EPA support recycling systems.

EPA has a Regional Office problem

However, it is also apparent that slow progress is being made at the Region level of EPA in some instances.

Region V of EPA, for example, published a position paper last year urging a ban on phosphates in detergents in all of the states in the Great Lakes Basin to help correct eutrophication problems in the Great Lakes.[16]. Particularly revealing is the 2nd paragraph of the <u>Introduction</u> to this paper on page 1 where EPA Region V says, "This position is a departure from the EPA policy on phosphorus control from 1971 to present. The policy was to rely on chemical treatment to reduce phosphorus levels in municipal sewage and industrial wastes discharged into waterways. Because this policy has failed to achieve the water quality goal of sufficiently decreasing and stabilizing rates of eutrophication in both inland lakes and the Great Lakes, not only in Region V of EPA, but State governments and other regional and international agencies have recently been reconsidering the need and means of reducing phosphorus loadings. No other single factor is so important for the future water quality of the Great Lakes Region."

On page three this paper discusses the International Joint Commission (IJC) considerations on phosphorus which state that a 1 mg/l effluent limit, the present standard, is so high that eutrophication will not be stabilized. The IJC suggests that the level may need to be lowered to 0.1 mg/l for all municipal plants, but as an immediate measure urges a 0.5% limit for phosphorus by weight for detergents used in the Great Lakes Basin.

It is not mildly ironic that the IJC reports discussed by EPA in this phosphate ban paper indicate phosphate concentrations in the effluents from municipal sewage treatment plants discharging directly or indirectly to the Great Lakes exceed 0.1 mg/l phosphorus in every instance; that is, every instance but one: Muskegon County. The best level achieved in 1976 in discharges by Michigan municipalities directly or indirectly to Lake Michigan, into which the Muskegon system discharges indirectly, is 0.5 mg/l. The best level except for Muskegon, that is. Muskegon was .07 mg/l or seven times better than the next best system. And the unweighted average phosphorus discharge for 26 systems discharging to the Lake Michigan watershed in Michigan was 1.8 mg/l in 1976 or 26 times the Muskegon rate. Fifteen of the 26 systems discharged at levels above the 1 mg/l standard and none but Muskegon, of course, met the 0.1 mg/l recommendation of the IJC.[17] Was this astounding accomplishment acknowledged by the EPA Region V Phosphorus Committee? In the Committee's report there is no discussion, of any kind, of Muskegon success. Incredible.

I am reminded of discussions in 1969 and 1970, among those of us who became strong advocates of land treatment at Muskegon, that the 80% phosphorus removal requirement of the Lake Michigan Enforcement Conference would have to be made more and more stringent if the quality of the Lake were to be protected. Obviously a percentage removal requirement would be helpful only if total flows did not increase and the resulting removal in total pounds of phosphorus were sufficient to protect water quality. I believe I can state without serious challenge that the 80% removal standard reflected the capabilities of the state of the art for conventional sewage treatment processes, taking costs into consideration, rather than the requirements for clean water. Thus municipalities have been encouraged for many years to build conventional systems while there has been substantial knowledge that they would have to add increments of phosphorus removal eventually, an expensive deception and one which has been very detrimental to land treatment.

I am reminded that the Executive Director of the Michigan Water Resources Commission at that time argued that the Commission could not afford to fund Muskegon, which would solve the problems of the year 2000, when the Commission had to meet the needs of 1975. Unfortunately, according to the IJC the problems of the year 2000 had arrived in 1977 leaving every community but those in Muskegon County with the need to upgrade its conventional process to remove expensive increments of phosphorus.

But Region V of EPA has not yet recognized the implications in this situation. And the logic of its failure must once again be placed on sanitary engineer myopia.

This EPA Region has been willing to fund a special study on the potential uses in outstate Michigan of sewage sludge produced in the southeast part of the State.[18] Detroit now produces about 3,800-cu m/day (1 million gallons) of sludge, incinerates 1,520-cu m/day (400,000 gallons) and discharges the remainder to the Detroit River. The study, completed this spring, indicates it would be less costly to recycle the sludge on forest land than it would cost to "waste" it all by incineration which has been the city's plan. It is interesting to see the support for recycling of sludge which is occuring out of necessity: incineration would violate air quality standards in southeast Michigan. It will also be interesting to review Detroit's progress in meeting phosphorus removal standards.

The cost curve problem

There is an additional factor that bears discussion at this point which is related to engineer myopia. It is an attitude given substance several years ago by the Director of Research at Minnesota Mining and Manufacturing Company, Dr. Joseph Ling. It relates to the slope of curves showing costs for removing additional increments of pollutants when 100% removal is approached. This gentleman whose premise was the subject of a Wall Street Journal editorial in 1972 argued that it was relatively cheap to remove 80-90% of the phosphorus in sewage, but when one tried to remove higher percentages the costs increased at geometric rates. He argued very persuasively that it would take huge amounts of energy and chemicals to remove those high percentage increments, and that the pollution caused by the generation of that energy and the production of those chemicals would exceed the pollution reduction resulting from their use in sewage treatment. You have all seen those cost curves. I must confess I don't like them. They give too many people the impression that we can't afford clean water. And you know, they are right if we are chained to conventional sewage treatment processes or rely on artifically produced energy and chemicals to do the job rather than soils and sunlight that do the work cheaply!

Disposal philosophy plays role

It is my belief that this myopia has also evolved in part from the "disposal" philosophy which has prevailed in sewage treatment circles for the better part of this century. Conventional systems are designed to destroy oders and pathogens and dispose of what is left. The add-ons for nutrient removal follow this same "disposal" philosophy. Engineers are all too willing to design the add-on plumbing systems and they carry over the philosophy to physical-chemical AWT alternatives. This is probably part of a larger societal problem and perhaps it is unfair to imply that the blame for its implications lies with sanitary engineers. Of course, engineering schools apparently have taught little about natural processes for cleansing wastewater and there seems to be little association between agriculture and engineering education relative to the recovery of nutrients in wastewater.

Muskegon provides new standards

Most of the sanitary engineers and our friend from 3M seem to have no idea that we can get from where we are to where we want to be by an alternative approach. They insist on continuing to beat the old horse harder to make him go faster rather than switching to another form of transportation. At some point the horse collapses. It is at this point in the analogy where the public begins to believe it can't afford clean water because the engineers can prove it costs too much. Clean water from conventional processes often does cost too much. But clean water from land treatment, in my experience, does not. Even though it is a very old process, far anteceding conventional treatment, it is a breakthrough, in effect, in its modern application. Reuse is often less expensive than "disposal."

Look at Muskegon efficiency: 97% removal of phosphorus in 1976 and 1977. Other reductions are 99% BOD, 95% COD, 97% suspended solids, total nitrogen reduced to 2.5 mg/l, heavy metals, low to begin with, 88% and fecal coliform at least 99.9%.[19] What is more, Muskegon soils are mostly sand and consequently are not as efficient as heavier soils in removing pollutants.

These numbers are particularly impressive when one considers operating costs are far less than the costs for any alternatives which approach the quality of water Muskegon produces. And construction costs were not high either at about $1 per gallon of treatment capacity, equivalent to the rule of thumb costs for conventional secondary treatment at the time.[20]

I won't spend any additional time discussing Muskegon because you will hear more about it from another speaker.

What the system illustrates is a unique leapfrog in technology whereby modern equipment dovetails with natural processes to produce synergistic effects: clean water and 140,800 hl (400,000 bushels) of corn.

Land treatment isn't complex

I am going to discuss one more aspect of the sanitary engineering syndrome. At the Water Pollution Control Federation Seminar on the 1977 Clean Water Act in March I found myself in a rather heated argument with a consulting engineer from Pittsburg who thought the 1977 amendents had been produced by individuals who knew nothing about sewage treatment. Having had something to do with the legislation I felt a certain need to respond. His principle contention was that it is difficult enough to train people to operate conventional systems, let alone expect to find people to manage Muskegon type facilities. I didn't try to offend him by asking what troubles he would anticipate in the operation of physical-chemical and other AWT facilities, none of which have worked right in this country as far as I know, that is if they could be built to work as designed, or if the community could afford to operate them.[21]

The point, though, is that land treatment can consist of simple aeration, simple storage, and simple irrigation combined with simple farming. We already train thousands of farmers every year on our farms and in our agriculture schools, irrigation is a well developed activity and the sewage pre-treatment needs to by accomplished anyway. But let me add one other element here that I believe is very important. Land treatment, in contrast to conventional systems, is a <u>positive</u> process. It <u>produces</u> something at the same time it <u>cleans</u> water. It is not a negative, "disposal" process. It makes use of pollutants. It converts them to beneficial use. It returns nutrients to the use cycle. It doesn't produce a slimy, gray-green river at its outfall, but tall, green stalks of corn or broad fields of hay or luxurient forests. In further contrast to conventional processes, it can be said that the cleaner it returns water to lakes and streams the less it costs. In other words, there is room for real credit to those who operate land treatment systems efficiently. The attitude at Muskegon is justifiably one of positive, substantial, measurable accomplishment. Perhaps it would not be so difficult to get good people to run conventional sewage treatment plants if they could be made to produce clean water.

I find myself automatically giving a sales talk for land treatment, and to an audience that already understands the subject. But maybe this just represents a tendency to respond to a strong feeling that those of us who do understand the concept must go out and sell it to the public, to governmental institutions, to educators and to the engineering profession.

Citizens' efforts very important

That leads me to a few observations about the general public. Where there has been a strong generation of public interest in land treatment it has come initially for the most part from a very few people, outside the normal wastewater management infrastructure, who have taken the time to develope an understanding of the concept. Perhaps that is true of any new concept, or in this case in the new application of an old concept. Of course, there aren't too many people who spend any time at all talking about sewage although I believe the number is growing as more become aware of the implications of sewer lines on all of the issues relating to growth and development.

There seem to be a number of motivations for those who do, however. But essentially citizens who become advocates appear to accept the philosophical premises on which land treatment is based, that is: the environment is a single unit in which air, land and water interact; the earth is a closed system and planning for those activities with a potential to degrade the environment must similarly strive to achieve closed systems; and pollutants are potential resources out of place.[22]

These premises may not be vocalized as such by individuals who support the concept. But in the long run they seem to play a quite decisive role in citizen support for the concept. Of course, if the premises are sound one would expect that to be the case.

For the time being, until the sanitary engineering profession makes a greater accomodation to the concept, citizen advocates will probably continue to find themselves in a position which requiresconfrontation with public officials and direct challenges in hearings to gain proper consideration of the concept. I am aware of a number of efforts of this kind over the past six years since the enactment of PL92-500, a few of which have been successful, but unfortunately many have failed essentially because state agencies and EPA Regional offices have rubber stamped the findings of engineering consultants. As I pointed out earlier in the case of Owosso, Michigan, even after EPA, Washington, conducted a review, the agency chose not to require that its findings be implemented.

However, there is one example of a substantial degree of citizen success of which I am aware that I will briefly describe and that has occured in Montgomery County, Maryland. As long ago as 1971 citizens began to object to the planned construction of an AWT system which would have discharged to the Potomac River. They argued that the system threatened the area's drinking water supplies, was too expensive and that the capacity of 228,000-cu m/day (60 mgd) was not all needed. They also argued that the consultant's land treatment study was inadequate. Several years of struggle and a gradual change in EPA Region III's attitude finally resulted in 1976 in the decision by Russell Train, which I mentioned earlier, not to fund the AWT facility.[23] At the same time Train called for additional study of land treatment in Montgomery County. County officials then proceeded to have such a study performed which was completed in June of last year and indicated land treatment to be feasible in the County and that it would cost substantially less than the AWT alternative. State and local officials took EPA to court to force approval of the AWT system funding, but lost that decision in April of this year.

While land treatment systems have functioned satisfactorily in several parts of the County and other small systems operating without problems exist in other parts of Maryland, the State has precluded substantial use of the process with requirements for pretreatment of sewage to drinking water quality before irrigation on cropland. The added cost of such pre-treatment, of course, is a major deterrant to its use and seems highly unjustified judging from the experience of other states. It is noteworthy that in order to protect shellfish waters Maryland requires the use of land treatment systems for discharges immediately upstream. It doesn't apply that stringent standard for discharges above drinking water intakes, however. Incredible.

It remains to be seen whether land treatment will be implemented in Montgomery County, but the effectiveness of citizen action is well demonstrated.

Conclusions

It is very apparent to me, and I am certain to you, that land treatment systems for sewage are not about to take over the Nation and the world in the near future. No one is prepared to make the committment to clean water and the conservation of resources that would be required to establish land treatment as a panacea, at least in the foreseeable future. But we are making substantial progress. Ten years ago no one in this Country but Penn State and the southwest gave the subject serious thought. Now we have strong support for the concept in the Congress, at the highest levels of EPA, in the Corps of Engineers, from environmental organizations led by Clean Water Action, a Washington, D.C. based grass roots, citizens action group and from many other citizens' groups. With Muskegon to grow from we are beginning to see a number of additional large systems being given serious consideration. The sanitary engineering profession has frequently been able to keep the "tiger" in the cage, but because the concept is grounded in sound principle, no pun intended, I am optimistic that we will see substantial growth in utilization of the concept in the next few years. It deserves to grow as the Nation recognizes the need to avoid waste and encourage reuse of its resources and finds that in the process it can have clean water that it can afford.

Footnotes

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PUBLIC, INSTITUTIONAL AND TECHNICAL ACCEPTABILITY

EXPERIENCES WITH BROAD-BASED LAND TREATMENT PLANNING IN PRINCE GEORGE'S COUNTY, MARYLAND

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ABSTRACT

Broad-based planning for land treatment is a no-constraints approach. It allows for all constraints, including the legal, regulatory, institutional, social, and planning considerations to be held open in order that a firm range of alternatives may be developed. Also included in this approach is an intensive public involvement program.

The planning effort for Prince George's County, Maryland, serves to illustrate the various facets of this approach. The result was a wide variety of land treatment options developed for consideration in that county for wastewater management, both present and future.

INTRODUCTION

Land treatment systems are generally more integral to community development than are conventional wastewater treatment systems. At the same time, these system exhibit the potential for a wider range of social benefits, depending on how the project is conceived and designed. Consequently, a broad, more comprehensive approach to wastewater planning is desirable--one which involves a great number of different disciplines and considers a wide range of different concerns and ideas.

The preliminary feasibility study of land treatment in Prince George's County, Maryland, is an example of how such an approach can be used. This study, which was conducted independently of the federally-funded Section 201 facilities planning program, addressed the full range of land treatment possibilities available to the county, including:

- Systems that will meet wastewater treatment requirements while concurrently optimizing other objectives such as maximum agricultural production and careful land use planning

- Wastewater treatment alternatives of both regional and local significance

- Alternatives that are consistent with ongoing facilities planning efforts and those that are primarily long-term options (20+ years) outside of current facilities planning projects

In examining the experiences with land treatment planning in Prince George's County, it is necessary to consider both the existing situation within the county and the expected conditions that would result from implementation of study recommendations. Of particular interest, also, is the institutional program established by the county for interacting with the public and review and implementation of various portions of the study.

EXISTING SITUATION

Prince George's County is a rapidly growing residential and employment center within the metropolitan area of Washington, D.C. (see Figure 1). From 1960 to 1974, its population nearly doubled to 684,000. This high level of growth taxed the capacity of all county services, including wastewater transmission and treatment facilities. The identification of public health hazards from continuing raw sewage overflows caused a large area of the county to be placed under a sewer moratorium by the Maryland Secretary of Health and Mental Hygiene. The moratorium severely limited new sewer hookups in major drainage basins within the county. (The construction of new transmission lines, expanded and upgraded treatment plants, as well as the correction of infiltration/inflow problems allowed the lifting of this moratorium in the Spring of 1978.) At the same time, increasing levels of treatment were being required by state and federal regulatory agencies to improve water quality conditions in the Potomac estuary. These treatment requirements included a high degree of nutrient removal, BOD reduction, and suspended solids reduction. Several major treatment plant projects have been planned to help alleviate these problems. For the most part, these projects are expansions and upgradings of the existing conventional wastewater treatment facilities (shown in Figure 2). However, the consequent rise in sewer service rates caused the county officials to become concerned and was one of the factors which led to this study of alternative treatment systems.

The county's rapid growth also impacted the local agricultural community, which is undergoing extensive social and economic change. Total farmland in production, as defined in the Census of Agriculture, has been reduced from 113,965 acres in 1964 to 70,272 acres in 1974 (a 38% decrease). Moreover, the number of working farms declined by 42% over the same period. Much of the remaining "farmland" is being held for speculative purposes.

The Maryland-National Capitol Park and Planning Commission's 1964 General Plan called for urban development to be concentrated along three basic corridors in Prince George's County, radiating outward from Washinton, D.C. Between the development corridors were to be "green wedges" of open space, agriculture and low density development. However, wedge implementation has not been very successful and there has been a substantial intrusion by urban uses into these areas. It was believed that land treatment systems in the county could help to preserve some remaining open space in the wedges and retain growth in the corridors. Such systems, at the same time, would provide a very high quality of wastewater treatment for the corridor development areas. These systems would also combine the resources of land, water, and nutrient value to provide a counter-incentive to preserve agricultural uses in the county. It should be noted that this concept is consistent with the objectives of the water quality planning process defined in the Clean Water Act of 1972 and in PL 95-217, amendments to the act.

APPROACH

In developing land treatment alternatives for Prince George's County, a no-constraints planning process was used. County goals and objectives were identified, and the ability of land treatment to help meet these objectives was assessed. Other benefits evaluated included the cost effectiveness of wastewater treatment, the production of cash crops through nutrient recycling, the preservation of agriculture land and open space areas, and the implementation of other land use planning objectives included in the general plan. Development of alternatives focused on those options that would be compatible with existing land ownership and current agricultural activities.

As can be seen in Figure 3, the process of identifying land treatment sites and alternatives was an iterative one. Land treatment opportunities were first conceived in general terms and then refined in several cycles on the basis of technical evaluations and assessments of planning implications.

One of the most significant features of the approach utilized is the conceptualization of land treatment opportunities solely on the basis of soil types, topography, and land use, while at the same time remaining independent of wastewater treatment needs. This enabled land treatment to be considered in a much more innovative manner. The experience of the authors was that where identification of treatment needs is established as a starting point, issues such as extent of

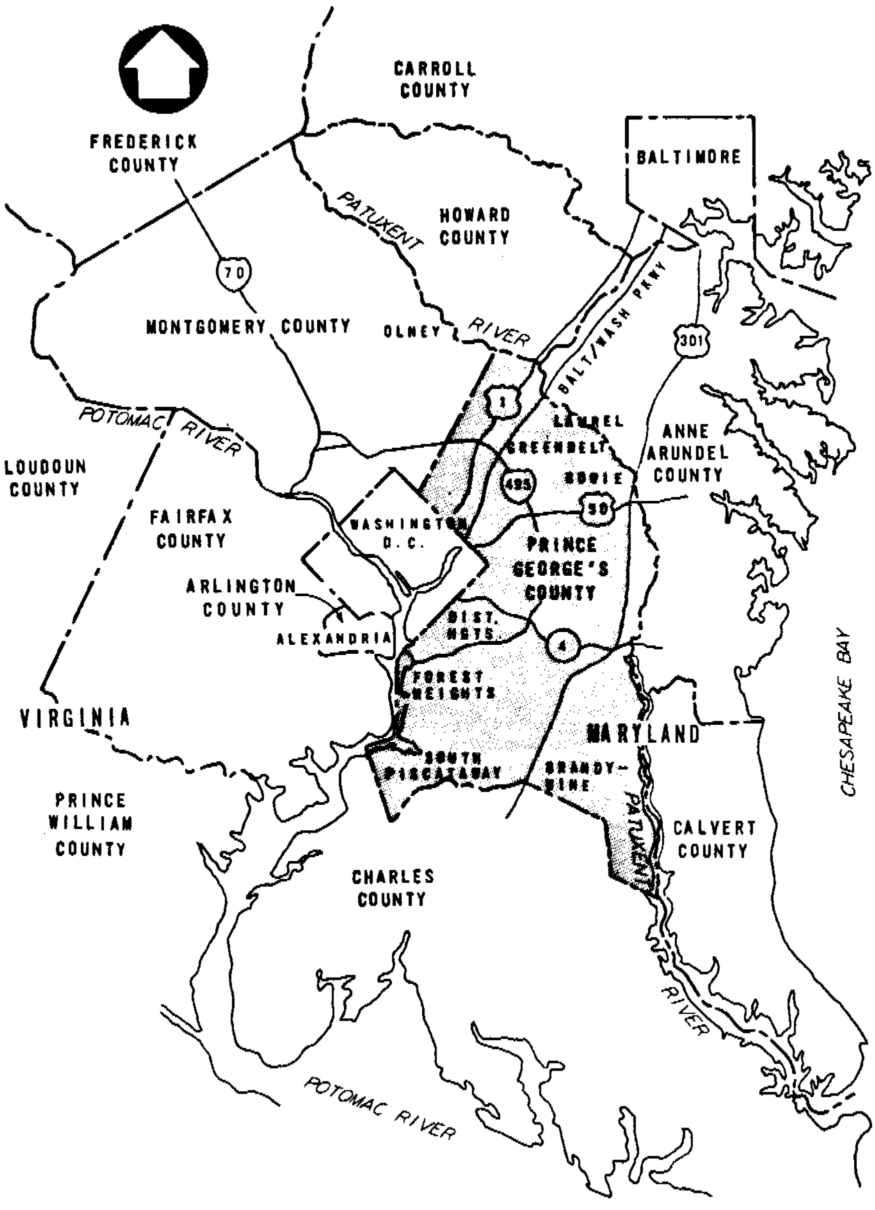

Figure 1. Location of Prince George's County, Maryland

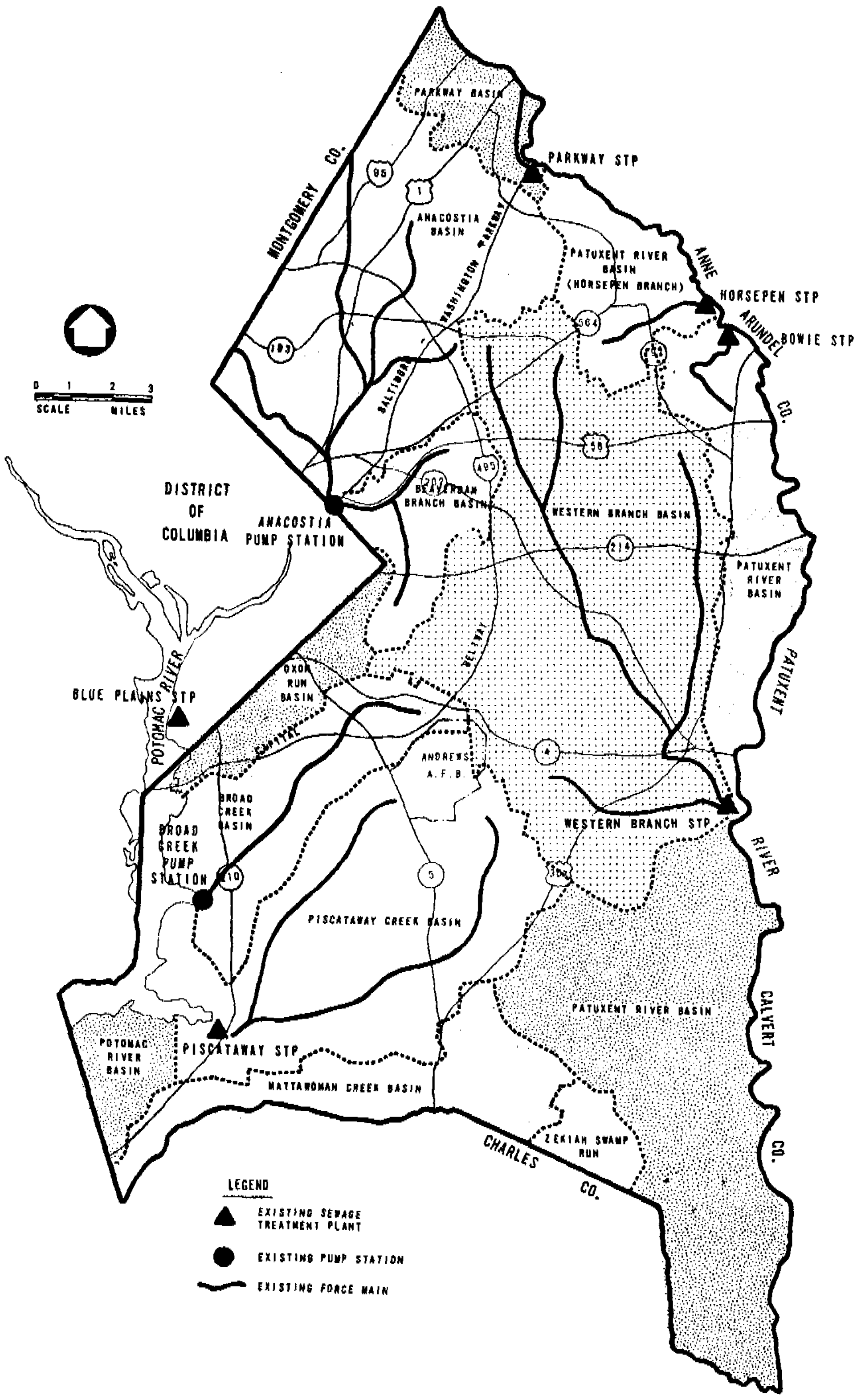

Figure 2. Major Wastewater Systems in Prince George's County, Maryland

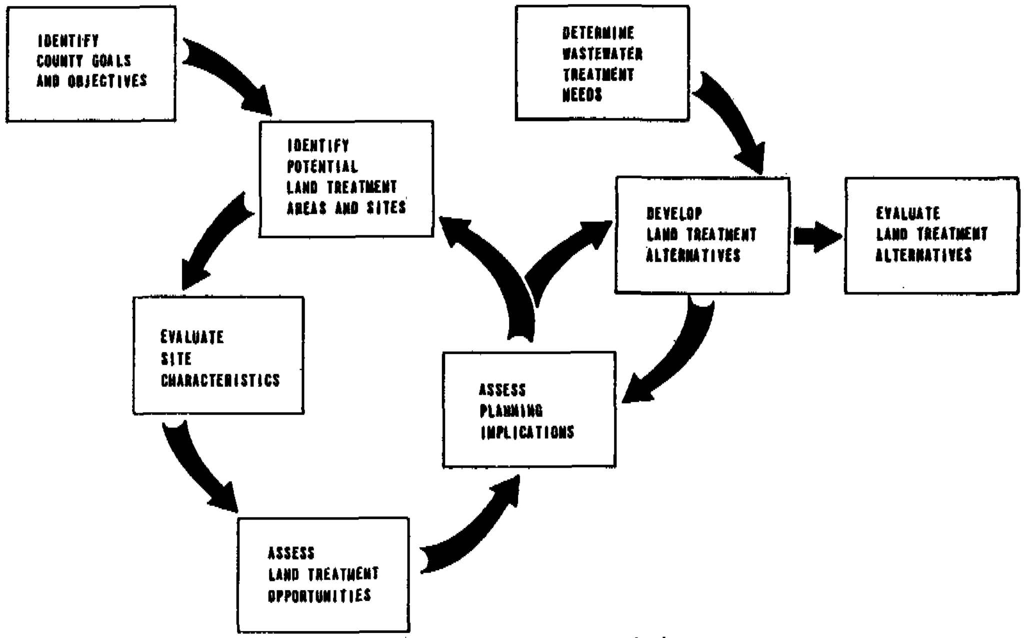

Figure 3. Planning Sequence Used in the Development of Major Land Treatment Alternatives

service area, rates of flow, and size and location of potential sites may become prematurally fixed and therefore less flexible in later analysis of strategies.

An integral part of the planning process was a comprehensive interaction program with public officials and citizens' groups, as well as industry and the agricultural community. The reasons for this program included:

- Initiation of a continuing dialogue between various facets of the community
- Initiation of the education process necessary to explain implementation of land treatment alternatives
- Solicitation of the views and concerns of the community in the development of alternatives to encourage public review and acceptance of any proposal that might result

Openness and flexibility of planning was emphasized throughout this program. From a technical standpoint, this approach called for full consideration of all applicable technologies. Each of the three methods of land treatment (shown in Figure 4), slow rate process (irrigation), rapid infiltration, and overland flow, as well as various combinations of processes, were examined. Conventional wastewater treatment unit processes were also examined, as were combinations of conventional and land treatment systems.

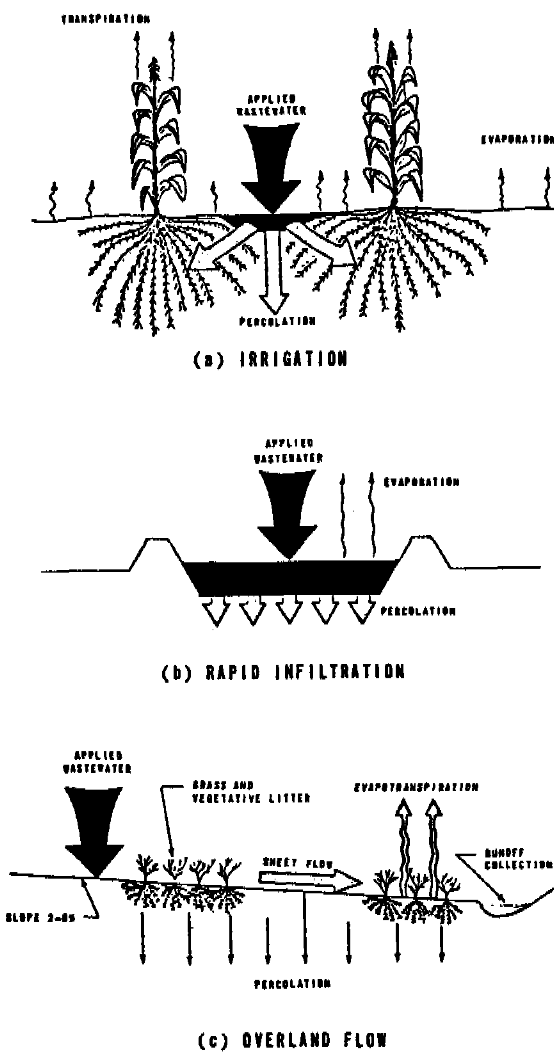

Figure 4. Methods of Land Treatment

LAND TREATMENT OPPORTUNITIES

Considering the diversity of terrain, land uses, and wastewater sources in Prince George's County, it became apparent that the many different opportunities for land treatment had to be reduced to a manageable set of options. After analysis, the three most promising options included: (1) major land treatment alternatives, (2) a wastewater reuse possibility involving silviculture, and (3) local land treatment alternatives.

Major Land Treatment Alternatives

Major land treatment alternatives would provide significant wastewater treatment capacity for the county (greater than 43.8 L/s), when used in conjunction with major wastewater treatment plants in the area.

The 12 sites shown in Figure 5 were identified as potentially suitable for these alternatives. They have the potential for utilizing the full range of land treatment methods including irrigation, overland flow, and rapid infiltration. The preliminary assessment of treatment capacity associated with each of these proposed sites demonstrated the availability of sufficient capacity to meet all of the county's needs for the next 20 years. The sites were identified on the basis of county plans for development over the next 10 years, existing land ownership (parcelage), and physical characteristics as determined from topography and SCS soil survey maps.

The options for utilization of these sites were divided into five alternatives (summarized in Table 1), on the basis of the source of wastewater and the location of sites involved. In addition to providing high level wastewater treatment, the groups included opportunities for the following:

1. Provision of additional treatment capacity and improved operational performance associated with existing facilities.
2. Productive secondary use of gravel extraction areas
3. Establishment of intraurban open space
4. Provision of process and cooling water from storage lagoon or recovered water for planned industrial uses
5. Establishment of a compatible land use with an adjacent sensitive environmental area
6. Preservation of open space in the Central County wedge area
7. Preservation of prime agricultural land
8. Protection and enhancement of rural residential areas

All five groups of major alternatives contain viable opportunities for land treatment that warrant further study. A definitive conclusion, i.e., recommendation of specific alternatives, was not considered possible or desirable at this level of planning. Instead, the options were compared in general terms on the basis of the following:

- Wastewater treatment effectiveness
- Costs associated with alternatives
- Impacts

Many of the major alternatives appear promising from the standpoint of wastewater treatment effectiveness, particularly as levels of treatment requirements become more stringent. All major discharges in the county will soon be subject to nutrient limitations of some sort. Phosphorus limitations are imminent and could be met easily with the slow rate process or rapid infiltration. There is also a possibility that a limit could also be placed on total nitrogen concentrations, in which case either slow rate systems or overland flow in combination with rapid infiltration would be applicable.

A preliminary investigation of the costs of the major alternative groups showed that land treatment, in most cases, could be cost effective when compared with similar levels of advanced wastewater treatment. Estimates were based on one or two typical designs for each group, and were intended to demonstrate the magnitude of costs that could be expected.

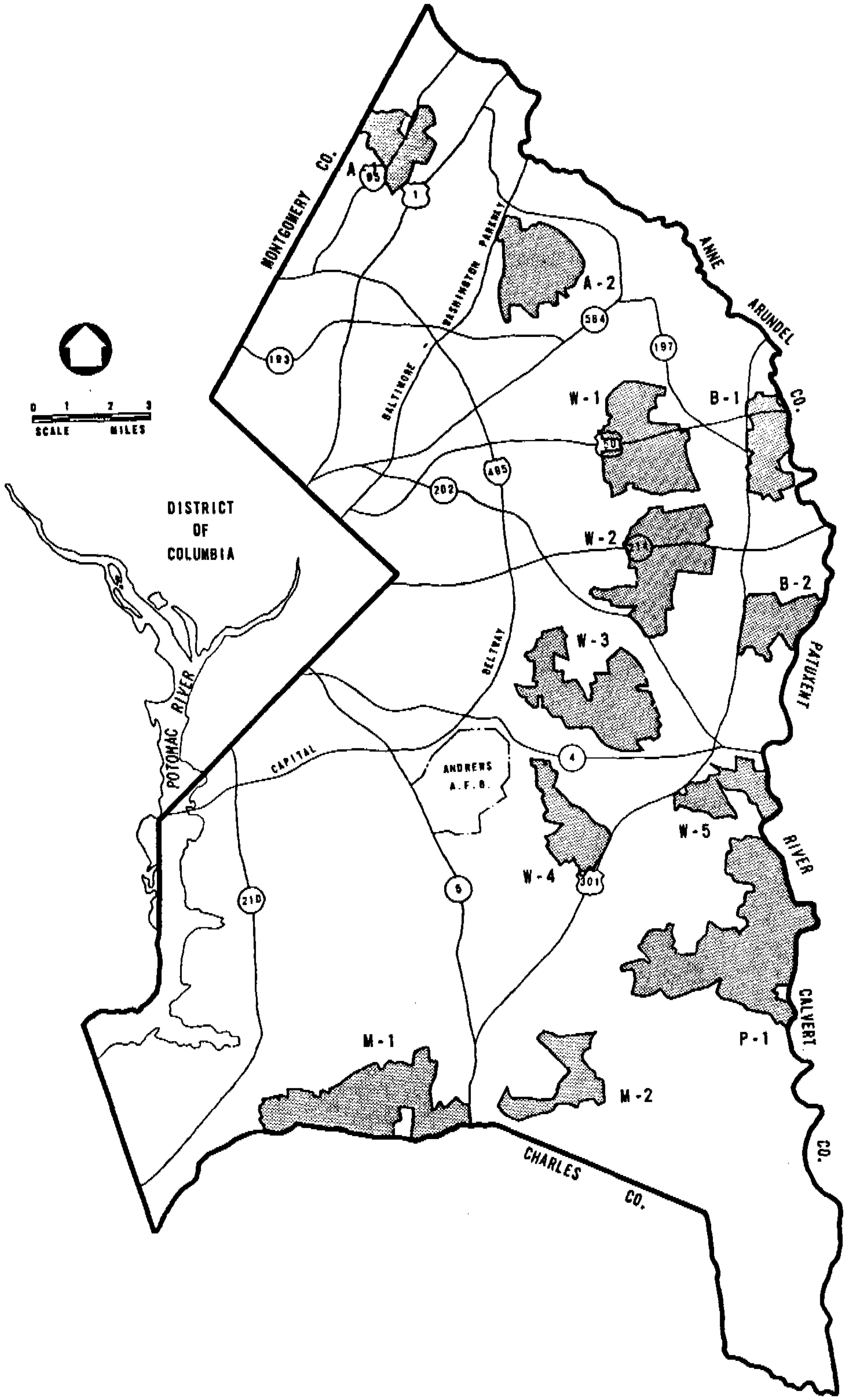

Figure 5. Potential Sites for Major Land Treatment Alternatives in Prince George's County

Table 1. Major Land Treatment Alternative Groups in Prince George's County

Alternative group	Description	Preapplication treatment	Land treatment process	Theoretical range of maximum capacity, m^3/s^a	Combined site areas, ha^b	Disposition of effluent
1	Treatment of flows from either Upper Anacostia Basin and/or Parkway STP at Anacostia sites (A-1, A-2)	Aerated lagoons at site, or Parkway STP	Rapid infiltration, overland flow, irrigation, or combinations	0.88-2.2	2,138	Primarily to surface streams of Anacostia Basin. Some aquifer recharge.
2	Treatment of Upper Western Branch or Bowie flows at Basin sites (W-1, W-2, W-3, W-4, B-1, B-2)	Aerated lagoons at site, or Bowie STP	Primarily irrigation	1.9	5,941	Primarily to surface streams of Western Branch and Patuxent basins. Some aquifer recharge.
3	Advanced waste treatment of secondary effluent from Western Branch STP at Sites W-5, P-1, W-3, or B-2	Western Branch STP	Rapid infiltration, irrigation	1.3-2.1	6,111	Primarily to surface streams of Western Branch and Patuxent basins.
4	Treatment of flows from Upper Anacostia Basin via pumpover to Upper Western Branch (Sites W-1, W-2, W-3, B-1, B-2)	Aerated lagoons in Upper Western Branch	Primarily irrigation	1.7	6,881	Primarily to surface streams of Western Branch and Patuxent basins. Some aquifer recharge.
5	Treatment of flows from Mattawoman Basin and/or Piscataway STP at Mattawoman sites (M-1, M-2)	Aerated lagoons at site or Piscataway STP	Primarily overland flow	1.7-2.5	2,337	To silvicultural/ agricultural reuse project or Piscataway outfall.

a. Assumes full utilization of each site within alternative groups.

b. Probably required land area includes application area plus 25% allowance for buffer zones and other areas but does not include an allowance for preapplication treatment and storage.

Impacts of the alternatives considered were divided into economic, social, environmental, public health, and land use planning. Evaluation of the impacts of land treatment alternatives led to a mixture or a combination of positive and negative effects.

Wastewater Reuse Opportunity

An opportunity for a large-scale wastewater reuse project was identified as an adjunct to the overland flow alternative in the southern part of the county (Alternative Group No. 5). Such a system would consist essentially of distribution networks for utilization of the treated wastewater in silviculture and agricultural activities in that part of the county. Not only would the implementation of this option provide an additional level of treatment, but it would also provide added benefits in the form of reduction of costs associated with utilization of artificial fertilizers and provision of water for crop management in this area. Further, or finally, it would provide a viable alternative to further expansion and upgrading of an existing wastewater treatment plant.

The major emphasis of this prospect would be on silviculture. Intensive management and irrigation of forest resources would substantially accelerate tree growth, broaden the existing agricultural base, and improve economic conditions for the residents of the area and of the county in general. The market potential for forest products in this area was investigated and was found to be very favorable. Irrigation of woodlands would be an efficient use of forest resources that are currently underutilized within the county.

Another essential feature of the project would be benefits accruing to local residents. In order to avoid relocation of residents by condemnation or other fee acquisition procedures, methods which would maintain control without relocation were considered. One possibility examined was the purchase of timber rights rather than the fee interest in the land itself. Such a system could be operated by a forest management company or a consortium of land owners under contractual arrangements with the wastewater authority. By the sale of the timber rights, rather than the land itself, the local residents would profit from use and enrichment of parts of their land which, at this time, offer no financial return. This incentive, combined with the fact that residents would not lose title to the land, should make the concept more readily acceptable. The extent of irrigation of agricultural land will be dependent on the needs and desires of the local farmers, as well as local climatic conditions. The system would be designed so that the wastewater would be accessible to farmers who wish to benefit from it.

Overland flow was considered to be the most favorable method of preapplication treatment for several reasons:

- Most trees do not require additions of substantial nitrogen because they recycle a major portion of their nitrogen back to the soil via leaf litter. The overland flow technique, by reducing total nitrogen concentrations in the applied wastewater, would protect the groundwater from nitrate contamination.
- It is impractical and undesirable to limit public contact in an irrigation area of this size (4050 ha or more). Disinfection prior to spraying must be adequate to allow for public use of this area. This technique would sufficiently reduce BOD and suspended solids to levels necessary for effective disinfection without excessive utilization of chemicals.

Local Land Treatment Opportunities

The possibility of small land treatment systems (generally with capacities much less than 43.8 L/s) was considered separately from the standpoint of local opportunities. These systems would not be considered as significant alternatives in relation to the major treatment plants in the county. Instead, they could serve local communities on an annual or seasonal basis and relate specifically to that community's needs and goals. The possibilities for local treatment systems exist wherever there is available land near a present or potential source of wastewater flow. Opportunities for these systems are generally more apparent in isolated areas than on a large-scale basis. Consequently, this analysis did not focus on the suitability of small systems for the county as a whole, rather it concentrated on several specific and more apparent alternatives.

With regard to planning considerations, small land treatment systems can be utilized to accomplish two major objectives:

- Provide needed sewage treatment capacity to remote communities
- Obtain secondary benefits associated with reutilization of water and nutrients in the treated wastewater

The first objective relates mainly to the outlying areas in the county where connections with major sewer systems are either uneconomical or undesirable. In such a situation, a local land treatment system can provide capacity for planned small-scale rural development. It could also be used to provide relief to small communities experiencing septic tank failures.

Aside from treating wastewater, small systems can be used to achieve numerous secondary benefits. An intraurban open space can be preserved within a developing community, providing such things as a small, local farming operation, an outdoor recreation area, or wooded parkland.

IMPLEMENTATION PROGRAM

The county staff responsible for the study recognized from the outset of the project that there could be many serious obstacles to implementing a land treatment alternative, no matter how technically sound any of the alternatives might be. Consequently, considerable time and effort was devoted by county staff, elected officials, and citizens' groups to address potential obstacles that might prevent implementation of any of the recommended alternatives.

As mentioned earlier, the approach taken during the course of the study was one based on an open planning process. Preliminary contact was made with many segments of the community that could eventually be involved or affected in some way if a land treatment system were implemented. This contact with both public and private interest groups was broadened extensively during an open-ended planning period following publication of the report. The report was used as a basis for discussion so that land-based treatment schemes could acquire a high potential for implementation.

The program for public participation in the second phase of the project (after publication of the report) was initiated with a formal presentation by the county government, Washington Suburban Sanitary Commission, and the consultants. Representatives of nearly every interest group in the county that might possibly be affected by land treatment were invited. Detailed presentations and meetings were also scheduled for any group that desired them. Groups that became involved included local civic associations and local agricultural organizations, including the Agricultural Extension Service of the University of Maryland and the Soil Conservation Service.

The overall reaction of these groups to the approach was quite good. Most parties were able to agree with the utilization of land treatment systems. However, there were concerns over certain aspects of such systems. These included concerns related to public health impacts and the aesthetics of such systems. Information was presented to address these concerns and most of the fears of risks of land treatment were reduced. However, it was quite obvious from the results of the dialogue with the public that an extensive public information program would be required prior to implementation of any alternative involving land treatment. Another concern not adequately addressed in the planning process was the acquisition of private lands by the public agencies (particularly agricultural land). On the basis of several meeting held with local agricultural groups, the importance of considering alternative methods of land control such as leasing and/or contractural arrangements with farmer-operators was quite apparent. This aspect of the development of land treatment options clearly requires extensive elaboration so that land owners and residents in the vicinity of the proposed project sites would fully understand the benefits from implementation of a land treatment option.

CONCLUSION

The prospects for implementing an effective land treatment system(s) in Prince George's County appear reasonably good. A range of technically feasible alternatives has been identified, and a framework has been established for future planning, acceptable to those concerned with the study area. This contrasts with several previous facilities planning efforts in the county in which land treatment was found not to be feasible. For the most part, these studies were from a more limiting point of view in which the positive aspects of land treatment could not be realized. Experience, in this case, indicates that by using a more comprehensive approach to planning it is possible to deal with the multifaceted nature of land treatment in ways which would not otherwise be apparent.

PUBLIC, INSTITUTIONAL AND TECHNICAL ACCEPTABILITY

AN INTEGRATED RESOURCES MANAGEMENT APPROACH TO LAND TREATMENT DESIGN

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ABSTRACT

A comprehensive resources management plan was formulated for a development in western metropolitan Chicago. The plan links land treatment with other resource management elements to achieve a highly integrated economic development plan. The plan is acceptable from both technical and political perspectives. This development demonstrates the opportunity to implement land treatment within a high density urban region. The management of wastewater within an urban region provides an opportunity to achieve concurrently, water supply, flood control, recreation, open space, and energy management benefits.

INTRODUCTION

A comprehensive resources management plan was formulated for a development that links land treatment with water supply, flood control, recreation, open space, and energy management. The plan integrates the proposed development with the physical environment of the site.

The discussion of the plan is presented in three parts. First, the physical environment is assessed. Second, resource management opportunities are identified. Finally, the elements of the comprehensive resources management plan are sketched out.

The final part of the paper presents the conclusions which can be drawn from the planning effort.

THE PHYSICAL ENVIRONMENT

The development comprises 112 hectares (277 acres) located in a high density urban region in western metropolitan Chicago, 6.4 kilometers (4 miles) west of O' Hare International Airport. Residential development surrounds three sides of the site. On the western side is a large industrial complex.

The land and water resources of the development provide the setting for the development. The natural resource features of the site can be discussed from the following perspectives: geology, topography, soils, climate, groundwater, and surface water. Each of these is discussed briefly in this section of the paper.

Geology

The surface geology of northeastern Illinois is attributed to glacial action. The advance and recession of huge ice masses across the region formed the topography and laid down formations which store and transmit groundwater.

There are three significant geological features of the site. The first is the glacial till that extends approximately 21 meters (70 feet) below the subsoil. This material is unconsolidated clay and silt with sand and gravel lenses only a few centimeters thick. The second is the Silurian dolomite bedrock 24 meters (80 feet) below the land surface. The bedrock is fissured and fractured, especially the top layers. The third is the Maquoketa shale below the bedrock. These shales are impervious and tend to restrict the downward movement of the groundwater that has percolated through the glacial till.

Topography

The land surface of the site

slopes gently toward swales and depressions near the middle. This topography was shaped by the glacial activity that occured on the site. Elevations range from 209 meters (685 feet) to 218 meters (715 feet) above sea level with maximum relief on the site being 9 meters (30 feet). Slopes are generally less than 3 percent. These slopes increase to 7 percent near the depressions and swales.

Soils

The soils on the site generally have silt and clay loam textures. Traces of oxidized iron are common in soil profiles. The soils can be divided into two groups according to the organic matter content and structure. The first group includes soils on the slopes. The second group includes those in swales and depressions.

The soils on the slopes have surface layers 8 to 46 centimeters thick. They tend to have medium and coarse subangular blocky structures and contain variable amounts of organic matter ranging from 6.2 to 8.9 percent. Surface layer permeabilities are moderate. Cation exchange capacities range between 12.6 and 25.9 meq/100 grams of soil. Subsoils contain more clay and have slowly moderate to moderate permeabilities. The swales and depressions contain deposits of organic material that extend to depths of 2.0 to 3.3 meters (8 to 13 feet). As a result, the organic matter and cation exchange capacities of these soils are slightly higher than the soils on the slopes. Organic matter comprises 10 percent of the soils in the swales and depressions, cation exchange capacities range between 25.8 and 28.9 meq/100 grams of soil.

Climate

The climatic regime of northeastern Illinois is classified as "humid continental type" with characteristically hot summers, cold winters, and rain distributed throughout the year. Precipitation is slightly more than 84 centimeters (33 inches). Slightly higher amounts of precipitation occur between April and September due to thunderstorms.

Groundwater

Underlying most of northeastern Illinois are shallow and deep aquifers. The shallow aquifers include sand and gravel aquifers and the Silurian dolomite aquifer. The only shallow aquifer underlying the site is the Silurian dolomite. The Cambrian-Ordovician aquifer is a deep aquifer that is confined by overlying Maquoketa shales and underlying Eau Claire shales. Both of these aquifers are used as public water supplies.

Surface Hydrology

The development is located in the Salt Creek drainage basin. Two intermittent watercourses enter the site which drain 267 hectares (660 acres). Formerly a third watercourse flowed into the site, but construction of a large highway storm sewer diverted this surface water resource from the site.

The floodplain of record on the site was delineated by the U.S. Geological Survey in a hydrological atlas (HA-143). Also the special flood hazard area or the 100-year floodplain was delineated on maps prepared by the Federal Insurance Administration. Both of these maps indicate that 18 hectares (45 acres) of the site is subject to innundation either from the flood of record or the 100-year flood.

RESOURCE MANAGEMENT OPPORTUNITIES

Developing a large urban complex presents several resource management problems. The development needed to attain water supplies, an increase in flooding problems had to be prevented, good surface water quality was necessary, open space recreation and the benefits of energy conservation were also desired. The following presents these resource management opportunties.

Water Supplies

The development needed a potable water supply and water for landscape irrigation and fire protection. Potable water will be consumed at the initial rate of $955m^3$/day (0.25 mgd). In this area of northeastern Illinois, these needs are secured from groundwater supplies.

The available groundwater in the area of the development is declining because groundwater is being extracted from storage areas faster than it is being replaced. The practical sustained yield of the shallow aquifer in this region was estimated by the Illinois Water Survey to be $16,700m^3$/day (4.4 mgd). The actual pumpage in 1974 was $17,000m^3$/day (4.5 mgd). The situation is much more critical in the deep aquifer, where the practical sustained yield is estimated to be $5,300m^3$/day (1.4 mgd) and the actual pumpage was $17,800m^3$/day (4.7 mgd).[1] There are regional plans that have considered importing Lake Michigan water 40 kilometers (25 miles) to this area because of the great concern over declining groundwater supplies.

Flooding Problems

Increased runoff and floodplain encroachment has caused severe flood problems in the Salt Creek drainage basin. To prevent an increase in these flooding problems, the development is required by local ordinance to store runoff from the 100-year storm. The development will have to provide 11.1 hectare-meters (90 acre-feet) of stormwater storage capacity to prevent increased flooding problems downstream.

Water Quality

The development will have to treat their wastewater including non-point sources of wastewater. Both must be economically treated to prevent degradation of surface water areas.

Most developers in this area discharge wastewater to municipal sewage treatment plants, but flows from the development cannot be treated by existing municipal wastewater treatment facilities because they are already at capacity. Furthermore, the development needs a water supply for landscape irrigation and fire protection. The initial point source wastewater flow from the development is estimated to be $955m^3$/day (0.25 mgd).

Open Space Recreation

One of the site development objectives is to provide open space recreation opportunities. Urban sprawl has rapidly reduced the available open space and wildlife habitat in the area. From 1970 to 1976, 30 percent of the remaining open space in Addison Township, DuPage County, Illinois was developed leaving only 567 hectares (6 percent of the Township) undeveloped.

Energy Conservation

The importation of energy to the development must be minimal. The operation costs must reflect wise use of energy and energy conservation.

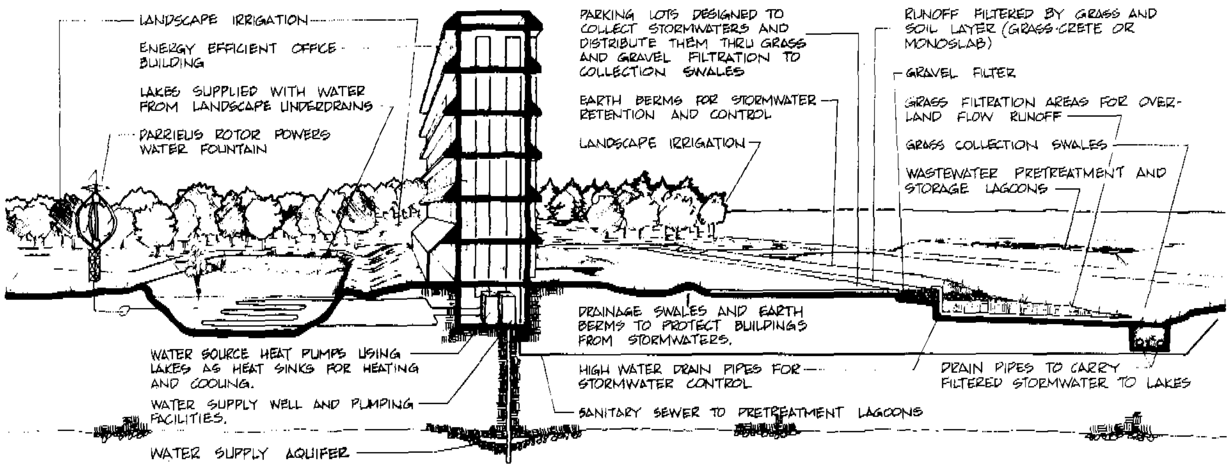
THE COMPREHENSIVE RESOURCES MANAGEMENT PLAN

The resources of the development are integrated with a comprehensive resources management plan. The plan features multiple use of land and water resources to provide an economic self-sustained urban module. Figure 1 shows a cross-section of the development. The water supply, stormwater, wastewater and energy management elements are presented in the following.

Water Supply Management

The potable water supply of the development will come from the shallow dolomite aquifer. However, once used, the wastewater will not be discharged to a river. It will be treated and used as a water supply for landscape irrigation. After the water has filtered through the soil, underdrains will collect

FIGURE 1
SCHEMATIC PROFILE OF THE DEVELOPMENT



and convey it to five interconnected lakes where it will be available for fire protection, enhancement of site aesthetics, and recreation. During this time, areas of the site will be recharging the underlying shallow dolomite aquifer. This will occur continuously through 11 hectares (27 acres) of lake bottoms and intermittently through the 34 irrigated hectares (85 acres).

Water conservation and future site activity may reduce the demand for water at the development. Studies have demonstrated that total water consumption can be reduced up to 27 percent with simple inexpensive water conserving fixtures.[2] Implementation of water conservation could reduce the ultimate water demand from $1,553 m^3$/day (0.41 mgd) to $1,023 m^3$/day (0.30 mgd). This would significantly reduce the size of wastewater management facilities needed for wastewater treatment.

Stormwater Management

Stormwater runoff is managed at the development by removing nonpoint source pollutants before it is conveyed to the lakes for storage. Land treatment will be used to filter runoff from impervious areas before it is collected in grass swales. First, the runoff will pass through a layer of soil and grass, then through gravel and again through grass. The parking lot cross-section in Figure 1 shows the placement of materials used for designing these treatment facilities. Porous concrete structures--grass pavers or monoslabs--will be installed along parking lot fringe areas. Water can slowly flow through these structures and grass can be grown on the soil placed between them. Underlying these structures is a layer of gravel that extends under a stormwater storage berm. Runoff will flow through the porous structures and through the gravel layer to an overland flow grass filtration area. The stormwater is then collected at the bottom of the grass filtration area by grass swales and underdrains before it is conveyed to the lakes. To prevent topping of the storage berm during major storms, relief standpipes will be placed at intervals along the pavement edge. The cost of these stormwater management facilities including the grass swale will be approximately \$55.75 per linear meter (\$17.00/linear foot). This cost is comparable to conventional storm sewers which do not provide treatment of non-point source pollutants.

Wastewater Management

The wastewater management system for the development will be a self-contained facility serving only the buildings on site. This system links wastewater management with water supply and flood control facilities. Open space recreation and energy conservation are also linked to the wastewater management system. Land treatment will keep water cycling through the development adding aesthetic, recreation and energy management benefits to the site. The system will have an initial design capacity of $955m^3$/day (0.25 mgd). The following presents the components of the system and then the design and cost details.

System Components

The wastewater management system consists of collection, pretreatment, storage, and irrigation components. Figure 2 shows a flow diagram of the system. Pretreatment consists of aeration in a 3 celled lagoon and filtration by intermittent sand filters. Pretreatment and storage will be combined for the initial design flow. The aerated lagoon will provide $133,600m^3$ (35.28 mg) of storage capacity, enough to store winter flows. Before the wastewater is applied to the land as a landscape irrigant, it will be chlorinated. After irrigation, the renovated wastewater is collected by underdrains and stored in the lakes for reuse. The irrigated areas will provide open space and recreation opportunities. Trailways will meander through irrigated areas providing access to surface water and landscaped areas.

This wastewater management system requires only an Illinois Environmental Protection Agency "construct, own and operate" permit to produce a 10 mg/l biochemical oxygen demand (BOD), 12 mg/l suspended solids (SS) effluent. A National Pollution Discharge Elimination System (NPDES) permit is not required, because the system is considered a zero discharge operation. Therefore, the system will be exempt from changes in water quality and effluent standards.

Design and Cost Details

Wastewater from the development will be generated from office, hotel, and commercial activities. The influent flow rates and loadings based on the initial development phase of the development are presented in Table 1.

Estimated wastewater flows from the fully developed site could include an additional $610m^3$/day (0.161 mgd) based on another $209,000m^2$ (2,250,000 sq. ft.) of office space. It this would occur, the full project average daily design flow would be $1,565m^3$/day (0.413 mgd).

A summary of the preliminary design and costs for treatment of the initial flow is shown in Table 2. The capital cost of the system is estimated to be $650,000. The annual operation and maintenance costs will be approximately $28,000. These costs do not include pumping, irrigation, and underdrains for landscaping, which can be eliminated because landscape irrigation facilities would be installed without a wastewater management system.

The entire pretreatment process of the initial design will require 2.8 hectares (7 acres) of land. Another 1.6 hectares (4 acres) will be required for storage of flows from the second phase of development.

The wastewater will be pretreated with aerated lagoons. The pretreatment design will consist of three treatment cells that will reduce BOD inflows by 75 percent. Provisions will be made so that each cell can be by-passed for maintenance purposes.

Intermittent sand beds will filter the lagoon effluent before chlorination and landscape irrigation. The beds will be flooded with 7.6 to 10.2 centimeters (3 to 4 inches) of effluent and allowed to drain before the next

Figure 2
FLOW DIAGRAM OF THE WASTEWATER MANAGEMENT SYSTEM

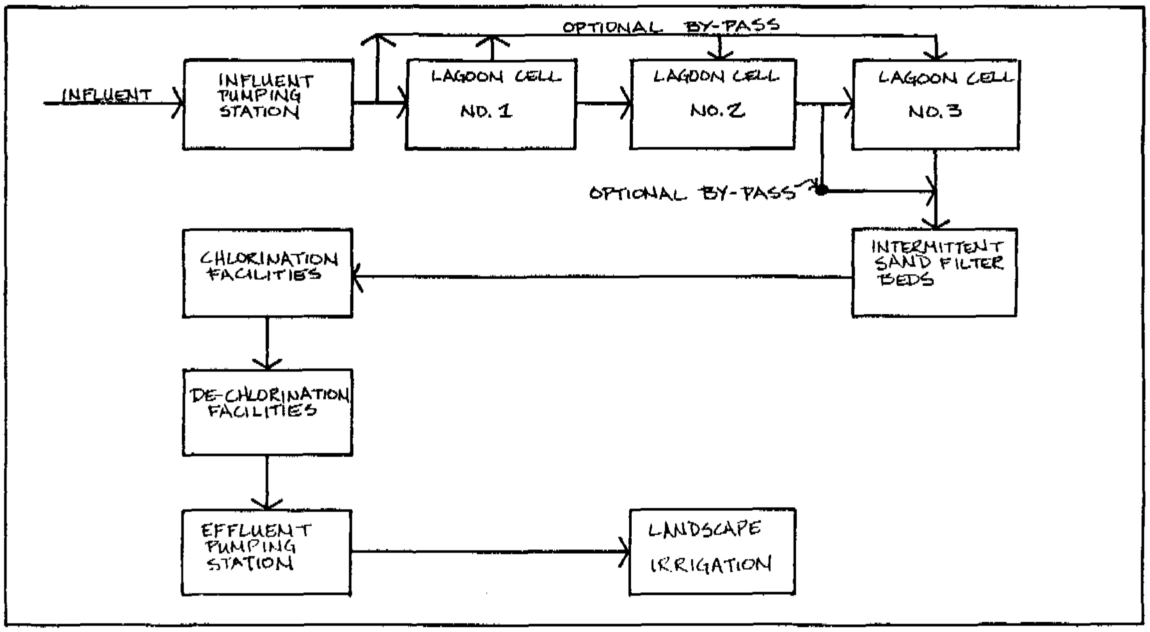


TABLE 1
WASTEWATER FLOW RATES AND SYSTEM LOADING

Parameter	Site Activity: office	hotel	retail/commercial	Total
Floor space (m^2)	209,100	27,900	13,000	250,000
Occupants	9,000	750		1,650
Mean weekday flow (m^3/day)	852	284	61	1,197
Mean weekday loading (kg/day)				
BOD	176	58	12	246
SS	204	68	15	287
NH_3-N	25	9	2	36
Total mean daily design flow (m^3/day)				955
Total mean daily design loading (kg/day)				
BOD				196
SS				228
NH_3-N				29

TABLE 2

SUMMARY OF DESIGN AND COSTS FOR WASTEWATER MANAGEMENT SYSTEM

COMPONENTS	PRELIMINARY DESIGN	CAPITAL	O & M
Raw Wastewater Pumping 2.5 MGD Installed Capacity		\$ 130,000	\$ 2,500
In-ground, factory built station	Peak flow capacity, 1.90 MGD		
Aerated Lagoons - Construction 82,000 yd^3 excavation x \$1.60/yd^3			
Cell No. 1	BOD_5 loading = 431 lbs/day = 205 mg/l Volume required = 858,000 ft^3 Additional winter storage required = 2,650,000 ft^3 Average detention time (summer operation) = 27 days Liquid depth (summer operation) = 15 feet Liquid depth - maximum = 35 feet BOD_5 effluent = 107 lbs/day = 51 mg/l		
Cell No. 2	BOD_5 loading = 107 lbs/day = 13 mg/l Volume required = 356,670 ft^3 Additional winter storage required = 1,392,000 ft^3 Average detention time (dummer operation) = 11 days Liquid depth (summer operation) = 15 feet Liquid depth - maximum = 35 feet BOD_5 effluent = 27 lbs/day = 13 mg/l		
Cell No. 3	BOD_5 loading = 27 lbs/day = 13 mg/l Volume required = 0.25 x x 356,670 ft^3 = 89,170 ft^3 Additional winter storage required = 668,900 ft^3 Average detention time (summer operation) = 3 days Liquid depth (summer operation) = 15 feet Liquid depth (maximum) = 35 feet	\$ 131,200	\$ 17,00
Lagoon Appurtenances and Piping		\$ 30,000	
Aeration - Tubes, Lines Blowers	750 CFM capacity	\$ 65,000	
Effluent Pumping to Sand Beds	0.55 MGD installed capacity	\$ 47,000	\$ 1,300
Recirculation Pumping	0.23 MGD installed capacity	\$ 39,000	\$ 1,500
Sand Beds: Construction 2740 yd^3 Sand @ \$4/yd^3 1370 yd^3 Gravel @ \$4/yd^3	Average flow rate = 0.44 MGD Average hydraulic design loading rate = 10 gallons/ft^2/day Total surface area = 44,000 ft^2 = 1.01 acres Number of beds = 7 Underdrainage system at 10-foot intervals discharges to chlorination facilities	\$ 16,500	\$ 2,800
Distribution and Underdrains for Bed	Peak flow capacity = 0.44 MGD	\$ 32,000	
Chlorine Feed, Contact Tanks, Stripping Tanks	Maximum dosage rate = 5 mg/l Contact time = 15 minutes Dechlorination time = 60 minutes	\$ 30,000	\$ 2,400
Pumping, Irrigation and Underdrains for Landscaping	Irrigation areas available = 85 acres Average flow rate = 0.44 MGD Irrigation period = 7 months Hydraulic loading rate = 1.33 inches/week Type of irrigation equipment: Permanent set sprinkler systems	\$ 215,000	\$ 12,800
	Sub Total	\$ 735,700	\$ 40,800
	Engineering and Contingencies - 25 percent	\$ 184,000	
	Total excluding Landscaping Comp.	\$ 920,000	
	TOTAL	**\$ 650,000**	**\$ 28,000**

flooding. Flows from the sand filters will be pumped to landscape irrigation facilities. An underdrainage system will assure drainage of irrigated areas. The collected water will be conveyed to the lakes to augment lake inflows.

Energy Management

Energy will be used efficiently at the development. The lakes will be used to manage building energy budgets on the site. The lakes will be used for evaporative cooling during the summer when heat is removed from the buildings. The lakes are also being considered as a heat source to augment energy supply of solar heated buildings.

Use of wastewater for landscape irrigation rather than groundwater will also conserve energy. If the water used for irrigation were pumped from groundwater supplies underlying the site 262,500 KW hours of energy would be required annually, valued at $13,000. Implementation of water conservation will also reduce the energy required for pumping and wastewater treatment.

CONCLUSIONS

This study demonstrates the opportunity to implement a land treatment system within an urban region. When properly planned, a range of benefits can be achieved. These include treated wastewater; including non-point sources, augmented water supplies, aquifer recharge, open space and recreation opportunities, preservation of wildlife habitat, flood control and energy conservation.

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SITE EVALUATION AND SELECTION

ECONOMIC EVALUATION AND MANAGEMENT OF LAND TREATMENT SYSTEMS

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ABSTRACT

Legislative and administrative mandates are focusing increased attention on land application of effluents in the United States. This presents an opportunity to shift emphasis from waste treatment and disposal to resource recovery and reuse.

The economic analysis of land treatment systems needs to address issues of both efficiency and equity. Efficiency is addressed through cost-effectiveness analysis. Equity is addressed through institutional analysis, where the influence of society's rules determine who pays and who benefits as a result of public policy choices.

Analytical frameworks are discussed for accounting for cost variations in important components of land treatment systems. A framework of options for acquisition and management of land is provided, and the implications are discussed.

Experiences from operational land treatment systems indicate that local factors as well as general frameworks influence the acquisition and management options selected.

INTRODUCTION

Water quality goals in the United States are articulated in state and federal laws. Most ambitious in terms of goals and potential expense are the 1972 Amendments to the Federal Water Pollution Control Act (PL 92-500) and the Clean Water Act of 1977 (PL 95-217). These laws contain significant provisions dealing with the control and treatment of wastewater from municipal and industrial sources. In particular they provide emphasis for the evaluation and use, when found cost-effective, of land application of municipal effluent.

Added emphasis has been given land treatment by EPA policy statements and by provisions in PL 95-217 providing special subsidies for innovative technologies applied to wastewater treatment problems (EPA, 1977; Federal Register, 1978). For example, innovative technology can be considered cost effective if it costs no more than 115 percent of the least costly alternative.

Interest in land treatment of effluents provides a unique opportunity to emphasize resource recovery and reuse in wastewater management rather than treatment and disposal. Numerous opportunities exist to reclaim water and nutrients for crop production and to recharge groundwater supplies. Under this philosophy, wastewater can be a valuable resource to be managed, rather than a nuisance for disposal.

Land treatment however, is no panacea. There are many instances where conditions are not condusive to the construction of land treatment systems. Careful case by case evaluation is required to identify those situations where land treatment is most viable. Such analysis requires a truly multidisciplinary approach. Technological, economic, social, and legal factors must be evaluated within a site specific context.

This paper focuses on economic and institutional issues encountered in the analysis of land application systems[1/] in the United States. Some factors encountered in an economic analysis of land treatment systems are addressed. It then identifies important institutional considerations for land acquisition and management, and concludes with examples from operating systems in the United States and around the world.

ECONOMIC EVALUATION

Care is needed when referring to an "economic" evaluation of a land treatment system. In the broadest sense, such an evaluation should quantify all costs and benefits associated with a system, both monetary and nonmonetary. Such a benefit-cost analysis is a formal procedure for evaluating alternative policies. Difficulties often arise in assigning dollar values to all costs and benefits. Benefits are often excluded from the evaluation, making the economic evaluation more correctly a cost evaluation.

When a policy objective is assumed, as in the case of wastewater treatment, one economic question is that of determining cost-effective methods for meeting the stated policy goal. Such cost-effective analysis identifies and quantifies the costs of alternatives to meet a given policy objective. Then the least costly method can be selected. Emphasis is primarily on monetary costs, with the benefits presumably reflected in the predetermined policy goal. In addition, only accounting costs are generally considered. The opportunity costs, the value of goods and services and opportunities foregone by the use of resources in a particular manner, are generally omitted from a cost-effectiveness analysis.

Cost-effective analysis applied to wastewater treatment problems thus seeks to identify the least costly way of meeting goals specified through standards for such pollutants as biological oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS), nitrogen (N) and phosphorous (P). The costs of land treatment systems are developed and compared with the costs of alternative treatment systems.

Cost Considerations

Cost-effective analysis of wastewater treatment alternatives needs a consistent data base including both the costs of component items and the resultant level of effluent treatment. It is important that cost comparisons between treatment technologies be based upon consistent underlying assumptions and data sets. An analytic framework is needed to explicitly account for all the factors influencing costs and identify the sensitivity of total costs to changes in the costs of the various components. Cost variables to be analyzed include land acquisition costs, land preparation costs, surface runoff control costs, subsurface drainage, irrigation system costs and relocation costs. Other variables to be evaluated include capital subsidies, crop selection, effluent recovery costs, design flows, transmission distance, reserve land requirements, discount rate and period, and input costs.

Some Results

Several simulation models and methodologies have been developed for evaluating land treatment systems (Corps of Engineers, 1976; Young, 1976; Pounds, Crites and Griffes, 1975). Cost effectiveness analysis data in this paper are based upon USDA research, particularly (Young, 1976).

Costs for land treatment of wastewater can be estimated using the Cost of Land Application Wastewater (CLAW) model discussed in Volume 2 of the Conference proceedings (Young, 1978b). This model consists of five basic steps: preapplication treatment, transmission, effluent storage, application system and recovery of renovated water.

Land treatment systems are most cost-effective for volumes up to ten MGD. At that level, most economies of size have been achieved (Young, 1976). Cost comparisons between two land treatment systems (solid-set irrigation and center pivot irrigation) and three advanced waste treatment systems are shown in table 1, along with some indicators of relative effluent quality. Based on the assumptions and representative systems evaluated, land treatment was found to be cost-effective for achieving tertiary wastewater treatment

1/ Discussions are limited to irrigation applications, but the results and ideas have applicability to overland flow and infiltration basins.

Table 1. Comparison of average total wastewater treatment costs for various facility types and sizes and effluent quality comparisons

Treatment technique	Facility size (MGD) 1/ 0.5	1.0	10.0	100.0
	----- Cents per 1,000 gallons -----			
Secondary Treatment				
Trickling filter	65.7	48.3	15.8	13.6
Activated sludge	70.0	42.4	19.1	13.6
Tertiary Treatment				
AWT1: Activated sludge followed by nitrification-denitrification	105.5	77.5	30.0	21.1
AWT2: Activated sludge followed by lime addition, filtration, and sludge recalcination	165.0	129.4	51.6	31.7
AWT3: AWT2 plus ion exchange	184.8	144.5	62.1	35.8
Aerated lagoon followed by solid-set irrigation	91.3	69.5	43.8	37.8
Aerated lagoon followed by center pivot irrigation	85.1	63.1	36.4	29.7

	Effluent quality parameters 2/ BOD	SS	Total N	P
	-------------- mg/ℓ --------------			
Secondary Treatment				
Trickling filter	40	50	30	8
Activated sludge	20	25	30	8
Tertiary Treatment				
AWT1: Activated sludge followed by nitrification-denitrification	15	16	3	8
AWT2: Activated sludge followed by lime addition, filtration, and sludge recalcination	5	5	30	0.5
AWT3: AWT2 plus ion exchange	5	5	3	0.5
Aerated lagoon followed by solid-set irrigation	1	1	3	0.1
Aerated lagoon followed by center pivot irrigation	1	1	3	0.1

1/ After (Young, 1978).

2/ Pounds, Crites, and Smith, 1975.

goals, especially for smaller communities (Young, 1978). However, it is important to remember that nutrient removal with land application depends a great deal upon the application rate, the crop grown, and specific site characteristics. The cost advantage of land treatment over advanced wastewater treatment (AWT) options reflected in table 1 is primarily due to the use of relatively inexpensive aerated lagoons for pretreatment prior to land application. However, for smaller communities it can also be cost-effective with other pretreatment techniques, such as activated sludge. Adjustments in costs for other pretreatment techniques can be made by adding the cost of a secondary treatment process such as a trickling filter and deducting the costs of aerated lagoons. Thus a community with an existing secondary treatment facility would not need to construct aerated lagoons for pretreatment.

Direct cost comparisons between a land treatment system in one location and a conventional AWT system in another location can be misleading. Local conditions influence cost, particularly for land treatment systems where standardization of cost items is less common than for AWT systems. In addition, effluent quality requirements lend variability to the analysis. Effluent quality from land application will vary depending upon local soil characteristics and the particular crop selected for irrigation. Communities may specify a level of treatment that can be obtained only with an advanced waste treatment system. For example, the treatment level specified by Lake Tahoe, California precludes land treatment systems.

Cost-effective analysis should consider combinations of systems in addition to single systems. For some communities, a system with a land treatment component for part of the effluent and an AWT component for the remainder may be most cost-effective and handle flow and seasonal variations more efficiently than a single system design.

INSTITUTIONAL ANALYSIS

An economic analysis of land treatment systems needs to investigate issues of equity as well as economic efficiency. Cost-effective analysis focuses on efficient use of resources. Equity issues should also be evaluated, particularly how the rules governing the distribution of costs and benefits influence the outcome of public policy decisions.

One of the factors influencing the cost-effectiveness of land treatment systems is land cost. The monetary cost of purchasing land can be large, but even more important, the social costs and resulting opposition from land owners can prevent development of land treatment systems. Land acquisition and management needs to be handled carefully, balancing the respective views and goals of both the community and the land owner.

One of the key factors distinguishing land treatment systems from other wastewater treatment systems is the requirement for land. While all systems require land for plant location, it is an integral part of the treatment process for land treatment systems. Also, type, location, and quantity of land is much more important for land treatment systems.

For the community, land is needed as an input for the primary goal of wastewater treatment. Land needed for treatment systems is also an important resource required by farmers for making a living. Thus a community needing land for treating municipal effluent needs to consider the farmer's perspective. Equity considerations require that landowners should not be penalized if society considers using their land for meeting its wastewater management goals.

Institutional analysis focuses on how land will be acquired and managed and the impact of methods selected on issues of equity, system management, treatment reliability, implementation, and acceptability. Institutions define the working rules of society and set the rules which govern the operation of physical and economic systems. Institutions are "collective action in control, liberation, and expansion of individual action" (Commons, 1934), and as "sets of ordered relationships among people which define their rights, exposure to rights of others, privileges and responsibilities" (Schmid, 1972). Institutions specify what is acceptable individual and group behavior, reflect political power, and imply a capacity for one segment of society to impose its will upon another. For this paper, institutions are the organizations, authorities and relationships by which land application systems are implemented.

Property rights are an important economic institution influencing land treatment systems as they establish the rules for land sales, lease, and use and govern contracts and conditions for the acquisition and use of land. They reflect people's perspectives and values and a distribution of political and economic power to influence and control individual behavior and group action.

Acquisition and Management Options

Property rights govern the transfer and use of land needed for several purposes including treatment lagoons, pretreatment facilities, conveyance pumping stations, and application sites. Options for acquiring land provide control over the resource itself. Management options obtain certain behavioral actions from both farmer and community in the use of land (table 2). A number of management options can be exercised in conjunction with acquisition options, particularly with fee simple acquisition (purchase) and contracts. Decision makers need to evaluate these options in light of site requirements, and impacts on costs, control, and public opinion.

Fee Simple Acquisition. Through fee simple acquisition, a community and farmer exchange the total "bundle of rights" ascribed to property, with the exception of those rights reserved by the State, such as taxation, eminent domain, and police powers. A community can obtain fee simple title through a normal market exchange between buyer and seller. If legal authority exists, a community may exercise the right of eminent domain and obtain the rights from a reluctant seller in exchange for just compensation.

Options for the management of land acquired in fee simple include purchase and manage, purchase and leaseback, and purchase and resale on condition. The wastewater authority makes the managerial and operational decisions with the purchase and manage option. With the purchase and leaseback option, most managerial and operational responsibilities are transferred to the lessee. The purchase and resale on condition option enables the authority to buy the required land and then resell it with conditions attached compatible with land treatment requirements.

Real Property Interest Other Than Fee. Easements are the primary example of ownership of only a part of the total "bundle of rights" vested in land, or real property interest other than fee. Easements have been used for a number of public purposes. They may be acquired in a number of ways, through donation, purchase, or condemnation.

Through the use of easements, a community can acquire limited rights to land, such as for passage for irrigation pipes, ditches, or other equipment. Easements can be used to obtain buffer strips around irrigation sites. The easement conditions can impose some constraints on a farmer's managerial decisions, but he retains primary farm management discretion. They can be used to compensate farmers, particularly if crop yields decline as a result of land treatment. A major problem with easements is that they often provide much less control at about the same cost as fee simple purchase.

Table 2. Alternatives for Acquisition and Management of Land for Waste Management Systems

Acquisition Options	Management Options
I. Fee Simple	1. Purchase and manage 2. Purchase and leaseback to present owner 3. Purchase and lease to third party 4. Purchase and resale
II. Less than fee simple	1. Easements
III. Contractual agreements --no real property	1. Waste management cooperative 2. Leases 3. Contracts

Contractual Agreements

Land owners can provide a community access to land without the transfer of ownership rights, through the use of individual contracts or through group contracts such as a wastewater cooperative.[2/] Contracts specify an agreement of actions to be taken or refrained from in exchange for a consideration. Most contracts for land application in the United States have been between a farmer and community wastewater authority, specifying agreements for applying effluent to private farms or to land owned by the city.

An alternative to two-party contracts is a cooperative venture where a number of farmers enter into a contract with a city to provide land for wastewater treatment. Such a cooperative approach has not been used in the United States, but was evaluated as a possibility for a large scale land treatment system (Christensen, 1975b).

The cooperative approach to the utilization of land waste treatment has been used successfully in Braunschweig, Germany (Tietjen, 1973; Schaerff, 1977). The Sewage Utilization Association of Braunschweig was organized in 1974 to expand the activities of a sewage farm operating in the area since the 1890's. The association is made up of the city of Braunschweig, population 325,000, 476 farmers, and 26 nearby communities. Approximately eight million gallons per day of raw sewage is applied to 10,400 acres of land. The total irrigation area consists of land in 12 communities which is divided into four districts of comparable size which are further divided into three rural districts and three government districts. Policy decisions are made by a committee of 20 farmers and four city representatives. The cost of the system is divided--farmers pay 25 percent and the city 75 percent. Many communities bordering the irrigation area have joined the association, paying annual charge for sewer and pumping station networks.

Implications for Farmers and Communities

Each acquisition and management option impacts differently on the respective goals of the farmer and community and needs to be considered in the evaluation of land treatment systems. Farmer's goals include income generation, wealth accumulation, firm growth, freedom of decision-making and the sense of belonging to a community (Christensen, 1975a). A community's goals include meeting water quality standards in an economical manner. Both parties are motivated by a combination of economic and other goals. Water and nutrients in effluent may be used to increase agricultural production, which in turn can increase farm income and reduce community treatment costs.

Some specific implications of acquisition and management alternatives for farmers and communities are:

Farmers.

1. The impact of land treatment on farmer's goals is the greatest when fee simple title is acquired by the wastewater authority. Easement acquisition and contractual arrangements have less impact as the farmer's activities are influenced only by the terms of the easement or contract.
2. With fee simple transfer of title, the impact on the farmer is influenced by the management option selected by the community. If the farmer can remain on the land under a tenancy or employment arrangement, the impact would be far less severe than if the farming operations were turned over to a third party and existing farmers relocated. A lease arrangement would afford a former owner the greatest stake in the operation, particularly if he shares in successes as well as in failures.
3. Contractual arrangements between the farmer and the community should be reviewed on a regular basis. The initial agreement should indicate agreement over the distribution of benefits and costs, but changes in the underlying factors would require a regular review of contractual terms.

Communities.

1. Fee simple acquisition gives a community complete ownership of the land, but at a high cost, particularly for large systems.
2. Fee simple acquisition enables a community to unilaterally pursue its primary objective of wastewater treatment and renovation. With other than

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2/ A wastewater cooperative is a collective management venture that is an alternative to a two-party contractual agreement, such as between individual farmers and a wastewater authority.

fee simple acquisition, the treatment goals of the community and the income goals of the farmers will require more land to simultaneously treat wastewater and maintain agricultural production.

3. Fee simple acquisition provides a community flexibility to use the lands for a variety of public purposes such as parks and open space.

4. Real property interest other than fee may be obtained through the use of easements. Title to the treatment site would be retained by the current owner and the community acquires only those property rights necessary to carry out the particular management practices and controls required by the land treatment system. Easements do not remove land from the local property tax base.

5. The potential problems associated with a community buying large tracts of land suggest that public ownership of treatment sites is more viable for smaller communities than for a large metropolitan area. Even when land remains in private ownership, the acreage required for large wastewater volumes suggests that the land treatment is most applicable for smaller communities or for treatments of only part of the total wastewater volume from a large metropolitan area.

Operational Experiences

A review of selected operating land treatment systems found fee simple acquisition and contracts the usual ways communities acquire land (table 3). No cases were found of easement use. Leases are the most frequently used management option. Leases exist to manage farming on city owned land, as in the case of Lubbock, Texas, and Mesa, Arizona, as well as to govern the sale of water for application to privately owned land, as in the cases of Tooele, Utah, and Dickinson, North Dakota.

No situation clearly emerges where one acquisition or management option is more likely to be used than another. Fee simple acquisition might be expected to be used more frequently by smaller communities due to smaller land needs and fewer problems associated with community opposition. Conversely, one might expect contracts and leases would be more likely to be used by larger communities. However for the communities shown in table 3, the larger sites are owned by the city, except for Braunschweig. While one of the smaller sites is owned by the city (Mesa), most small sites are owned by farmers who contract with the city for water.

The fact that no clear pattern emerges highlights the uniqueness of each individual land application system and the difficulty of generalizing from one location to another. While concepts and guidelines provide some general assistance, the particular mix of technical, economic, legal, regulatory, social, and political factors operating at each site have the most influence on management and acquisition option selection. For example, the Muskegon, Michigan land treatment systems developed out of the combination of special water quality needs, a certain mind-set of county officials towards establishing the system and handling public opposition, large areas of sparsely settled, relatively non-productive land, and research grants. The extent to which such combinations of factors would result in other large scale, publicly owned land treatment systems in the United States is an unanswered question.

CONCLUSIONS

Land application of municipal effluent is receiving considerable attention in the United States due to legislative mandates and an increasing recognition that the concept of resource reuse and recovery has advantages over that of waste treatment and disposal.

Evaluation of land treatment systems needs to consider issues of both economic efficiency and equity. Cost-effective analysis seeks to insure efficient resource use in wastewater treatment. Cost-effectiveness analysis needs an analytical framework to identify the sensitivity of total costs to changes in costs of the key components. These include the costs of land, irrigation systems, transmission, labor, utilities, maintenance, and the availability of subsidies. Land application is a very cost-effective means of providing treatment, especially for facility sizes up to 10 million gallons per day. When comparing land treatment systems with alternatives, it is important that cost comparisons be consistent with the level of treatment provided.

Institutional analysis provides insight to the equity issues surrounding land application systems. The rules for distributing the costs and benefits of a system determine the equity of public policy choices. This is particularly

Table 3. Applications of options for land acquisition and management

Location	Type of Waste	Site Size (acres)	Acquisition Option	Management Option
Muskegon County, Michigan	Wastewater and sludge	10,600	Fee simple title	Managed by county
Braunschweig, Germany	Wastewater and sludge	10,400	Contract	Wastewater cooperative
Bakersfield, California	Wastewater	2,400	Fee simple	Leaseback to farmer; cash rent
Lubbock, Texas	Wastewater and sludge	4,000	Fee simple and contract	Leaseback of city owned land to single farmer who uses water on city owned land, his own land, and sells water to other farmers and an electric utility.
San Angelo, Texas	Wastewater	750	Fee simple	Managed by municipal employees
Chicago, Illinois	Sludge	15,000	Fee simple	Managed by Metropolitan Sanitary District and contracts let for farm operations.
Dickinson, North Dakota	Wastewater	250	Contract	Cash lease for water sale to farmer
Tooele, Utah	Wastewater	550	Contract	Cash lease for water sale to farmer
Roswell, New Mexico	Wastewater	200	Contract	Cash lease for water sale to farmer
Mesa, Arizona	Wastewater	160	Fee simple	Leaseback for cash rent
Camarillo, California	Wastewater	475	Contract	Landowner provides land in exchange for water and leases land to a third party

important when deciding how land should be acquired and managed for land application systems.

Alternatives exist for land acquisition and management, each with different implications for the community and farmer. These need to be considered in planning future land treatment systems.

A review of land treatment systems currently operating in the United States found considerable variation in both the acquisition and management options used. No unique sets of conditions were found where certain options are used exclusively or more likely to be experienced than another. Site specific technical data, economic issues, political and social environment, and the legal and regulatory environment combine to influence the selection of a particular acquisition and operating option.

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SITE EVALUATION AND SELECTION

THE USE OF REMOTE SENSING TECHNIQUES AND OTHER INFORMATION SOURCES IN REGIONAL SITE SELECTION OF POTENTIAL LAND TREATMENT AREAS

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ABSTRACT

Landsat, Skylab S190A Multispectral Photographic Camera, and Skylab S190B Earth Terrain Camera satellite data products were used to prepare land use maps for regional site selection of potential land treatment areas. The satellite data products were photographically enlarged to scales of 1:500,000 and 1:250,000 to facilitate the land use mapping. Interpretation of tonal and textural characteristics on the photography corresponded to vegetation, urban and agricultural land use categories. Color and color infrared transparencies augmented the land use mapping, which was accomplished on black and white photographic prints.

The Landsat and Skylab S190A photographic data products were useful for rapid regional land use surveys at the level I classification as described in the U.S. Geological Survey national land use and land cover classification system. The Landsat land use map took two hours to prepare and the Skylab S190A map required four hours. A Landsat image covers an area of 34,225 km^2 (13,225 mi^2) with a ground resolution of 70 m (230 ft), whereas a Skylab S190A photograph covers 26,569 km^2 (10,201 mi^2) with a resolution of 25 m (82 ft). The S190A land use map was more detailed than the Landsat map due primarily to the increased resolution and scale.

The resolution of the Skylab S190B photography was sufficient to prepare a detailed land use map at the level I and II classifications. Although the mapping time increased to eight hours, the S190B map was considerably more detailed. An S190B photograph covers an area of 11,881 km^2 (4,624 mi^2) with a resolution of 12.5 m (41 ft).

The resolution (5 m [16 ft]) and scale (1:120,000 or 1:130,000) of NASA high-altitude aircraft photography provided for detailed level I, II and III land use mapping. The NASA high-altitude aircraft photography with an areal coverage of 729 or 900 km^2 (289 or 324 mi^2) can be effectively used in mapping land use for potential land treatment sites within a certain radius of existing wastewater treatment plants.

Land use data obtained from the satellite and aircraft data products can be combined with readily available information on regional and site-specific characteristics of soils, geology and topography to evaluate land treatment potential. Potential land treatment sites can be identified for future consideration in detailed site-specific design studies.

INTRODUCTION

Regional site characterization of an area is required to provide the basic information necessary to make a preliminary assessment of land treatment potential [1]. The important physical regional features addressed in this paper include land use, soils, geology and topography.

Corps of Engineers attendees at land treatment training courses presented during the past three years have indicated a critical need for techniques, procedures and methodologies for assessing regional site characteristics in the evaluation of land treatment potential. Since the launch of the Landsat-1 satellite and the Skylab satellite mission, satellite data products have been readily available for public use. In addition, NASA high-altitude aircraft photography is available for many areas of the United States.

In many instances the resolution of these satellite and aircraft data products is adequate for identification of land use and various parameters related to hydrology, geology and soils. A methodology using state-of-the-art photointerpretation techniques is discussed for mapping regional land use information to be used in the evaluation of land treatment potential.

The objective of this paper is to demonstrate the use of satellite and aircraft data products in the mapping of regional land use. In addition, available sources for hydrologic, geologic, soils and topographic information are discussed for use in evaluating sites for land treatment potential. The Metropolitan Region of Nashville, Tennessee, Urban Study Area was selected as the case example for demonstrating the use of satellite data products in regional site selection (Fig. 1). The Phase One report entitled "Feasibility of Land Application of Wastewater in the Metropolitan Region of Nashville, Tennessee, Urban Study Area" has recently been published [2]. In addition, the Sacramento, California, area was selected to demonstrate the use of NASA high-altitude aircraft photography for mapping detailed land use. These study areas provided an opportunity to demonstrate remote sensing techniques in current urban study programs.

CHARACTERISTICS OF THE SATELLITE AND AIRCRAFT DATA PRODUCTS

Landsat

The Landsat-1 satellite (formerly called ERTS-1), launched on 23 July 1972, provides repetitive coverage of the earth every 18 days by means of the multispectral scanner (MSS) system. The satellite operates in a circular, sun-synchronous, near-polar orbit at an altitude of approximately 920 km (572 mi) [3]. It circles the earth every 103 minutes, completing approximately 14 orbits each day [3].

A black and white photographic data product can be processed from the MSS and covers an area of approximately 34,225 km^2 (13,225 mi^2) in the following spectral regions: band 4 (0.5-0.6 μm), band 5 (0.6-0.7 μm), band 6 (0.7-0.8 μm) and band 7 (0.8-1.1 μm). The multispectral data can also be obtained in digital form on computer-compatible tapes (CCTs) [3].

Landsat-2, launched on 22 January 1975, provides the same 18-day repetitive coverage. Landsat-2 was launched so that its orbit followed the track of Landsat-1 at a delay of nine days. In February 1977 the sequence changed to a 6-12 day coverage so that Landsat-1 followed Landsat-2 by 6 days and Landsat-2 followed Landsat-1 by 12 days. The Landsat-1 satellite operations terminated on 16 January 1978. Landsat-3 was launched on 5 March 1978. Repetitive coverage on a 9-day basis now occurs between the Landsat-2 and Landsat-3 satellites.

Skylab

The Skylab satellite program consisted of three separate missions--Skylab-2 (25 May-22 June 1973), Skylab-3 (28 July-25 September 1973) and Skylab-4 (16 November 1973-8 February 1974). The satellite was launched with a 50-degree inclination to the equator so that the theoretical coverage was limited to an area 50 degrees north and south of the equator [4]. The Skylab satellite completed an orbit in 93 minutes and thus the ground track was repeated every five days [4]. The coverage along the ground track was not entirely complete due to weather conditions and the astronauts' schedules. The satellite, at a nominal altitude of 435 km (270 mi), provided visible and near-infrared photographic coverage using the S190A Multispectral Photographic Camera and the S190B Earth Terrain Camera [4].

The S190A camera provided a ground coverage of 26,569 km^2 (10,201 mi^2) for six wavelength bands taken simultaneously: 0.5-0.6 μm and 0.6-0.7 μm (Film type SO-022, Panatomic-X B&W), 0.7-0.8 μm and 0.8-0.9 μm (Film type EK 2424, B&W infrared), 0.4-0.7 μm (hi-resolution color) and 0.50-0.88 μm (color infrared) [4]. The S190B camera provided a ground coverage of 11,881 km^2 (4,624 mi^2) for a selected wavelength band from the following spectral regions: 0.4-0.7 μm (Film type SO-042, hi-resolution color), 0.5-0.7 μm (Film

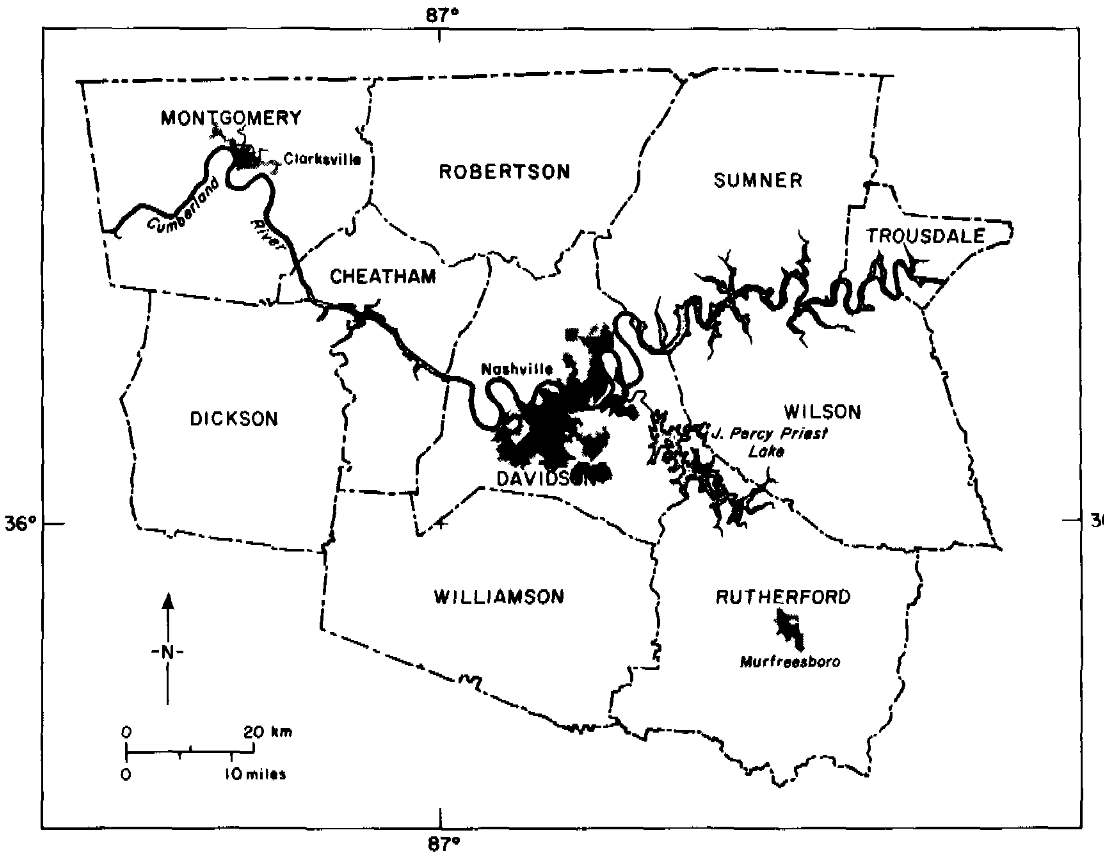

Figure 1. Location map of the Metropolitan Region of Nashville, Tennessee, Urban Study Area.

type EK 3414, hi-definition B&W) and 0.50-0.88 μm (Film type EK 3443, infrared color or SO-131, hi-resolution film infrared color) [4].

NASA Aircraft

NASA high-altitude aircraft photography is currently available for many areas of the United States. There are two NASA aircraft and camera systems which are normally used: an RB-57 aircraft using RC-8 cameras and a U-2 aircraft with an RC-10 camera. Both camera systems provide color infrared photography (Film type 2443, 0.51-0.90 μm) or color film (Film type 2445 or SO-397, 0.4-0.7 μm) in a 9-inch format. The U-2 photography is acquired at an altitude of 19,800 m (65,000 ft). The scale of the photography is 1:130,000. The RB-57 photography is acquired at an altitude of 18,300 m (60,000 ft), resulting in a scale of 1:120,000. Ground coverage for the U-2 and RB-57 photography is 900 km^2 (324 mi^2) and 729 km^2 (289 mi^2), respectively.

Resolution and Scale

The general film characteristics of the Landsat, Skylab and NASA high-altitude aircraft photography are summarized in Table 1 [5]. Table 2 shows the minimum size of detectable objects (ground resolution) which was determined by inspecting the NASA data products and comparing the object to ground measurements [5]. The smallest features that can be recognized on the Landsat imagery are linear features such as roads, bridges, etc. about 70 m (230 ft) in width that contrast sharply with the surrounding terrain [6]. The minimum size of circular or oblate objects detectable on the Skylab S190A photography is about 4,900 m^2 (1.2 acres), whereas on the Landsat imagery the minimum size is about 24,300 m^2 (6 acres) [5].

Table 1. Film characteristics of the NASA data products [modified from 5].

Data product	Transparency dimension (cm)	(in.)	Scale	Estimated ground coverage (km)	(mi)	Area (km^2)	(mi^2)
Landsat Multispectral Scanner (0.6-0.7 μm)	18.6	7.3	1:1,000,000	185	115	34,225	13,225
S190A Multispectral Photographic Camera (0.6-0.7 μm)	5.7	2.25	1:2,850,000	163	101	26,569	10,201
S190B Earth Terrain Camera (CIR 3443)	11.4	4.5	1:950,000	109	68	11,881	4,624
RB-57 RC-8 Camera (CIR 2443)	22.9	9.0	1:120,000	27	17	729	289
U-2 RC-10 Camera (CIR 2443)	22.9	9.0	1:130,000	30	18	900	324

Table 2. Ground resolution of the NASA data products [5].

Data product	Linear features, width (m)	(ft)	Circular features, area (m^2)	(acre)
Landsat	70	230	24,300	6.0
S190A	25	82	4,900	1.2
S190B	12.5	41	3,200	0.8
RB-57	5	16	800	0.1

APPROACH

Acquisition and Selection of Data Products

A listing of all available Landsat imagery can be requested from the NASA Browse Facility*. In addition, a computer search of all available Landsat, Skylab S190A and S190B photography, NASA high-altitude aircraft photography and low-altitude aerial mapping photography can be requested from the EROS Data Center**. Black and white or color paper prints or film transparencies of these data products can be purchased either from the EROS Data Center or the Aerial Photography Field Office***.

The final selection of satellite and aircraft photographs should be based on the cloud cover, date of acquisition and the type of information required.

*Address: NASA Goddard Space Flight Center, Greenbelt Rd., Greenbelt, MD 20771

**Address: U.S. Geological Survey, Sioux Falls, SD 57198

***Address: U.S. Dept. of Agriculture, Agricultural Stabilization and Conservation Service, 2222 West 2300 South, P.O. Box 30010, Salt Lake City, UT 84125

The percentage of cloud cover should be at a minimum so that mapping can be easily accomplished. The optimum season for photographic coverage is dependent on the type of map information required. The photography should be fairly recent as current information is necessary.

Land Use Mapping

The land use classification scheme recommended for the land use mapping has been modified from the U.S. Geological Survey Land Use Classification System [7]. This scheme uses the best criteria of existing land use classification systems to the extent that they are amenable for use with remote sensing data products. The open-ended concept presented by Anderson can be used by regional, state and local agencies to develop detailed land use classification systems for compatibility with the national system. Table 3 shows the land use classification scheme used during the land use mapping.

Land use maps were prepared from black and white contact prints of a Landsat MSS band 5 and 7 image at a scale of 1:500,000 and of an S190A color infrared (0.50-0.88 μm) photograph and S190B color (0.4-0.7 μm) photograph at a

Table 3. Land use classification system [modified from 7].

Level I	Level II
U. Urban and built-up land	1. Residential 2. Commercial and services 3. Industrial 4. Extractive 5. Transportation, communication and utilities 6. Institutional 7. Strip and clustered settlement 8. Mixed 9. Open and other
A. Agricultural land	1. Cropland and pasture 2. Orchards
F. Forest land	1. Mixed
W. Water	1. Streams and waterways 2. Lakes 3. Reservoirs
N. Nonforested wetlands	1. Vegetated 2. Bare
B. Barren land	

scale of 1:250,000. These wavelength bands were selected because the land use was prominently displayed for the defined spectral regions. Black and white photographs were contrast-enhanced photographically to portray the maximum number of grey tones. During the mapping, reference was made to the original color and color infrared transparencies.

The MSS band 5 Landsat photomosaic and land use map of the Nashville, Tennessee, area are shown in Figure 2. The Landsat imagery can be enlarged to scales of 1:500,000 and 1:250,000 to map land use on a regional basis. The level I land use units, such as forest, agricultural, and urban and built-up land, can be easily differentiated on the MSS band 5 image. However, the water category was mapped using MSS band 7 because water appears black and contrasts sharply with the surrounding terrain in the near infrared band.

The tones, textures and patterns evident on the black and white imagery were the interpretation factors used in the land use mapping. The black tones on MSS band 7 indicate water. The Cumberland River and J. Percy Priest Reservoir are the two main water bodies and smaller tributary streams and lakes can also be seen.

The dark grey tones on the MSS band 5 image are the forest areas. The agricultural land use unit shows a characteristic rectangular pattern indicating cropland, pasture and row crops and exhibits grey and white tones. Smaller parcels of forest land in the agricultural areas are characterized by predominantly light grey tones.

The urban and built-up land use unit is composed of predominantly white tones on the photography. Interstate highways serving the Nashville area can be easily delineated on the MSS band 5 image.

A potential advantage in using Landsat imagery is its repetitive coverage on a seasonal and annual basis. This can enable more efficient updating of land use data and identification of areas where changes in land use are occurring [8]. Evaluation of maps prepared from Landsat data in other studies indicates that all level I land use categories can be correctly identified to an accuracy exceeding 90% [9].

The Skylab S190A photomosaic and land use map of the Nashville, Tennessee, area are shown in Figure 3. The same photointerpretation techniques of mapping tones and textures on the Landsat imagery were applied to the S190A photograph. Reference was often made to the color infrared transparency product for ease in discriminating land use. In general, more detail was observed on the S190A photography at the mapping scale of 1:250,000.

Level II land use units could be identified on the S190A photograph, particularly in the urban areas. Airports, secondary roads, commercial complexes and housing density were more easily discerned than on the Landsat image.

The Skylab S190B photomosaic and land use map of the Nashville, Tennessee, area are shown in Figure 4. Again, photointerpretation techniques used in the Landsat and S190A land use mapping were employed. The greatest detail of all the satellite data products was observed on the S190B photography at the mapping scale of 1:250,000. The level I and II units observed on the S190B photography were more detailed than on the S190A photograph.

NASA high-altitude aircraft photography was not available for the Nashville, Tennessee, area. Therefore, the Sacramento, California, area, where a

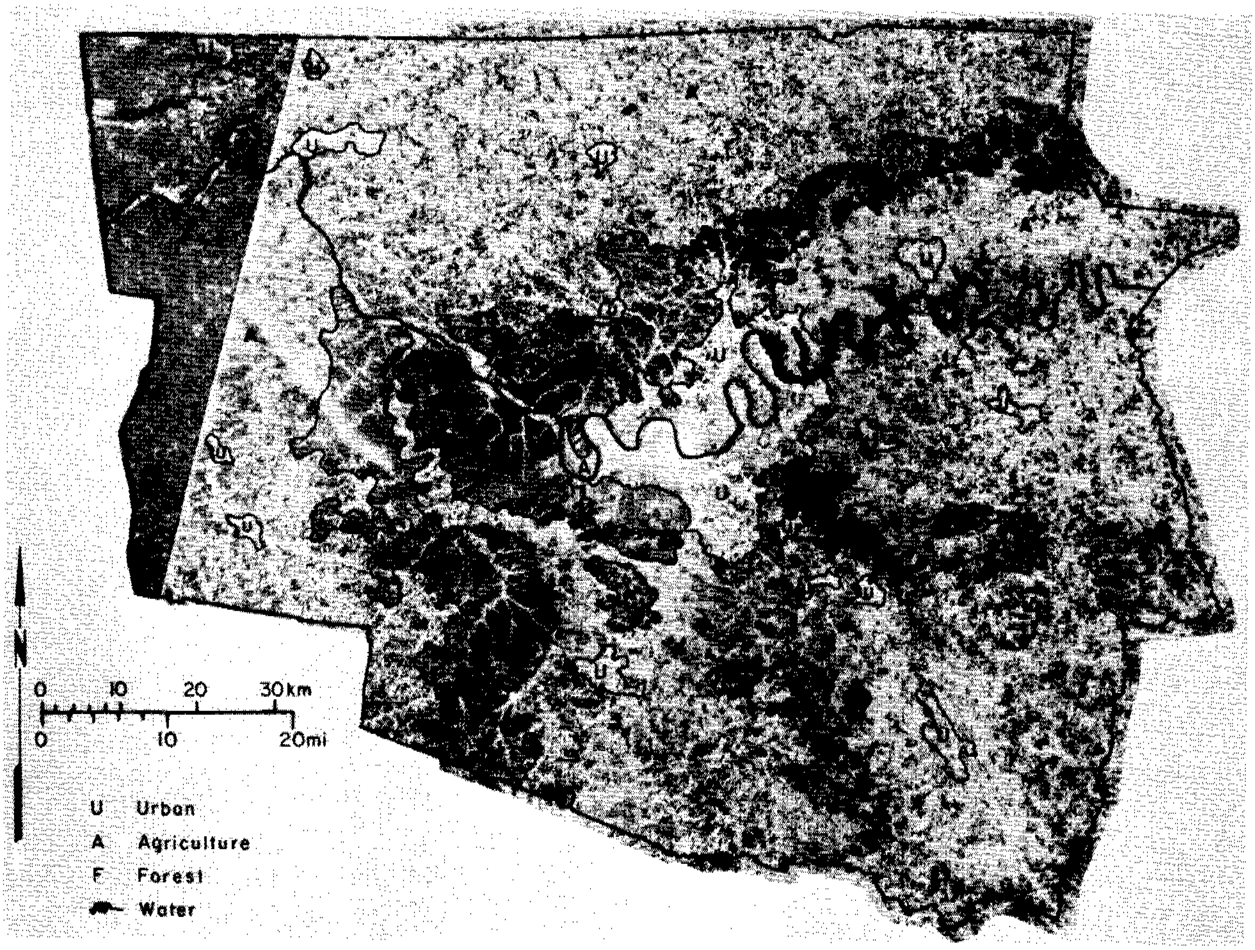

Figure 2. Landsat-2 photomosaic (MSS band 5) and land use map of the Nashville, Tennessee, area (NASA scene IDs 2443-15442 and 2586-15344; acquired 9 April and 30 August 1976, respectively).

similar wastewater management study had been conducted, was selected to illustrate the use of NASA high-altitude aircraft photography in the mapping of potential land treatment sites. U-2 color infrared photography (scale 1:130,000) acquired in July 1972 utilizing an RC-10 camera was used as a data base to define potential land treatment sites (Fig. 5) [10]. Potential land treatment sites were evaluated within a 1.6-km (1-mi) radius of existing wastewater treatment plants.

It can be seen from Figure 5 that very detailed land use mapping can be accomplished using high-altitude aircraft photography. Fallow and vegetated agricultural land can be easily mapped using the original color infrared photographic product as the fallow lands appear blue and the vegetated areas appear bright red. Very detailed mapping of the urban and built-up land use units can be made and the rural area housing patterns can be delineated. Land treatment area requirements for an 8,634 m^3/yr (7 acre-ft/yr) application rate were determined based on general soil associations and associated soil permeability rates [10]. Preliminary consideration of the land use mapped from high-altitude aircraft photography and soils information indicated that potential land areas were available for 13 out of 18 wastewater treatment plants [10]. Therefore, it was determined that land treatment could be used to upgrade these existing wastewater treatment plants in the Sacramento area as an alternative to conventional treatment [10].

RESULTS AND DISCUSSION

The time involved in mapping the ten-county area of 12,200 km^2 (4,700 mi^2) surrounding Nashville for the

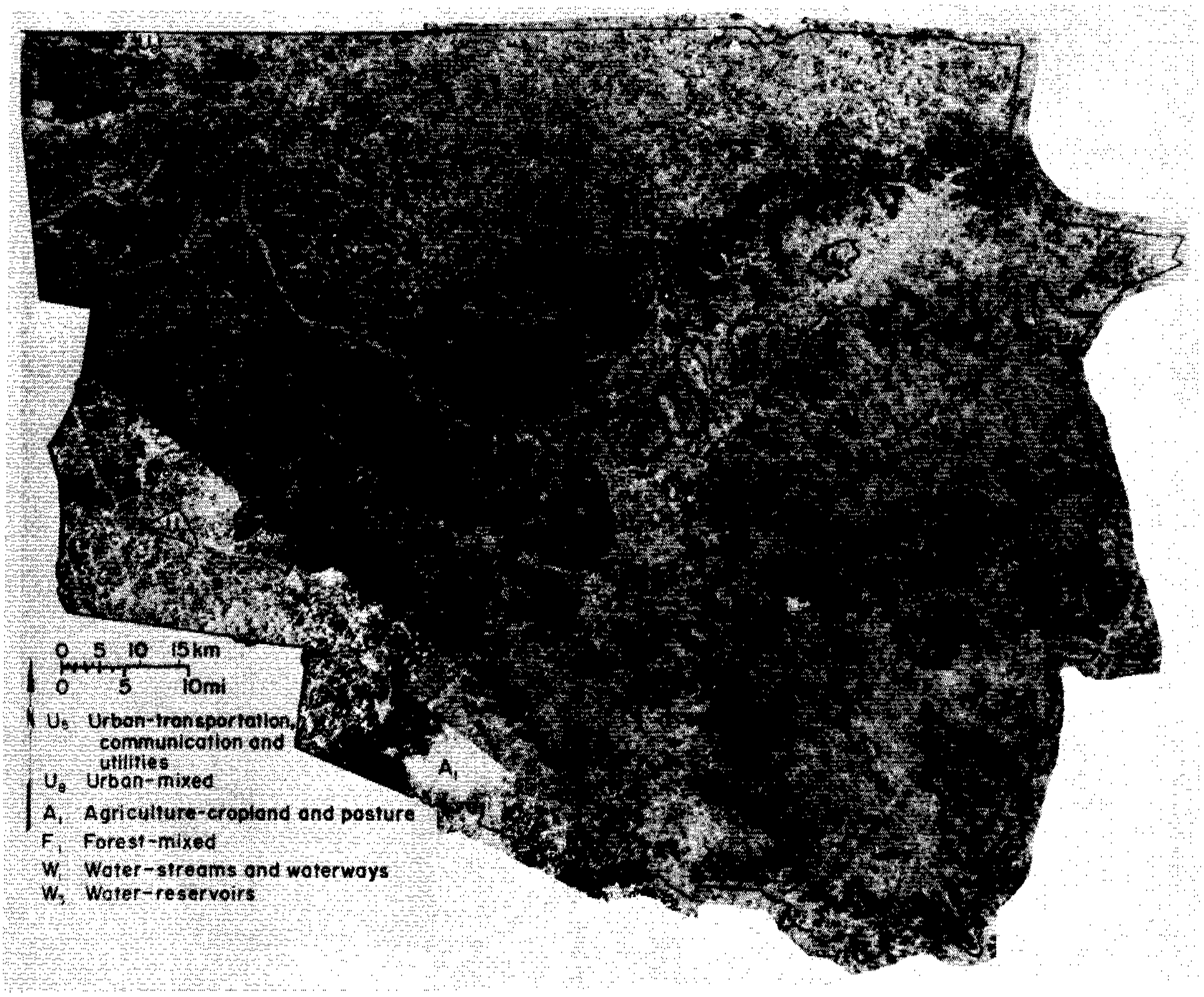

Figure 3. Skylab S190A photomosaic and land use map of the Nashville, Tennessee, area [NASA Scene IDs 51-062 and 51-063 (acquired 30 November 1973); 09-151 and 09-152 (acquired 9 June 1973)].

Landsat image was approximately two hours. The Landsat map was the least detailed of the maps prepared, but can be used effectively for mapping regional land use at a resolution of 70 m (230 ft). The Landsat imagery provides for a large areal coverage of 34,225 km^2 (13,225 mi^2) on a repetitive basis.

The Landsat map can be used to identify where land treatment potential would be low, moderate or high. The urban and built-up land, nonforested wetlands and barren land use units can be considered as low potential areas for land treatment. The forest land would be considered as moderate potential for land treatment. The agricultural land can be considered as the highest potential for land treatment.

The S190A map took four hours to complete and the land use map was significantly better than the Landsat map. An area of 26,569 km^2 (10,201 mi^2) can be readily screened for potential land treatment sites. In addition, more detailed delineation of moderate or high potential areas can be accomplished using the S190A photography.

The S190B map took approximately eight hours and was the most detailed of all the satellite land use maps. Although the S190B map took the longest time to prepare, it provided the most information on land use. More detailed maps of selected moderate and high potential land treatment areas, previously defined by the Landsat and S190A land use maps, can be accomplished using the S190B photography. A Skylab S190B photograph covers an area of 11,881 km^2 (4,624 mi^2) and has a resolution of 12.5 m (41 ft).

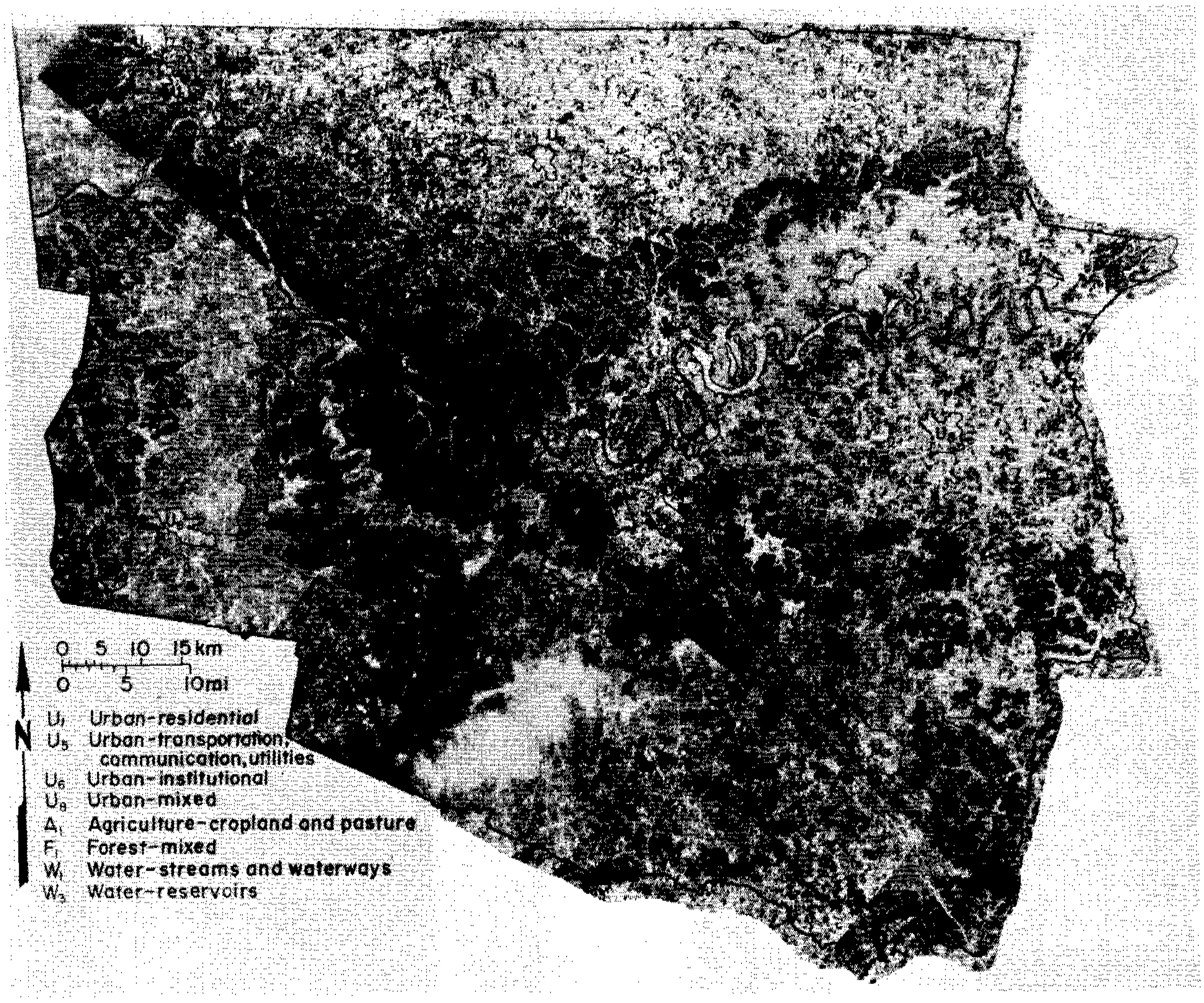

Figure 4. Skylab S190B photomosaic and land use map of the Nashville, Tennessee, area [NASA Scene IDs 90-034 and 90-036 (acquired 30 November 1973) 81-198, 81-199 and 81-200 (acquired 9 June 1973)].

Available NASA high-altitude aircraft photography can be used for detailed level I, II and III* land use classification purposes. In addition, medium- and low-altitude photography can be obtained from the EROS Data Center, USDA-ASCS, U.S. Forest Service, Bureau of Land Management and local Soil Conservation Service offices for mapping land use in a site-specific analysis of land treatment.

The land use maps prepared from satellite and aircraft data products can be used in conjunction with other information to evaluate land treatment potential. Some of the necessary data include information on soils, geology and topography.

*Level III land use units are more detailed delineations of level II land use categories.

Other Available Information for Site Evaluation of Land Treatment

Soils. A soil association map can provide sufficient information on physical and hydraulic soil characteristics in the evaluation of soils for potential land treatment. A soil association comprises one or more major soil series and at least one minor soil series [11].

When evaluating soil association maps, the physical and hydraulic characteristics of a soil which should be considered in a regional analysis are:

Figure 5. NASA U-2 high-altitude aircraft photography of the Sacramento, California, area (Scene ID Uag 1045 153.22 1973; acquired 27 July 1972). Each circle represents a 1.6-km (1-mi) radius surrounding existing wastewater treatment plants.

depth of soil to bedrock, soil permeability and slope. Soil series information within the soil association can be used in a rating process for determining the overall land treatment potential of the soil association [2].

For site-specific analyses detailed soil surveys have been published for many counties of the United States and can be obtained from the U.S. Department of Agriculture-Soil Conservation Service [12]. In addition, detailed soil surveys are available for many portions of foreign countries. Soil surveys contain detailed and general soils maps, general information about the agriculture and climate of the area, descriptions of each soil, percent slopes occurring within a soil unit, discussion on soil formation and classification, and laboratory data on physical and chemical characteristics of soils. Soil surveys published since 1957 are printed on a photomosaic base at scales of 1:24,000, 1:20,000 or 1:15,840 and provide soil interpretation keys which are useful for engineering purposes [12].

During the last several years computer processing of Landsat digital data has been used in the preparation of regional soils maps [13, 14]. These soils maps could be combined with ancillary digitized information on topography, geology, vegetation, land use and physiographic information for one, two or more passes to provide comprehensive information on environmental conditions [8].

Another publication available from the U.S. Department of Agriculture contains a map of major land resource areas [15]. The dominant physical characteristics of each land resource region and area are described briefly in terms of land use, elevation, topography, climate, water and soils [15]. This map would provide valuable information on the regional geographic and physical setting of an area.

Geology. The geology of an area is another important factor for regional evaluation of potential land treatment sites. Various geologic maps are available to obtain regional information on geology.

A Geologic Map of the United States (scale 1:2,500,000) is available with explanatory texts to describe the Precambrian, Paleozoic and Mesozoic rocks [16, 17, 18]. A Glacial Map of the United States East of the Rocky Mountains (scale 1:1,750,000) can be used for interpreting glacial deposits [19]. A Tectonics Map of North America (scale 1:5,000,000) and the Tectonic Map of the United States (scale 1:2,500,000) can be used in geologic interpretation of lineaments and faults [20, 21].

Geologic maps can be obtained for each state from the U.S. Geological Survey* or the respective State Geological Survey. A geologic map index available for each state indicates the areas for which geologic maps have been published. A bedrock and/or surficial geology map at a scale of 1:500,000 is available for many states for use in evaluating regional geology.

The U.S. Geological Survey has published a list of geologic and water-supply reports and maps for each state which are related to the geology, mineral and water resources of an individual state. This publication contains a list of available bedrock and/or surficial geologic quadrangle maps (scale 1:24,000).

During the last few years Landsat, Skylab and aircraft photography have been used in a number of studies for geomorphic interpretation of the landscape [22, 23, 24, 25, 26, 27]. Satellite and aircraft photographs can be used as a source of geologic information for engineering purposes. This geologic information would include: identification and location of soil materials, both granular and clay and clayey silt soils; analysis of structural geology; and location of sample areas for detailed investigation of soil and rock materials [27]. Interpretations of surficial materials and bedrock can be made based on diagnostic photographic characteristics such as drainage characteristics, surface expression of the soil materials or soil patterns, identification of landforms, erosional characteristics, photographic tones and colors, and the distribution and type of vegetation [27]. Satellite and aircraft photographs also provide significant information on the location of faults and lineaments as some of these structural trends are recognized from subtle expressions on photographs, but are identified only with difficulty on the ground [27].

Topography. For regional analysis combined slope categories can be used to indicate generalized topographic characteristics of an area. Various regional topographic maps (1:1,000,000, 1:500,000 and 1:250,000) and other special maps and sheets can be obtained for each state from the U.S. Geological Survey. Also, 7 1/2-minute (scale 1:24,000), 15-minute (scale 1:62,500) and 30-minute (1:125,000) quadrangle map series are available for each state. The contour intervals of these maps vary according to the map scale and the terrain relief. A state index of topographic maps indicates the extent of map coverage and the availability of these maps.

In addition, limited coverage of orthophotoquad maps of the United States are available from the U.S. Geological Survey. An orthophotograph is a map showing a photograph overprinted onto a standard topographic quadrangle map. These maps can be quite useful if photographic coverage is limited for an area.

*Address: Branch of Distribution, 1200 So. East St., Arlington, VA 22202

SUMMARY AND CONCLUSIONS

Landsat and Skylab photographic data products were used in a demonstration of regional site selection techniques for mapping land use in the Nashville, Tennessee, area. Land use maps were prepared using black and white photographic prints at a scale of 1:500,000 for the Landsat imagery and 1:250,000 for the Skylab S190A and S190B photography.

The resolution, large areal coverage and repetitive coverage of the Landsat imagery can provide regional level I land use surveys with minimal mapping time. It required two hours to map over 12,200 km^2 (4,700 km^2) in the Nashville, Tennessee, area. Areas with a resolution of 70 m (230 ft) could be rapidly scanned within the 34,225 km^2 (13,225 mi^2) areal coverage of a Landsat image. Large areas, such as the urban and built-up land, could be eliminated and considered as low potential areas for land treatment. The Landsat land use map can also be used to identify land areas, such as agricultural and forest land, where land treatment potential would be moderate or high.

The Skylab S190A multispectral photography can provide detailed level I and II land use maps. The mapping time for the S190A land use map of the Nashville, Tennessee, area was four hours; but the S190A map provided for more detailed delineation of the level I land use categories. Areas could be rapidly evaluated within the 26,569 km^2 (10,201 mi^2) areal coverage at a resolution of 25 m (82 ft) of an S190A photograph.

The Skylab S190B photography provided for more detailed level I and II land use mapping. The mapping time of the S190B land use map increased to eight hours, but the map was more detailed than the Landsat and S190A maps. The Skylab S190B photography can be used for more detailed delineation of moderate and high potential land treatment areas previously defined using the Landsat and S190A photography. The S190B land use map can be used to evaluate land areas to an accuracy of 12.5 m (41 ft) within a 11,881 km^2 (4,624 mi^2) areal coverage of an S190B photograph.

More detailed information can be observed from the NASA high-altitude aircraft photography. The large scale photography (scale 1:120,000 and 1:130,000) can be used for detailed land use mapping within an area of 729 or 900 km^2 (289 or 324 mi^2) at a resolution of approximately 5 m (16 ft). Potential land treatment sites, which can be previously identified on an area-wide basis using the satellite photography, can be mapped in more detail using the NASA high-altitude aircraft photography. The potential sites can then be evaluated in conjunction with other information sources for the slow infiltration, rapid infiltration and overland flow modes of land treatment.

The land use information mapped from satellite and aircraft photographic products can be combined with readily available information on regional and site-specific soils, geology and topography. Using geomorphic interpretation of the photographs and map overlays of soils, geology and topography, potential land treatment sites can be identified for future consideration in site-specific design studies.

ACKNOWLEDGMENTS

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LAND TREATMENT MATHEMATICAL MODELING

REVIEW OF PHYSICAL/CHEMICAL/BIOLOGICAL MODELS FOR PREDICTION OF PERCOLATE WATER QUALITY

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Physical/chemical/biological models for quality and quantity of drainage water, and thus input for optimal design and management of land treatment systems are reviewed.

Differences and similarities among various models are pointed out in terms of: (a) type of flow, (b) initial and boundary conditions, (c) presence of plant roots, (d) type of salt flow mechanisms, (e) initial condition of salt flow models, and (f) various interactions with soil particles or with other salts in solution. The models can be classified into: (a) analytical solutions, (b) numerical solutions, or (c) a combination of analytical and numerical solutions.

An illustration of predicted water quality using the Bureau of Reclamation's combination type model versus measured values from an experiment at Apple Valley, Minnesota, is included. The computed versus measured quantity and quality of drain water on a seasonal basis appears good.

INTRODUCTION

Land application of wastewater provides a means of returning good quality water to its natural cycle while also benefitting agriculture. However, proper application of wastewater coupled with wise soil-crop management decision is essential. Over or under application of wastewater can dramatically affect the nutrient balance in soil, nutrient availability to plants, and thus the quality of wastewater as it passes through an irrigation cycle. The principal factors for consideration in salt balance in soils are evaporation, transpiration, ion exchange, leaching, and precipitation of salts. These processes, in turn, are influenced by the quality and quantity of applied water, physical and chemical characteristics of the soil, types of crops, method of water application, type of drainage, climatic conditions, and biochemical reactions. Most of the models that describe the simultaneous flow of salt and water are based on the principles describing the above processes. These models offer the possibility of quickly assessing the long-term effects of wastewater application schemes on salt balance in soil and the quality of percolate water.

The objective of the paper is to review the physical/chemical/biological models that can provide the best available automated procedure for predicting the quality of percolate water and thus input for optimal design and management of land treatment systems. Because of the space limitation, the review in this paper is not comprehensive and only important features of each model are presented. Although duplication is unavoidable, this review article will supplement some of the earlier reviews, e.g., Gardner (1965), Frissel and Poelstra (1967), Biggar and Nielsen (1967), and Boast (1973).

Table 1 summarizes some of the salt and water flow models discussed in the literature. Differences and similarities among various models are pointed out in terms of: (a) type of flow, (b) initial and boundary conditions,

Table 1. Review of available salt and water flow models.

Authors/Year	Water flow model											Salt flow model															Validation	
	Type of flow		Initial conditions				Boundary conditions					Type of flow			Initial conditions		Sink or source		Exchange reactions				Precipitation & Solubilization					
	Steady	Transient	Saturated	Unsaturated	Uniform initial water content	Variable initial water content	Infiltration	Redistribution	Evaporation	Transpiration	Water table	Convective flow	Molecular diffusion	Hydrodynamic dispersion	Uniform initial salt concentration	Variable initial salt concentration	Non-interacting solutes	Interacting solutes	Exchange reaction considered	Ca-Mg	Ca-Na	Mg-Na	$CaSO_4$	$CaCO_3$	Undissociated Ca, Mg, SO_4, ion pair	Nitrogen transformation	Laboratory	Field
												ANALYTICAL SOLUTIONS																
1. Wilson (1940)	x		x	x	x		x					x			x			x	x									
2. DeVault (1943)	x		x	x	x		x					x			x			x	x									
3. Glueckauf (1949)	x		x	x	x		x					x			x			x										
4. Lapidus & Amundson (1952)	x		x	x	x		x					x	x		x			x										
5. Ribble & Davis (1955)	x		x		x		x					x			x			x	x	x	x						x	
6. Molen (1956)	x			x	x		x					x			x		x											x
7. Bower et al. (1957)	x		x	x	x		x					x			x			x	x	x	x						x	
8. Gardner & Brooks (1957)	x		x	x	x		x					x			x			x	x	x	x						x	
9. Nielsen & Biggar (1962)	x		x	x	x		x					x	x		x		x										x	
10. Biggar & Nielsen (1963)	x		x	x			x					x					x	x	x								x	
11. Elrick et al. (1966)	x		x	x	x		x					x	x	x	x		x	x								x	x	
12. Cho (1971)	x		x	x	x		x					x	x		x		x	x								x		
13. Misra et al. (1974)	x		x	x	x		x					x	x		x		x	x								x	x	
14. Skopp & Warrick (1974)	x		x	x	x		x					x	x		x			x		x								
15. Cassel et al. (1975)	x		x	x	x		x					x	x		x		x										x	
16. van Genuchten & Wierenga (1976)	x		x	x	x		x					x	x	x	x		x	x										
												NUMERICAL SOLUTIONS																
17. Bresler (1967)	x			x		x	x					x				x	x											x
18. Bresler & Hanks (1969)	x	x	x	x		x	x	x	x		x	x				x	x										x	
19. Lai & Jurinak (1971)	x			x	x		x					x	x		x			x	x	x	x	x					x	
20. Warrick et al. (1971)	x	x	x	x		x	x					x	x				x											x
21. Dutt et al. (1972)	x	x	x	x		x	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x	x	x	
22. Gupta (1972)	x	x	x	x		x	x	x	x	x	x	x				x	x	x	x	x	x	x	x	x	x		x	x
23. Bresler (1973)	x	x	x	x		x	x	x	x		x	x	x	x													x	x
24. Kirda et al. (1973)	x	x	x	x		x	x	x				x	x			x	x										x	
25. Shaffer et al. (1977)	x	x	x	x		x	x	x	x	x	x	x	x			x	x	x	x	x	x	x	x	x	x	x	x	x

(c) presence of plant roots, (d) type of salt flow mechanisms, (e) initial condition of salt flow models, and (f) various interactions with soil particles or with other salts in solution. These models can be classified into: (a) analytical solutions (1 through 16), and (b) numerical solutions (17 through 24), or (c) a combination of both (25).

Salt and water flow models have also been classified as rate or plate models. However, this classification only applies to the source or sink term in the model. In a rate model, kinetics is the controlling mechanism in the calculation of the source or sink term, whereas in plate models, steady-state equilibrium is assumed between the adsorbed solution and solid phase in each segment. Either of these approaches could be used in models based on analytical or numerical solutions.

Models Based on Analytical Solutions

In these models, an entire soil column is considered as a unit of calculation and thus initial water and salt contents are assumed to be constant throughout the column. These models only predict the quality of percolate water and provide very little information on salt and water content distribution in the soil column. Using miscible displacement experiments, these models have been tested in the laboratory on homogeneous soil columns under steady-state infiltration conditions. Predictions from these models have been reasonably good. However, water flow in the field is far from steady state, and is often simultaneously moving downward (infiltration) and upward (evaporation). The system is further complicated by removal of water by roots from deeper layers. Thus the applicability to field situations is limited because of simplified assumptions. However, experiments where the miscible displacement technique has been used for validation of these models has proved useful in furthering knowledge about the basic mechanism of simultaneous flow of salt and water in soil.

Models Based on Numerical Solutions

In these models the soil column is divided into a number of segments, each of which is considered uniform with respect to chemical and physical properties. Steady state equilibrium between adsorbed solution and solid phases is then assumed within each segment. With the availability of digital computers, application of numerical methods to problems of salt and water flow in soils with complex initial and boundary conditions has been made easier.

Models of Gupta (1972) and Dutt et al. (1972) are examples of use of numerical solutions. They are essentially a combination of the water flow model by Hanks and Bower (1962), the convective salt flow model by Bresler (1967), and the model on the interaction of salt with soil or other salts by Dutt (1962).

Some of the models (Gupta, 1972, and Nimah and Hanks, 1973) assume that the quality of percolate water in drains is similar to that of the layer in the soil immediately above the water table. This is erroneous in artificially drained soils because of differences in leaching patterns between the drains. Even in homogeneous soils, if drainage is achieved by means of tile drains or ditches, leaching patterns will not be uniform. Several times as much water will pass through the soil located adjacent to the tile line as will through that midway between lines. The Bureau of Reclamation's Irrigation Return Flow Model (Shaffer et al., 1977) is a further modification of the model of Dutt et al. (1972) to take into account the presence of drains and is a combination-type model. Water flow models above drains are based on numerical solutions, whereas drainage discharge is calculated from the analytical solution of the water flow equation. Combinations of both dynamic and steady-state equilibrium are used in the calculation of source and sink terms. Other modifications included in the Bureau's model account for chemical changes and transformation of nitrogen compounds in the saturated zone adjacent to drains and also during mixing of discharge water in the drains from different stream lines.

THEORETICAL BASIS

In general, models developed for simultaneous flow of water and salt are based on the laws of conservation of mass. They state that the amount of material applied to the soil layer minus that amount leached and the amount absorbed by plants is equal to the net increment (positive or negative) of the material in the soil layer. These principles have been supplemented with statistical or emperical relationships when required.

The Bureau's model for predicting the quality of irrigations return flow seems to be the most complete model available to date, which can be extended to the problems of land treatment of wastewater. Certain assumptions which are simplified for a given system may need further improvement. In the next section we will discuss the important processes considered in the construction of the Bureau's model. Differences and similarities with other models are also discussed. The Bureau's model is essentially made up of two submodels: (a) the water flow model and (b) the salt and nutrient flow model. Other models in Table 1 are also based on the following basic relationships or some variation of these **relationships.**

Water Flow Model

The water flow submodel is based on the numerical approximation of the basic water flow equation

$$\frac{\partial \theta}{\partial t} = \frac{\partial \bar{V}}{\partial Z} \qquad [1]$$

Where $\bar{V}$ is the volumetric flux of water as given by Darcy's Law

$$\bar{V} = -K(\theta)\frac{dH}{dZ} \qquad [2]$$

Symbols in these equations and those to follow are defined at the end of the paper.

A review of the solutions of equation [1] has been given by Freeze (1969).

To account for water loss by evapotranspiration and thus root extraction, adjustment in the water content as predicted from equation [1] is made by subtracting a fraction of the consumptive use that is being lost from a particular layer. This fraction is proportional to the root distribution with depth. Root distribution was assumed being constant throughout the season in the Bureau's model. Although this is a simplified approach to account for roots, sophisticated models given by Nimah and Hanks (1973) were included by Gupta (1972) and can be incorporated in the Bureau's model. Most of the models based on the analytical solution of equation [1] have assumed a steady-state condition and thus bypassed the need for a water flow model.

A computed (equation [1]) quantity of water moved into the water table from the unsaturated profile above the drains is then used to calculate the position of the water table from equation [3]

$$\frac{\partial Y_{x,t}}{\partial t} = \frac{K(\theta_s)D_a}{S}\frac{\partial^2 Y_{x,t}}{\partial X^2} \qquad [3]$$

Daily increment of flow into the water table could result in instantaneous rise (if water moves from the unsaturated to saturated zone) or instantaneous decline (if water moves from the saturated to unsaturated zone by capillary flow) in the water table. Basically, the water table shape is chosen as a fourth-degree parabola:

$$Y_{x,o} = \frac{8Y_{m,o}}{L^4}(L^3X-3L^2X^2+4LX^3-2X^4) \qquad [4]$$

Drain discharges from both sides of the parallel circular drains are calculated from Darcy's Law (equation [2]) in conjunction with equation [3]. Two situations are considered in the Bureau's model:

(a) Drains above the barrier

$$q = \frac{2\pi K(d' + Y_{m,o})}{L} Y_{m,t} \qquad [5]$$

(b) Drains on the barrier

$$q = \frac{3.64\ K(\theta_s)Y^2_{m,t}}{L} \qquad [6]$$

Salt and Nutrient Flow Model

The salt flow rate at any plane in the direction of flow may be given by:

$$\left[\frac{\partial(\theta C)}{\partial t}\right]_Z = \left[\frac{\partial}{\partial Z}\left(\theta D\frac{\partial C}{\partial Z}\right) - \frac{\partial(\bar{V}C)}{\partial Z} + SS\right]_Z \qquad [7]$$

where the first term on the right represents the contribution from diffusion and dispersion to the flow of solute and the second term respresents the contribution from convective flow. The source or sink term, SS, is due to solubility of minerals and exchange of ions in solution with solid phase.

The diffusion coefficient which appears in equation [7] is not the diffusion coefficient which applies in the absence of water movement (Gardner, 1965). It is rather an apparent diffusion coefficient which is the sum of molecular diffusion and the hydrodynamic dispersion. Hydrodynamic dispersion results from the variation of pore water velocity between (a) the edge and the center of pores and (b) the large and small pores or the dead end pores. This dispersion can enhance the diffusion process and, at sufficiently high velocity, may completely obscure it (Gardner, 1965). Even though some of the models discussėd in Table 1 have not explicitly stated the difference between molecular diffusion and hydrodynamic dispersion, the values used in most of these models were the apparent diffusion coefficient, which is a summation of both molecular diffusion and hydrodynamic dispersion. The Bureau's model assumes the contributions of the diffusion and dispersion process to be negligible compared to convective flow of salts. However, plate models like the bureau's model tend to introduce an error or so-called numerical dispersion in the calculations of convective flow. Experience has shown that proper selection of the number of plates can be used to introduce the correct amount of dispersion in the calculations.

Numerical approximation for the convective flow of salt in equation [7] is used to predict the concentration of salts at various soil depths. These concentrations are then modified for the sink or source term in equation [7]. Various chemical and biological processes that affect the sink or source terms are: (a) nitrogen transformations, including the hydrolysis of urea, mineralization-immobilization of organic-N and ammonium-N, nitrification of ammonium, and immobilization of nitrate, (b) salt reactions, and (c) crop uptake of nitrogen.

The nitrogen rate equation used in the Bureau's model was derived from using multiple regression analysis, reaction rate theory, and statistical thermodynamics. Variables include ammonium, nitrate, organic-N, and ion concentrations; temperature; carbon nitrogen ratio; ionic strength and soil moisture content. Some models in Table 1 have assumed constant temperature, uniform soil and steady-state conditions, and thus eliminated the need for a functional relationship between rate constant and soil properties.

Chemical reactions included in the salt (inorganic) chemistry portion are the solubilization or precipitation of gypsum and calcium carbonate, undissociated ion pair reaction for calcium and magnesium sulfate, calcium-magnesium exchange, calcium-sodium exchange, sodium-ammonium exchange, and pCO_2-Ca^{++} -HCO_3-interactions. The equations used to describe exchange reactions are:

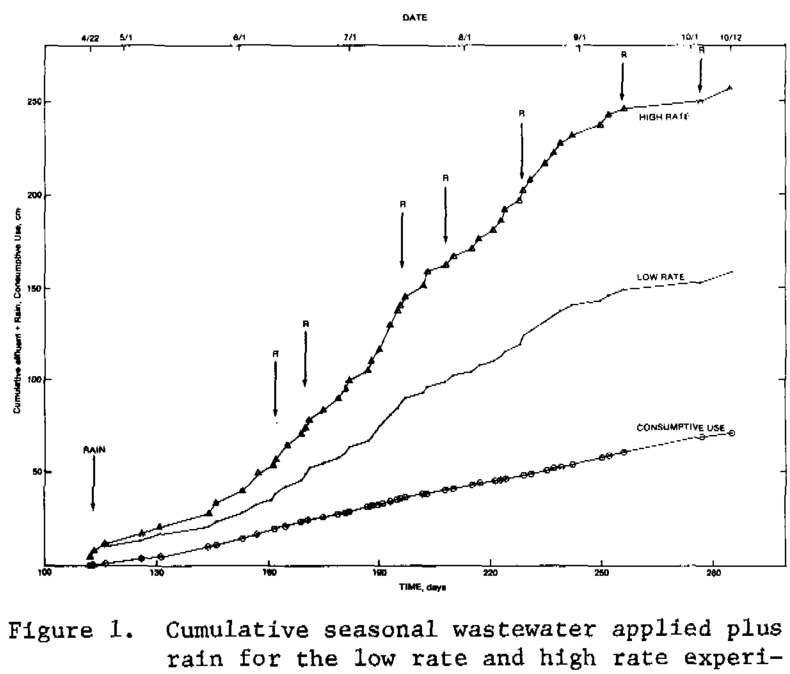

Figure 1. Cumulative seasonal wastewater applied plus rain for the low rate and high rate experiment and consumptive use of water of an experiment at Apple Valley, Minnesota.

$$\frac{a_{Ca}}{a_{Mg}} = K_{Ca-Mg} \frac{E_{Ca}}{E_{Mg}} \qquad [8]$$

$$\frac{\sqrt{a_{Ca}}}{a_{Na}} = K_{Ca-Na} \frac{E_{Ca}}{E_{Na}} \qquad [9]$$

Exchange coefficients were assumed to be constant with solution concentration in the Bureau's model. This may not be true in all soils. Thus models based on variable exchange coefficients (Lai and Jurinak, 1971) should be used for those soils.

Constituents presently considered in the Bureau's model are nitrate, ammonium, calcium, magnesium, sodium, bicarbonate, chloride, carbonate, and sulfate.

Applicability of the Bureau's model to problems of wastewater renovation by agricultural land was checked on experimental data from Apple Valley, Minnesota (Clapp et al., 1977). Input data for the model are given in Table 2.

Table 2. Input data from Apple Valley, Minnesota, experiment used in the Bureau of Reclamation's Irrigation Return Flow Model.

Hydraulic properties of soil above drain

$K(\theta)$	$= 1.059 \times 10^{9}\ \theta^{24.06}$ cm/day
$D(\theta)$	$= 2.058 \times 10^{8}\ \theta^{12.53}$ cm^2/day
ψe	= 10.0 cm
$K(\theta_s)$	= 60.0 cm/day
θ_s	= 0.55 cm^3/cm^3
Bulk Density	= 1.2 g/cm^3

Hydraulic properties of soil below drain

$K(\theta_s)$	= 6420 cm/day
θ_s	= 0.36 cm^3/cm^3
S	= 0.40 cm^3/cm^3
Bulk Density	= 1.7 g/cm^3

Drainage design

Drain spacing	= 1350 cm
Depth to drains	= 150 cm
Depth to barrier	= 1650 cm
Gravel envelope size	= 12.6 cm

Wastewater application

Irrigation schedules and amount - Figure 1

Effective precipitation dates and amount - Figure 1

Soil and wastewater analysis

Parameters	Soil Saturation Extract mg/l	Wastewater mg/l
NH_4-N	0.00	15.12
NO_3-N	3.08	2.38
Ca	72.55	67.53
Na	209.07	294.50
Mg	15.44	24.32
Cl	199.64	414.88
HCO_3	0.00	0.00
CO_3	124.5	36.00
SO_4	253.12	12.01
Organic-N (μg/g)	2.0	0.00
C:N	10:1	
Extract ratio of water to soil	0.25	
$CaSO_4$	0.00	
$CaCO_3$	0.00	
CEC (0-60 cm)	22.0 meq/100 g	
(60-150 cm)	1.0 meq/100 g	

Crop information

Growing season = April 22-Oct. 10, 1976

Crop = reed canarygrass (*Phalaris arundiancea* L.)

Root distribution

Depth (cm)	Roots (Fraction)
0 - 30	0.50
30 - 60	0.25
60 - 90	0.25
90 - 150	0.0

Consumptive use - Figure 1

Soil temperature - Figure 2

N-uptake by plants - Figure 4

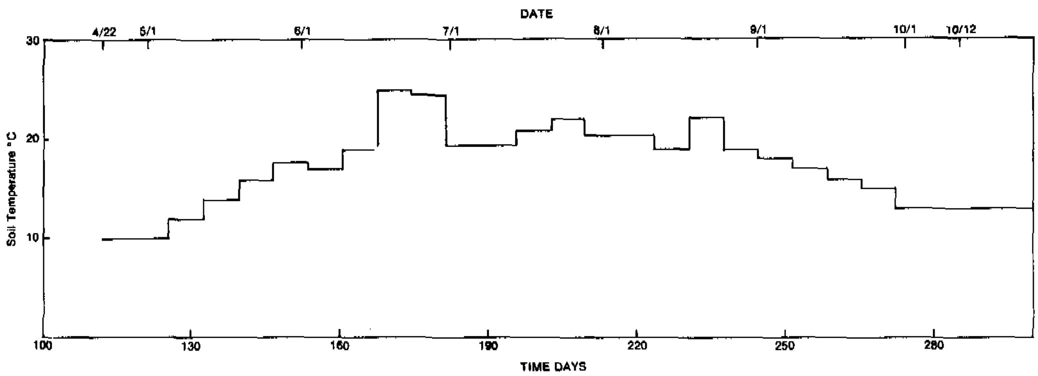

Figure 2. Weekly temperature of the soil profile for an experiment at Apple Valley, Minnesota.

RESULTS AND DISCUSSION

Figure 3 gives the cumulative leachate entering the water table. About 49 and 67% of the wastewater plus precipitation is lost in drainage under low and high amounts of wastewater application, respectively. When estimated by months, the maximum amount of drainage was in the month of July (Table 3) followed by August and June under both high and low treatments. On most months the amount of drainage from the high treatment was about double the drainage from the low treatment. In both cases the change in soil water storage was about 8 cm over the season.

Predicted concentration of various salt species in the drainage water was practically the same for both treatments (Table 3). As expected, these predictions indicated a trend toward steady state chemical equilibrium. Concentrations of salt species (Cl^-, Na^+), which were higher in the wastewater compared to the initial soil solution, gradually increased in their concentration in the drainage water, thus indicating an accumulation of these salts in the soil profile. In the later part of the season Cl^- and Na^+ concentrations in the drainage were close to the concentration of the wastewater applied (Tables 1 and 3). On the other hand, salt species ($CO_3^=$, $SO_4^=$) which had low concentration in the wastewater and high in the soil profile showed some losses from the soil profile. Concentration of these salts ($CO_3^=$, $SO_4^=$) in the drainage decreased with time and were very close to the concentration in the soil profile at the end of the season. In the absence of

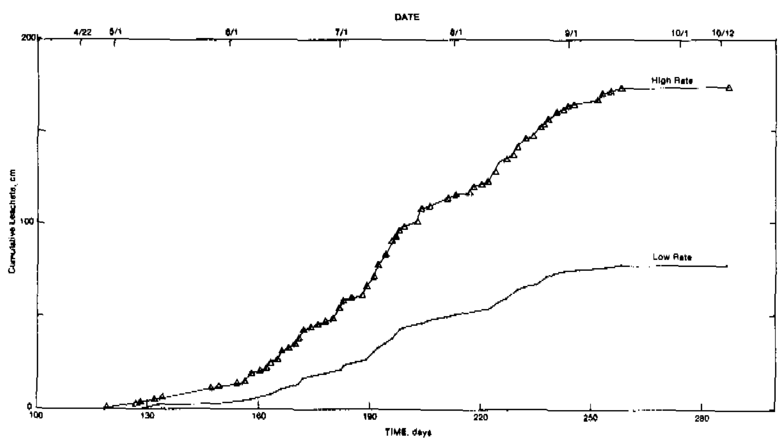

Figure 3. Predicted cumulative leachate using the Bureau of Reclamation's model for the low rate and high rate treatments at Apple Valley, Minnesota.

Table 3. Predicted monthly and predicted seasonal average water quality using the Bureau of Reclamation's Irrigation Return Flow Model and measured seasonal average water quality for Apple Valley, Minnesota, experiment.

Months	Drainage	NO_3-N	NH_4-N	Ca^{++}	Na^{+}	Mg^{++}	Cl^{-}	$CO_3^{=}$	$SO_4^{=}$	Total
	(cm)	mg/l								
						LOW				
May	3.4	3.5	0.00	88	244	18	238	148	296	1036
June	15.9	3.1	0.03	98	262	21	272	147	288	1089
July	30.2	2.5	0.05	113	286	24	334	134	249	1142
August	24.8	2.2	0.06	118	295	25	350	135	250	1176
September	2.9	2.0	0.07	129	309	28	394	121	210	1194
Average Predicted		2.6	0.05	111	283	24	325	137	256	1139
Measured		0.3			255		425			1104
						HIGH				
April	0.6	3.6	0.00	87	243	18	236	147	293	1027
May	12.1	3.1	0.02	90	248	19	243	149	298	1050
June	36.3	2.6	0.05	104	274	22	321	122	223	1069
July	64.5	2.0	0.08	111	290	24	366	106	179	1078
August	50.5	1.9	0.10	112	293	24	368	107	181	1085
September	8.8	1.7	0.13	121	308	26	420	86	125	1088
October	1.5	1.3	0.12	120	308	26	419	84	121	1079
Average Predicted		2.2	0.08	109	286	23	351	111	194	1077
Measured		1.0			289		506			1222

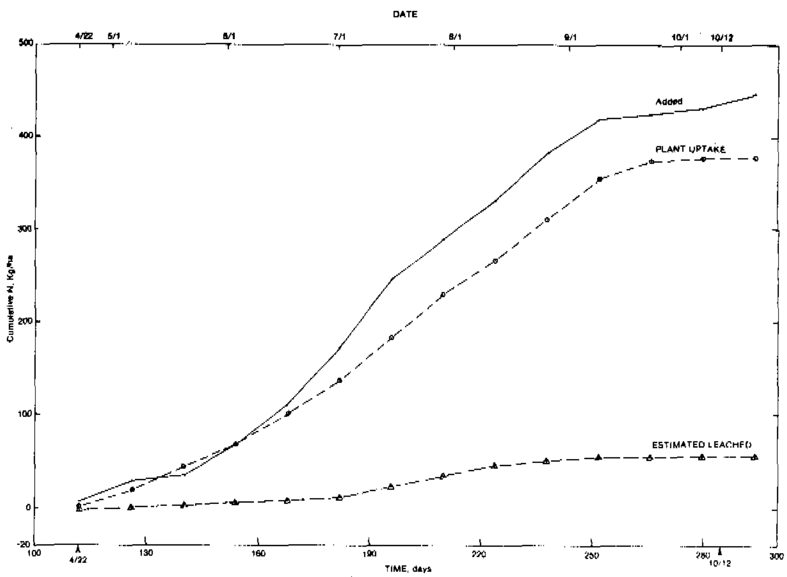

Figure 4. Predicted nitrogen balance using the Bureau of Reclamation's model for the high rate treatment at Apple Valley, Minnesota.

any source, continued application of wastewater would deplete the soil profile of these salts.

Predicted concentrations of some salt species were also compared with measured values (Table 3). These values corresponded to the concentration in the soil water at the interface of coarse sand overlaid with silt loam soil. Soil water was extracted by suction cup samplers. Since permeability of coarse sand is considerably higher than that of silt loam, quality of drainage water was assumed to be same as that of soil water at the interface of these layers. In general, average predicted concentration of NO_3-N, Na^+, Cl^-, and total salts in drainage water were in close agreement with measured concentrations of soil water. Concentrations of other salt species were not measured. Predicted total salt concentrations were high in low treatment and low in high treatment which is reverse of the measured values. Though the concentrations of various salt species in the drainage water were similar under low and high treatment yet the total amount of salts leached was higher under high treatment than low treatment. This was due to the higher amount of leachate under high treatment.

Figure 4 gives the predicted nitrogen balance for the high application treatment at various times during the growing season. The negative leaching on day 112 is due to the upward movement of water and thus nitrogen from the water table into the initial dry profile (θ_i = 0.40). Between 134 and 155 days, plant uptake of nitrogen is greater than the amount added with the wastewater. Differences between these amounts was contributed by mineralization of soil organic nitrogen. Over the season the nitrogen balance showed minor contributions to leaching and plant uptake from organic nitrogen. The nitrogen rate reactions in the model were not developed for land application of wastewater and may need modification for this purpose.

CONCLUSIONS

The irrigation return flow model of the Bureau of Reclamation satisfactorily predicted over a season, the quantities and qualities of drainage water resulting from a wastewater experiment at Apple Valley, Minnesota. Further validations are needed for monthly and more frequent predictions. The model needs testing on other soils. The model may need modifications to better handle land application of wastewater situations. Modifications to account for variable root-depth distribution with time and improved estimates of nitrogen transformation coefficients and their relationships with other soil properties may be helpful. This may be achieved from the large volume of literature published after the inception of this model.

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SYMBOLS

C = concentration of salt species

CEC = cation exchange capacity

D = diffusion or dispersion coefficient

D_a = average saturated aquifer thickness

E_{Ca}, E_{Mg}, E_{Na} = concentration of exchangeable calcium, magnesium, and sodium ions on the soil exchangeable complex

H = hydraulic head

$K(\theta_s)$ = saturated hydraulic conductivity

K_{Ca-Mg}, K_{Ca-Na} = exchange constants

L = horizontal drain spacing

S = specific yield or storage coefficient

SS = source or sink term

$\bar{V}$ = volumetric flux of water

X = horizontal axis

$Y_{m,o}$ = initial water table height above the drains at position m

$Y_{m,t}$ = water table height above the drains at position m and time t

$Y_{x,o}$ = initial water table height above the drains at position x

$Y_{x,t}$ = water table height above the drains at position x and time t

Z = depth

a_{Ca}, a_{Mg}, a_{Na} = activities of calcium, magnesium, and sodium ions in the soil solution

d' = effective depth

m = horizontal midpoint between drains (i.e., $m = L/2$)

q = drain discharge rate per unit length

t = time

θ = water content

θ_s = saturated water content

ψ_e = air entry potential

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LAND TREATMENT MATHEMATICAL MODELING

AN EVALUATION OF ONE- AND TWO-DIMENSIONAL SOIL MOISTURE FLOW MODELS

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ABSTRACT

Since the turn of the century attempts have been made to describe moisture flow in porous materials (as soils) by means of physically derived relations. During the first half of the century research was focused on water-saturated soils. Many soils however are for most of the time water-unsaturated. Since about 1950 the unsaturated zone has also received due attention and basic flow equations for unsaturated porous materials have been derived. One equation that has been widely used is the so-called soil moisture flow equation. This equation is a diffusion type of equation that does not arise in other fields of science or engineering. Because of its uniqueness, and even more because of its complexity, much effort was necessary to make the equation suited for numerical computations. Although many physical and mathematical aspects as related to the soil moisture flow equation still are unsolved, the equation is increasingly used in field studies that deal with moisture and solute movement, with deep seepage and evaporation, and with storage and plant availability of soil water. In the present paper four such studies are discussed. It is shown that soil moisture flow models can provide data about movement and storage of soil water that may be accurate enough for many practical purposes.

THE UNSATURATED SOIL MOISTURE FLOW EQUATION

With use of a three-dimensional cartesian coordinate system the unsaturated soil moisture flow equation can be expressed as:

$$\partial(k\partial h/\partial x)/\partial x + \partial(k\partial h/\partial y)/\partial y + \partial(k\partial h/\partial z)/\partial z = \partial\theta/\partial t \qquad (1)$$

In eq. 1 k denotes the hydraulic conductivity of the soil, h stands for the hydraulic head, θ for the volumetric moisture content of the soil, x, y and z are space coordinates and t represents time. A derivation of eq. 1 can be found in Kirkham and Powers (1972, p. 239). The hydraulic conductivity k, in eq. 1 is not a constant, but a function of the water content θ. Only for saturated soils can k be considered as a constant. Since it is usually also assumed that the water content in saturated soils is constant, eq. 1 for saturated soils can be written as:

$$\partial^2 h/\partial x^2 + \partial^2 h/\partial y^2 + \partial^2 h/\partial z^2 = 0 \qquad (2)$$

Eq. 2 has been extensively used in the soil literature, especially in drainage studies. In the present paper however no saturated flow problems will be dealt with. For one- and two-dimensional flow problems in saturated soils the reader is referred to Kirkham and Powers (1972).

For a flat unsaturated soil eq. 1 usually is written as

$$\partial(k\partial h/\partial z)/\partial z = \partial\theta/\partial t, \tag{3}$$

in which expression z denotes the soil depth, perpendicular to the soil surface. If water uptake by the root system of a plant canopy must be taken into account, eqs. 1 and 3 can be expanded. This can be done by adding a sink term, Q to the left hand side of eqs. 1 and 3. Eq. 3 then can be written as:

$$\partial(k\partial h/\partial z)/\partial z - Q = \partial\theta/\partial t \tag{4}$$

Solutions to equations like 3 and 4 allow the calculation of the moisture content θ at any depth and at any time in the soil profile. Such solutions also enable the calculation of changes in storage and seepage rates, quantities that are experimentally not easily determined. However it is not a simple matter to do computational work with eqs. 3 and 4.

NUMERICAL METHODS AND APPLICABILITY OF THE SOIL MOISTURE FLOW EQUATION

A variety of methods has been developed to solve the unsaturated soil moisture flow equation, especially for one-dimensional flow. None of the methods so far developed is completely satisfying, and new techniques are continuously being propose. Analytical solutions, as discusses by Philip (1969), have only minor relevance to field problems, since they require too many simplifying assumptions. Numerical solutions have a much larger range of applicability, but require large digital computers and extensive amounts of computer time. A review of some of the commonly used numerical techniques to solve eq. 3 is given by Haverkamp et al. (1977). Hornung (1977) has recently developed a new method that appears promising. Seminumerical methods, that do not use as much computer time as truly numerical solutions can be found in Wind and Van Doorne (1975) and Hayhoe (1978).

Most of the numerical methods that have been used to solve eq. 1 are finite difference methods. A recent development is the use of finite element methods. Neuman et al. (1975) and Feddes et al. (1975) have used this technique to handle two-dimensional flow problems. Other examples of finite element solutions to unsaturated flow problems can be found in Gray et al. (1977). A main advantage of finite element methods over finite difference methods is the ease with which irregularly shaped flow regions (two- and three-dimensional flow) can be handled. A discussion of numerical methods in general (finite differences and elements) may be found in Remson et al. (1971).

In order to see how well an equation like eq. 3 describes unsaturated moisture flow in porous materials, experimental data and computational results obtained with eq. 3, can be compared. To solve eq. 3 it is necessary that a boundary condition at the soil surface is specified. Under controlled laboratory conditions this usually is no problem. To test the applicability of eq. 3, many researches have, for a variety of soils, performed such laboratory experiments, i.e. Youngs (1957), Haverkamp et al. (1977), Zaradny (1978). Either a constant hydraulic head or a known flux of water is maintained at the soil surface during the course of the experiment. Fig. 1 is taken from the work of Haverkamp et al. (1977) and shows calculated and measured moisture profiles at different times (in hours) in a 80 cm long sand column. Youngs (1957) and Zaradny (1978) show similar results. Also two-dimensional unsaturated flow under controlled laboratory conditions have been analysed, for example by Brandt et al. (1971) and Bresler et al. (1971) and these authors also report close agreement between calculated and measured moisture profiles.

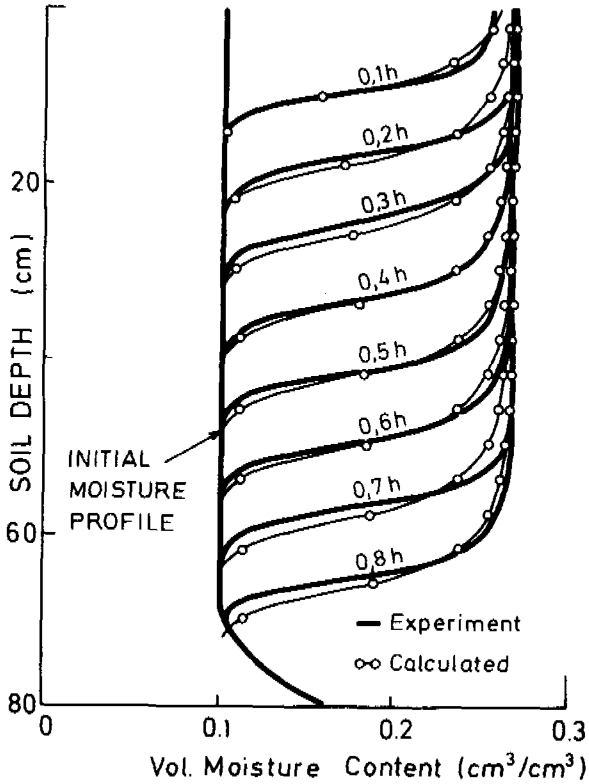

Fig. 1. Calculated and measured moisture profiles in a sand column (after Haverkamp et al., 1977)

Eq. 3 has also been experimentally tested in the field. Nielsen et al. (1961), and Black et al. (1969) for example, have conducted field experiments on infiltration and evaporation, respectively, that also indicate the validity of eq. 3. In both works a time-independent boundary condition (constant hydraulic head) at the soil surface was assumed. It must be admitted however that in all the works mentioned a complete agreement between calculated and measured soil moisture data was never observed.

A few works have been cited, and many others exist in the soil literature, that indicate the usefulness of eq. 3 to describe moisture flow in unsaturated soils. Yet eq. 3 has been used only sporadically in hydrologic studies. The main reason herefore is the difficulty in specifying the boundary condition at the soil surface under field conditions.

THE BOUNDARY CONDITION AT THE SOIL SURFACE

A necessity for any solution of the unsaturated soil moisture flow equation is that the boundary condition at the soil surface can be specified. Under field conditions this is usually extremely difficult. Neither the hydraulic head, nor the soil moisture content at the soil surface, nor the upward flux through the soil surface are directly measurable. Moreover these quantities change rapidly with time. To overcome the experimental difficulties, it has been tried to relate the upward flux through the soil surface to atmospheric quantities that are more easily measured than soil properties near the soil surface.

Frequently used in this context is the concept of potential evaporation, an upper limit of evaporation in case the soil surface is thoroughly wet. Well-known expressions for the potential evaporation are given by Penman (1948) and by Van Bavel (1966). The equation by Van Bavel (1966), which is a modified Penman equation, reads:

$$E_p = \frac{(\Delta/\gamma)\,R_n + B\,(e_s - e)}{\Delta/\gamma + 1}, \qquad (5)$$

in which expression E_p stands for the potential evaporation rate, R_n for the net radiant energy available for evaporation, B represents a ventilation vector depending on the wind velocity and the crop height, $e_s - e$ is the vapor pressure deficit of the air and Δ/γ a coefficient depending on the air temperature. In the work of Van Bavel (1966) lysimeter data are shown that support the validity of eq. 5. Another appealing aspect of eq. 5 is that is does not contain empirical quantities. However, the expression only yields an upper limit for the evaporation rate; about the actual evaporation rate E, that occurs when the soil is drying, eq. 5 does not provide information.

Various expressions have been propose for the actual evaporation rate when the water supply of the soil is limited. Usually such expressions have an empirical nature and are suited for regional use only. In the United States the Blaney-Criddle relationship, see Eagleson (1970) is widely used, whereas in Germany the Haude equation (Haude, 1955) frequently is applied to estimate the actual evaporation rate. Others have tried to relate the Penman equation or eq. 5 with some property of the soil surface to estimate reduced evaporation rates. Such attempts can be found for example in Staple (1974), in Jackson et al. (1976) or in Beese et al. (1977).

In the just cited works only evaporation from a bare soil is considered. Fig. 2 is taken from Beese et al.(1977), and

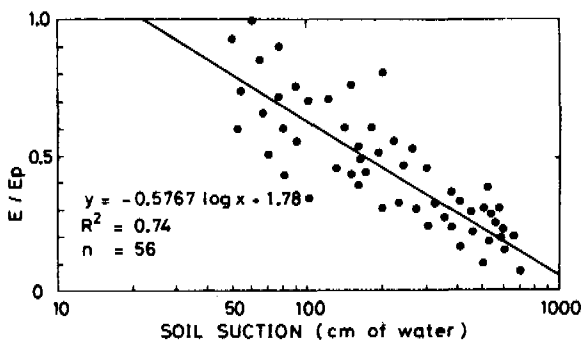

Fig. 2. Relation between actual/ potential evaporation and soil suction head at the soil surface (Beese et al., 1977)

shows a relation between the ratio of actual and potential evaporation and the suction head (the main component of the hydraulic head) at the soil surface. The relation was determined with a weighing soil monolith, and will be referred to later on in this paper.

In a similar way as the potential evaporation from a bare soil is reduced, attempts have been made to reduce the transpiration rate from a cropped soil as the soil water becomes depleted. Nimah and Hanks (1973a,b) Saxton et al. (1974), Neuman et al. (1975), Feddes et al. (1975) and Van der Ploeg et al. (1978) discuss possibilities to achieve such a reduction. The reduction is accomplished by relating the potential evaporation with the root distribution of the crop and the moisture status of the soil in the root zone. A most simple way to account for the dryness of the soil in the root zone has been proposed by Feddes et al. (1976). In slightly modified form this relation is shown in fig. 3. The

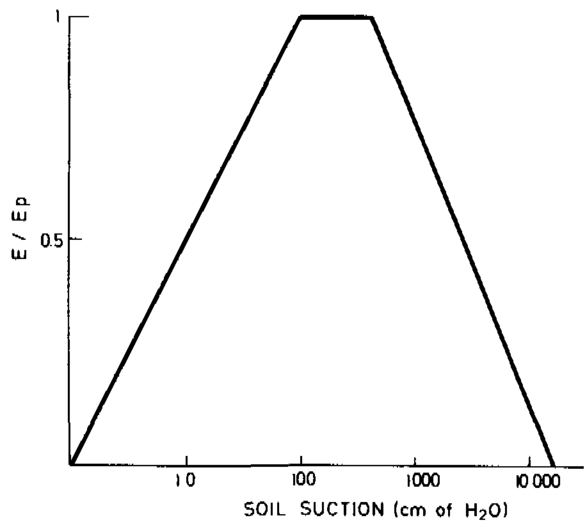

Fig. 3. Relation between actual/ potential transpiration and soil suction in the root zone (Feddes et al., 1976, modified)

ratio of actual and potential evapotranspiration as function of the suction head in the root zone is shown. It is remarked that all the reduction procedures are rather speculative, and additional research is needed for a better insight in this phenomenon. In the next section results of model calculations will be presented that were obtained with use of reduction procedures as just described.

SOME ONE- AND TWO-DIMENSIONAL MODEL RESULTS

Fig. 4 shows results published by Van der Ploeg and Benecke

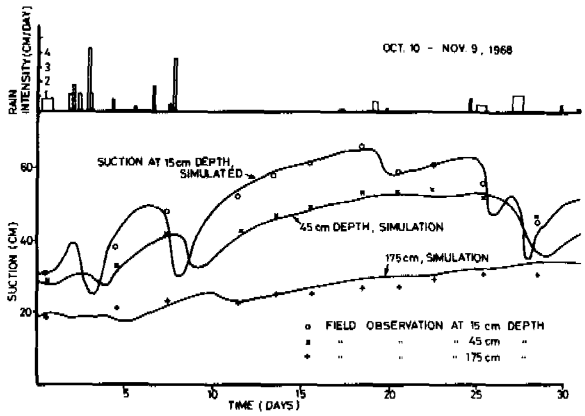

Fig. 4. Calculated and measured suction values in a field soil in a period without evapotranspiration (Van der Ploeg and Benecke, 1974a)

(1974a). Shown are calculated and measured suction heads in a layered forest soil over a one-month period. The flat soil is located in a plateau-like mountanous area. During the period under consideration evapotranspiration was neglected. A numerical solution to eq. 3 was obtained with a method as described by Van der Ploeg (1974). By assuming zero evapotranspiration the boundary condition at the soil surface could be stated in terms of the occurring precipitation only. The occurrence and intensity of precipitation are shown as an insert in fig. 4.Fig.4 shows a good agreement between calculated and measured suction values in three different depths.

Fig. 5 shows similar results. Now however evaporation from the soil surface is taken into account. Fig. 5 is taken from the work of Beese et al. (1977), who used the relation shown in fig. 2 to estimate the actual daily evaporation rate from a bare loess soil. From the good agreement between measured and calculated suction heads, Beese et al.(1977) infer that the evaporation and the deep seepage as calculated by their model, also must be close to the actually occurring amounts, that cannot be measured directly.

Results from another one-dimensional model are given in fig. 6. Shown are measured and

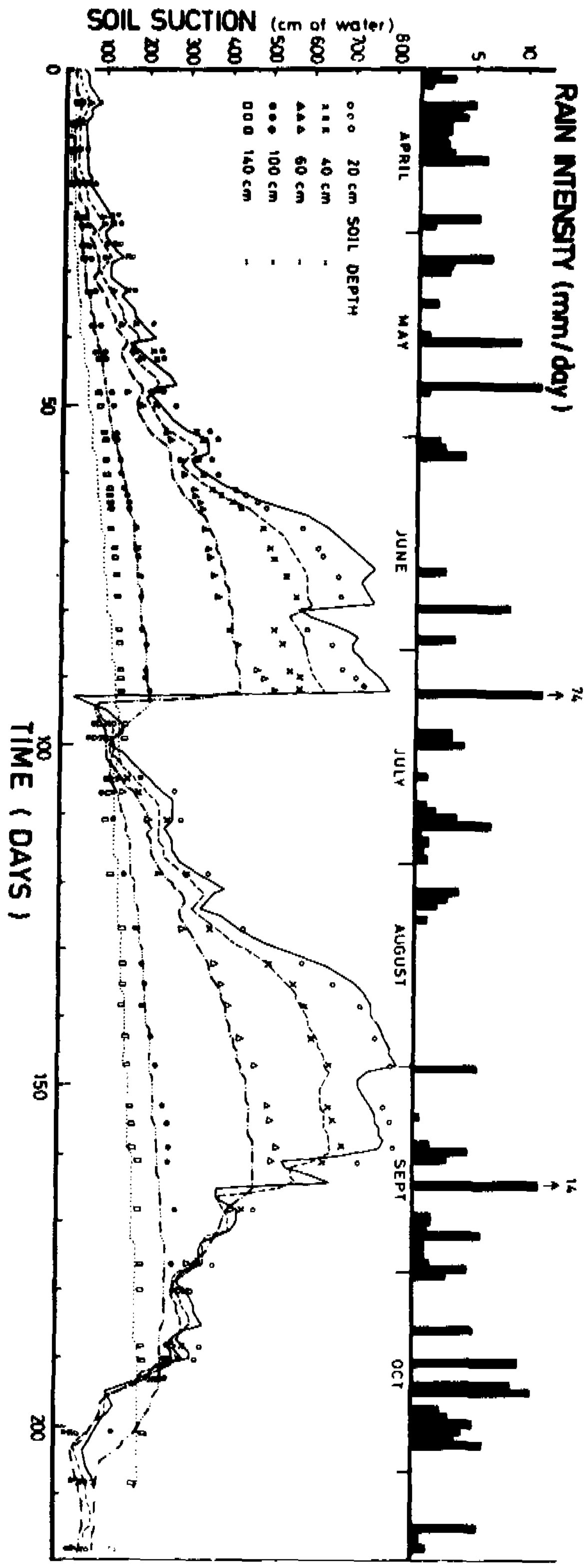

Fig. 5. Calculated and measured suction values in a bare field soil (Beese et al., (1977)

calculated soil moisture data of a sugar beet crop, between May 4 and Oct. 13, 1975. The curves in fig. 6 denote the calculated data.

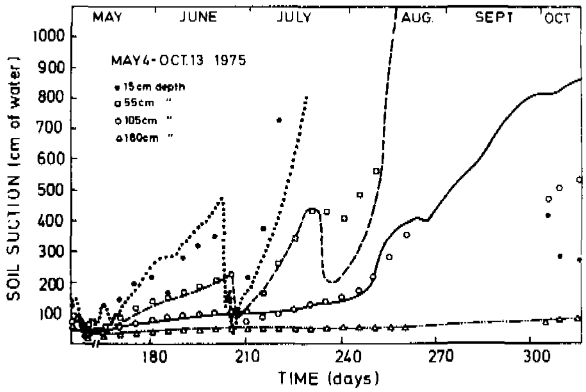

Fig. 6. Calculated and measured suction values in soil under a sugar beet crop (Van der Ploeg et al., 1978)

The figure is taken from Van der Ploeg et al. (1978). Especially early in the season, evaporation as well as transpiration must be considered. Instead of eq. 3 now eq. 4 is used to describe the moisture flow in the soil. The potential evapotranspiration for fig. 6 was calculated with eq. 5. A distinction between potential evaporation (from the soil surface) and potential transpiration (from the crop) was accomplished by use of data published by Ritchie (1972) for row crops. In Van der Ploeg et al. (1978) computer results and field measurement for the same soil pertaining to the period Dec. 1, 1974 - May 4, 1975 are also shown. During that period the soil was bare. Table 1 is also taken from the work of Van der Ploeg et al. (1978). The data shown in Table 1 reflect the entire period of study, Dec. 1, 1974 - Oct. 13, 1975. Monthly values in mm for the precipitation P, the interception I, the evaporation E, the transpiration T, the seepage S at a depth of 170 cm, and the change of storage in the profile, R are shown. Only the precipitation P was measured, the other components were calculated. The data of Table 1 compare well with results that were experimentally derived, see Van der Ploeg et al. (1978).

	P	I	E	T	S	R
December	91		19		4	68
January	34		16		43	-25
February	6		8		8	-10
March	54		23		5	26
April	45		37		30	-22
May	64	0	46	7	22	-11
June	62	2	20	35	6	-1
July	71	6	9	81	1	-26
August	24	5	2	103	0	-86
September	52	11	0	44	0	-3
October (12 days)	16	4	0	25	0	-13
Total	519	28	180	295	119	-103

Table 1. Monthly values in mm of the components of the water balance equation of an agricultural field (Van der Ploeg et al., 1978)

All the models considered so far were dealing with one-dimensional moisture flow. Van der Ploeg and Liebscher (1978) also used the unsaturated soil moisture flow equation in a watershed study, where two-dimensional unsaturated flow had to be considered. For calculation purposes the elongated watershed, overlying impervious bedrock material, was subdivided into two representative slopes. The slopes were of unequal steepness. The 76-ha large watershed was completely covered with a spruce stand. No deep seepage was allowed; only lateral subsurface flow was considered. The subsurface lateral flow is collected in a small creek, which drains the watershed. The discharge of the creek was recorded at a weir. Schematically the watershed flow configuration is shown in fig. 7, which is derived

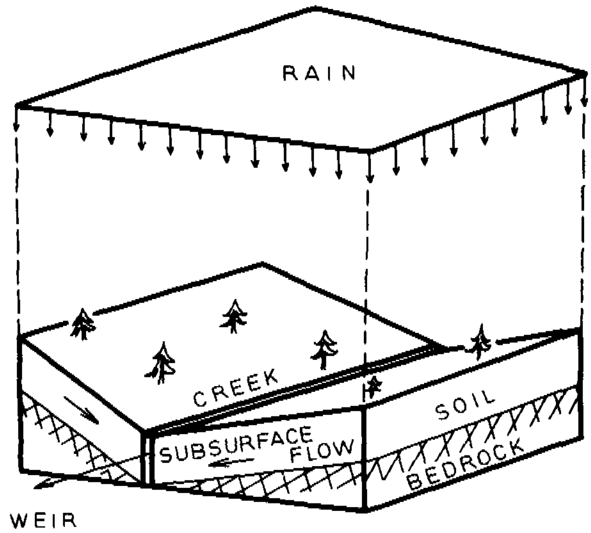

Fig. 7. Schematic representation of an elongated watershed (after Eagleson, 1970)

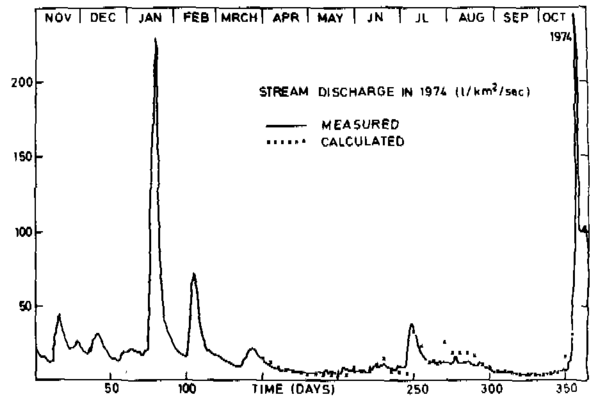

Fig. 8. Measured and calculated stream discharge data from a small watershed (Van der Ploeg and Liebscher, 1978)

from Eagleson (1970). Also described in Eagleson (1970, chapter 15) is a model that describes water flow in small streams. This model was, slightly modified, used in the present study. The lateral subsurface flow in the soil was calculated with a two-dimensional space equivalent of eq. 4. The potential evapotranspiration was reduced to actual evapotranspiration with use of fig. 3. Details of the numerical procedure are given by Van der Ploeg and Benecke (1974b).

For periods without a snow cover (which is roughly speaking between April and November every year) model calculations were carried out to calculate the stream flow in the creek. Also the actual transpiration for each of the two slopes was calculated. For three consecutive years simulations were performed. In fig. 8 results for 1974 are shown. During the period April 1 - October 21 of fig. 8 the cumulative amount of rain per m^2 was 569 mm. An equivalent amount of 143 mm discharge was recorded at the weir during the same period, whereas the model yielded 154 mm.

CONCLUSIONS

Some general features of one- and two-dimensional soil moisture flow models have been discussed. It has been pointed out that the soil moisture flow equation appears to describe moisture flow in many soils rather accurately. Use of the equation in field studies is limited because of the difficulty to specify the boundary condition at the soil surface. Four model studies were presented in which the soil moisture flow equation constituted a central part. In each of the four examples it was indicated how the boundary condition at the soil surface was handled. The shown results suggest the usefulness of such models in studies in which data about amounts of seepage water and about flow velocities through soil are needed. Such data can subsequently be utilized in studies about the movement of agricultural or industrial chemicals through soil. The four sample models that were presented may be considered as a stimulus to utilize soil hydrologic models in handling environmental problems.

ACKNOWLEDGMENT

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LAND TREATMENT MATHEMATICAL MODELING

EVALUATION OF THE MOVING BOUNDARY THEORY IN DARCY'S FLOW THROUGH POROUS MEDIA

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ABSTRACT

Traditionally in hydrology and soil physics neither the water table nor the wetting front in Darcy's flow were believed to be singular surfaces. Recently a new and conflicting theory has been advanced using two different approaches. It has been shown, based upon continuum physics, that across both the water table and the wetting front local acceleration generally suffers a non-zero jump, and these two boundaries can be interpreted as acceleration waves. This interpretation was found consistent with reported regularity results obtained from a purely mathematical viewpoint.

INTRODUCTION

At any given time a porous medium may be uniquely divided into several parts according to three kinds of flow conditions that are mutually exclusive with respect to the specific incompressible fluid under consideration. If all the voids in a part of the medium are completely filled with this fluid we refer to it as saturated; if not we refer to it as unsaturated. There is another situation where the content of fluid in a part of the medium is not high enough to induce flow; we refer to this as stagnant.

Among the three kinds of interfaces between the three kinds of parts two kinds of interfaces are of practical importance. In hydrology an interface between a saturated and an unsaturated part is called a water table, and an interface between an unsaturated and a stagnant part is called a wetting front.

For the case in which the fluid under consideration is compressible, the distinction between a saturated and an unsaturated part is unnecessary. Only one kind of interface is of practical importance, that is an interface between a stagnant part and one in which flow takes place. For the sake of convenience, we also refer to such an interface as a wetting front.

Traditionally in hydrology and soil physics neither the water table nor the wetting front appearing in porous media described by Darcy's law were believed to be singular surfaces [17,19]. However, in recent years a conflicting theory, that these surfaces were essentially singular, was advanced using two different approaches. One of them was based upon the theory of singular surfaces in continuum physics [11,12,13]. It was proven that both the water table and wetting front can be interpreted as singular surfaces of order 2, i.e. propagating acceleration waves.

Another approach was based upon pure mathematical theory in functional analysis. A special class of wetting front problems attracted mathematicians [9,14] and its theory is now fairly well advanced [1,2, 5,6]. For this class of problems it was established that singularities occur in their solution.

It has been recognized for some time that similarity solutions may be limiting solutions, as time approaches infinity, of more general solutions of parabolic equations. The first proof of this observation was given in theory of boundary layers by Serrin [21]. Recently similar results were proved for a special class of wetting front problems [15]. Traditionally in hydrology and soil physics the similarity solution was assumed to be the general solution of the wetting front problem [16]. This assumption is contradictory to mathematical theory [4].

In this paper we examine a new theory for both the water table and the wetting front. In the following section we present the theory of acceleration waves applied to Darcy's flow. The theory of singular surfaces and acceleration waves originated in the work of Christoffel [3] and Hugoniot [8] and was extended in a treatise by Truesdell and Toupin [22]. In recent years the acceleration wave has been studied in many branches of continuum physics [10].

ACCELERATION WAVES

We consider a porous medium that consists of rigid solids and pores. We assume that the pores are uniformly distributed throughout the medium. Let the pores contain a mixture of two bodies $\underset{a}{B}$, where a = 1, 2, and $\underset{1}{B}$ and $\underset{2}{B}$ represent an incompressible fluid and a compressible fluid, respectively. Let $\underset{a}{d}$ represent the mass density of the a^{th} constituent per unit volume of the mixture and $\underset{a}{\rho}$ represent the mass of the a^{th} constituent per unit volume of the mixture. Then $\underset{a}{\theta} = \underset{a}{\rho}/\underset{a}{d}$ will be the volume fraction of the a^{th} constituent.

Let $\underset{a}{X}$ be the place occupied by a particle of $\underset{a}{B}$ in some reference configuration. The motion of $\underset{a}{B}$ is the mapping of $\underset{a}{B}$ onto a time-sequence of configuration in space.

$$x = \underset{a}{x}\,(\underset{a}{X}, t) \tag{2.1}$$

Let a backward prime denote the time derivative when $\underset{a}{X}$ is held constant, i.e., the time derivative that is material with respect to $\underset{a}{B}$. The velocity $\underset{a}{\grave{x}}$ and the acceleration $\underset{a}{\grave{\grave{x}}}$ are defined as

$$\underset{a}{\grave{x}} = \frac{\partial}{\partial t}\,\underset{a}{x}\,(\underset{a}{X}, t) \tag{2.2}$$

$$\underset{a}{\grave{\grave{x}}} = \frac{\partial^2}{\partial t^2}\,\underset{a}{x}\,(\underset{a}{X}, t) \tag{2.3}$$

The deformation gradient $x^k_{a,\alpha}$ of the a^{th} constituent is defined by

$$x^k_{a,\alpha} = \frac{\partial}{\partial \underset{a}{X}^{\alpha}}\,\underset{a}{x}^k(\underset{a}{X}, t) \tag{2.4}$$

The vorticity vector $\underset{a}{w}$ is defined as

$$\underset{a}{w} = \text{curl}\ \underset{a}{\grave{x}} \tag{2.5}$$

The spatial equation of continuity for the a^{th} constituent is given as

$$\partial\underset{a}{\rho}/\partial t + \text{div.}\,(\underset{a}{\rho}\underset{a}{\grave{x}}) = 0 \tag{2.6}$$

Suppose there exists an acceleration wave $\sigma(t)$ of the a^{th} constituent.

Then let the surface $\sigma(t)$ be represented as

$$f\,(\underset{a}{x}, t) = 0 \tag{2.7}$$

An alternative representation of $\sigma(t)$ is

$$F\,(\underset{a}{X}, t) = 0 \tag{2.8}$$

By the use of standard notation [22] the general jump conditions for acceleration waves of the fluid phase across $\sigma(t)$ are given as

$$[\grave{x}^k_{a,\alpha}] = -\,U_N\,s^k N_\alpha \tag{2.9}$$

$$[\underset{a}{\grave{\grave{x}}}^k] = U_N^2\,s^k \tag{2.10}$$

where $s^k = [N^\gamma N^\delta\, x^k_{a,\gamma\delta}]$

$[A] = A^+ - A^-$

U_N = speed of propagation

N = unit normal to the surface defined by (2.8)

If we choose the material coordinates as being equal to the spatial coordinates at the instant the wave passes, then (2.9) and (2.10) become [22]

$$[\grave{x}^k_{a,m}] = U\,s^k_o\,n_m \tag{2.11}$$

$$[\underset{a}{\grave{\grave{x}}}^k] = U^2\,s^k_o \tag{2.12}$$

where

$$s_o^k = \frac{U_N}{U} s^k \frac{\sqrt{f_{,p}\, f^{,p}}}{\sqrt{F_{,\beta}\, F^{,\beta}}}$$

$$U = u_n - \underset{a}{\grave{x}}_n$$

u = speed of the surface defined by (2.7)

n = unit normal to the surface defined by (2.7)

$$\underset{a}{\grave{x}}_n = \underset{a}{\grave{x}} \cdot n$$

From (2.11) and (2.12) we obtain

$$\left[\frac{\partial}{\partial t}\ \underset{a}{\grave{x}}_n\right] = -\, u_n\, [\operatorname{div} \underset{a}{\grave{x}}] \tag{2.13}$$

$$[\operatorname{div} \underset{a}{\grave{x}}] = -\, U\, s_o \cdot n \tag{2.14}$$

$$[\underset{a}{w}] = [\operatorname{curl} \underset{a}{\grave{x}}] = -\, U\, s_o \times n \tag{2.15}$$

Interpretation of these identities yields Hadamard's theorem: longitudinal acceleration wave carries a jump in expansion but vorticity is unchanged, while a transverse acceleration wave carries a jump in vorticity but does not affect expansion [22].

We further examine a longitudinal acceleration wave. It is necessary to examine the jump condition for $\underset{a}{\grave{x}}_n$, because if $\underset{a}{\grave{x}}_n$ suffers a non-zero jump across $\sigma(t)$, (2.13) no longer represents an acceleration wave. The balance of fluid mass across $\sigma(t)$ yields

$$[\underset{a}{\rho}\, (\underset{a}{\grave{x}}_n - u_n)] = 0 \tag{2.16}$$

or

$$[\underset{a}{\rho}\, \underset{a}{\grave{x}}_n] = u_n\, [\underset{a}{\rho}] \tag{2.17}$$

(2.17) includes the following five cases:

Case 1 The fluid is incompressible and $[\underset{a}{\theta}] = 0$

Case 2 The fluid is incompressible and $u_n = \underset{a}{\grave{x}}{}^{+}_n = \underset{a}{\grave{x}}{}^{-}_n$

Case 3 The fluid is compressible and $[\underset{a}{d}] = 0$

Case 4 The fluid is compressible, $[\underset{a}{d}] \neq 0$, and $[\underset{a}{d}\, \underset{a}{\grave{x}}_n] = 0$

Case 5 The fluid is compressible, $[\underset{a}{d}] \neq 0$, and $[\underset{a}{d}\, \underset{a}{\grave{x}}_n] \neq 0$

For the incompressible fluid, if the condition of either Case 1 or Case 2 is not met, then the velocity suffers a non-zero jump and $\sigma(t)$ becomes a shock wave, which contradicts the assumption that the fluid is incompressible.

It is easy to see that in Cases 1, 2 and 3 $[\underset{a}{\grave{x}}_n] = 0$; therefore $\sigma(t)$ is an acceleration wave. In Case 4 $u_n = 0$ and the wave is stationary. Case 5 merits further examination. In this case the jump of $\underset{a}{\grave{x}}_n$ may be written as

$$[\underset{a}{\grave{x}}_n] = \frac{1}{\underset{a}{d}^{-}}\, [\underset{a}{d}]\, (u_n - \underset{a}{\grave{x}}{}^{+}_n) \tag{2.18}$$

(2.18) implies that if $u_n = \underset{a}{\grave{x}}{}^{+}_n$, then $[\underset{a}{\grave{x}}_n] = 0$; however, if $u_n \neq \underset{a}{\grave{x}}{}^{+}_n$, then $[\underset{a}{\grave{x}}_n] \neq 0$ and $\sigma(t)$ possibly becomes a shock wave rather than an acceleration wave. We examine the latter case. The jump condition of momentum across $\sigma(t)$ is given as

$$[\underset{a}{d}\, \underset{a}{\grave{x}}_n\, (\underset{a}{\grave{x}}_n - u_n)] = 0 \tag{2.19}$$

From (2.16) and (2.19) we obtain

$$(u_n - \underset{a}{\grave{x}}{}^{+}_n)\, \{[\underset{a}{\grave{x}}_n]\, (u_n + \underset{a}{\grave{x}}{}^{+}_n) - [\underset{a}{\grave{x}}{}^{2}_n]\} = 0 \tag{2.20}$$

Since $u_n \neq \underset{a}{\grave{x}}{}^{+}_n$ in this case, from (2.20) we obtain

$$u_n = [\underset{a}{\grave{x}}{}^{2}_n]/[\underset{a}{\grave{x}}_n] - \underset{a}{\grave{x}}{}^{+}_n \tag{2.21}$$

The density of the fluid generally suffers a non-zero jump across $\sigma(t)$ and the jump condition is given as

$$[\underset{a}{d}]/\underset{a}{d}^{-} = [\underset{a}{\grave{x}}_n]^2/([\underset{a}{\grave{x}}{}^{2}_n] - 2\underset{a}{\grave{x}}{}^{+}_n\, [\underset{a}{\grave{x}}_n]) \tag{2.22}$$

In the following sections we apply the theory of acceleration waves to three specific cases which are important for practical applications.

WATER TABLE OF AN INCOMPRESSIBLE FLUID

We consider a porous medium that contains an incompressible fluid B and air B. By the use of a Cartesian coordinate frame (x_i, i = 1, 2, 3) that is attached to the solid phase with the x_3-axis being coincident with the direction of gravity, Darcy's law for saturated flow is given [19] as

$$\dot{\underset{1}{x}} = - (K/\theta_s) \text{ grad } (\underset{1}{h} - \underset{1}{x_3}), \ \underset{1}{h} \geq 0, \quad (3.1)$$

where K = saturated hydraulic conductivity that is constant

$\theta_s = \underset{1}{\theta}\ (\underset{1}{h};\ \underset{1}{h} \geq 0)$ = constant

$\underset{1}{h} = (\underset{1}{p} - p_o)/\underset{1}{d}g$ = pressure head

$\underset{1}{p}$ = pressure of fluid

p_o = atmospheric pressure

g = gravitational acceleration

From (2.6) and (3.1) the spatial differential form of fluid mass balance in saturated flow is given as

$$\text{div } \dot{\underset{1}{x}} = 0 \quad (3.2)$$

Darcy's law for unsaturated flow is given [19] as

$$\dot{\underset{1}{x}} = - (k(\underset{1}{h})/\underset{1}{\theta}(\underset{1}{h})) \text{ grad } (\underset{1}{h} - \underset{1}{x_3}), \ \underset{1}{h} \leq 0, \quad (3.3)$$

where $k(\underset{1}{h})$ = unsaturated hydraulic conductivity and k(0) = K

The function $k(\underset{1}{h})$ is usually determined empirically. We assume that $k(\underset{1}{h})$ is a smooth function for $\underset{1}{h} < 0$.

From (2.6) and (3.3) the spatial differential form of fluid mass balance in unsaturated flow is given as

$$\frac{\partial}{\partial t} \underset{1}{\theta} = - \text{div } (\dot{\underset{1}{x}}\ \underset{1}{\theta}) \quad (3.4)$$

We assume that θ is a function of the pressure head $\underset{1}{h}$ in unsaturated flow

$$\underset{1}{\theta} = h\ (\underset{1}{h}) \quad (3.5)$$

and the unique inverse h^{-1} is assumed to exist.

We assume that the porous medium consists of a saturated part V^-, in which (3.1) holds true, and an unsaturated part V^+, in which (3.3) and (3.5) hold true.

In the past, several works were reported on the nature of flow described by (3.1) [20], and it is well established that (3.1) accurately describes saturated flow of a Newtonian fluid in a rigid porous medium, if the inertial terms are negligibly small, because (3.1) completely neglects the inertial terms. Neglecting the inertial terms is well justified because they are indeed small [7, 18]. However, it is important to note that the acceleration is finite regardless of how small it might be. This small but finite acceleration is supposed to be the limit of the acceleration as $\sigma(t)$ is approached upon paths interior to V^+. Since in V^- the acceleration is absent while in V^+ the acceleration is present, the discontinuity of the local acceleration across $\sigma(t)$ is generally inevitable. Therefore, the water table of an incompressible fluid belongs to either Case 1 or Case 2 in the previous section and constitutes a longitudinal acceleration wave.

Next we examine whether or not $\sigma(t)$ constitutes a transverse acceleration wave. We compute the jump of the vorticity vector $\underset{1}{\mathbf{w}}$ across $\sigma(t)$.

From (3.1) we obtain

$$\underset{1}{\mathbf{w}} = 0 \quad \text{in } V^- \quad (3.6)$$

and from (3.3) we obtain

$$\underset{1}{\mathbf{w}} = \begin{pmatrix} \beta \dot{\underset{1}{x_2}} \\ -\beta \dot{\underset{1}{x_1}} \\ 0 \end{pmatrix} \quad \text{in } V^+ \quad (3.7)$$

where $\beta = (\underset{1}{\theta}/k) \frac{d}{d\underset{1}{h}} (k/\underset{1}{\theta})$ and the assumption that $\partial^2 \underset{1}{h}/(\partial x_i \partial x_j) = \partial^2 \underset{1}{h}/(\partial x_j \partial x_i)$ was used in the derivation of (3.7). (3.6), (3.7) and (2.15) yield

$$[w]_1 = \begin{pmatrix} \beta^+ \grave{x}^+_{12} \\ -\beta^+ \grave{x}^+_{11} \\ 0 \end{pmatrix} = -U s_o \times n \tag{3.8}$$

where β^+ is the limiting value of β if it exists when $\sigma(t)$ is approached upon paths interior to V^+. (3.8) implies that, if $\beta^+ \neq 0$, the vorticity vector can be discontinuous across $\sigma(t)$ and a transverse acceleration wave propagates. Since β^+ depends solely upon the material property of the medium, it is not certain that the boundary $\sigma(t)$ can be interpreted as a propagating transverse acceleration wave.

WETTING FRONT OF AN INCOMPRESSIBLE FLUID

We again consider a porous medium that contains an incompressible fluid B_1 and air B_2. We assume that the porous medium consists of an unsaturated part V^+, in which (3.3) and (3.5) hold true, and a stagnant part V^-, in which the following conditions hold true:

$$\theta_1 = \theta_o \text{ (constant)} \tag{4.1}$$

$$h_1 = h_o \text{ (constant)} \tag{4.2}$$

$$\grave{x}_1 = 0 \tag{4.3}$$

In the physical problem, it is clear that the interface $\sigma(t)$ between V^+ and V^- should move with a finite speed. When $\sigma(t)$ is approached upon paths interior to V^+, the limiting material time derivative of the velocity is given as

$$(D\grave{x}_{1n}/Dt)^+ = (\partial \grave{x}_{1n}/\partial t)^+ + \grave{x}^+_{1n} \,(\text{grad } (\grave{x}_{1n}))^+ \tag{4.4}$$

If the local acceleration $(\partial \grave{x}_{1n}/\partial t)^+$ and the velocity $\grave{x}^+_{1n}$ vanish simultaneously, then the particles stop moving and therefore $\sigma(t)$ stops moving. Since both local acceleration and velocity vanish in V^-, a singularity of either the local acceleration or the velocity must appear so that the interface σ moves with a finite speed. On the other hand, if the velocity suffers a non-zero jump across σ, σ becomes a shock wave. Since in this case the fluid is incompressible and does not admit a shock wave, the local acceleration must suffer a non-zero jump across σ, and σ becomes an acceleration wave. (2.13) is reduced to

$$\left(\frac{\partial}{\partial t} \grave{x}_{1n}\right)^+ = -u_n \,(\text{div } \grave{x}_1)^+ \tag{4.5}$$

and either one of the following two conditions, which correspond to Case 1 and Case 2 in the general theory, must be satisfied:

$$\theta^+_1 = \theta_o, \; \grave{x}^+_{1n} = 0 \tag{4.6}$$

$$u_n = \grave{x}^+_{1n} = 0 \tag{4.7}$$

The wetting front of an incompressible fluid can be generally interpreted as an acceleration wave. On the front the volume fraction and the velocity of the fluid are continuous, however the local acceleration of the fluid is discontinuous.

We must now examine whether or not the above conclusion is consistent with the regularity results derived from purely mathematical analysis. Gilding [6] and Gilding and Peletier [5] studied the Cauchy problem for a special case of the present situation, in which (3.3) takes the form

$$\theta_1 \grave{x}_1 = -D_o \,\theta_1^{m-1} \,(\partial \theta_1/\partial x_{13}) + K_o \,\theta_1^n \tag{4.8}$$

After suitable rescaling of the independent variable the equation of continuity (2.6) is written as

$$(\partial\theta/\partial t) = \partial^2\theta^m/\partial x^2 + \partial\theta^n/\partial x \tag{4.9}$$

where $\theta = \theta_1 \left(\frac{D_o}{m}\right)^{1/m} = \theta_1 K_o^{1/n}$

$$x = -x_{13}$$

and (4.8) reduces to

$$\grave{x} = -\frac{m}{m-1} \frac{\partial \theta^{m-1}}{\partial x} - \theta^{n-1} \tag{4.10}$$

We consider the Cauchy problem for (4.9). Let S_T denote the strip $(-\infty, \infty) \times (0, T]$ in the x, t-plane, for some fixed number $T>0$, and let L denote the non-linear operator

$$L\theta = (\theta^m)_{xx} + (\theta^n)_x - \theta_t$$

Then we consider the problem

$$L\theta = 0 \qquad \text{in } S_T \tag{4.11}$$

$$\theta(x, 0) = \theta_i(x) \qquad -\infty < x < \infty \tag{4.12}$$

in which θ_i is a given non-negative, bounded and continuous function in $(-\infty, \infty)$.

It was shown by Gilding and Peletier [5] that problem (4.11) and (4.12) need not have a classical solution, and because of this the notion of a weak solution was there introduced. They show that if θ_i^m is uniformly Lipschitz continuous and $m > 1$ and $n \geq \frac{1}{2}(m+1)$, then

(a) Problem (4.11), (4.12) has a unique weak solution.

(b) θ satisfies a Hölder condition in $\bar{S}_T$.

(c) θ^m has a continuous derivative with respect to x in S_T.

(d) θ^{m-1} has a bounded generalized derivative with respect to x in any strip of the form $(-\infty, \infty) \times [\tau, T]$, $\tau \varepsilon (0, T)$.

(e) θ_x exists and is continuous in S_T if $m < 2$.

(f) θ is infinitely continuously differentiable and a classical solution of (4.11) in a neighborhood of any point in S_T where it is positive.

The above regularity results may be interpreted as follows. In order to avoid unnecessary complication, we consider a situation in which only one interface $\sigma(t)$ exists. We refer to the part where $\theta > 0$ as V^+ and to the part where $\theta = 0$ as V^-. We exclude the case where $2 > m > 1$ because it is not physically relevant. From (b) and (c) we get

$$\theta^+ = 0 \tag{4.13}$$

and

$$(\theta \dot{x})^+ = 0 \tag{4.14}$$

where the + sign denotes the limiting value when σ is approached upon paths interior to V^+. (4.13) and (4.14) do not necessarily imply that $\dot{x}^+$ exists. If $\dot{x}^+$ exists, then $\dot{x}^+ = 0$. Since $\dot{x}^+ \neq 0$ implies that the fluid is compressible, we consider here only the case where $\dot{x}^+ = 0$ and postpone our discussion on a compressible fluid to the next section. (d) and (f) imply that θ_x^{m-1} and θ_x exist and are continuous everywhere except on σ. The reason why these two derivatives are generally singular on σ merits further discussion. The singularity of θ_x^{m-1} is directly related to the singularity of $\dot{x}$. According to the present theory for an incompressible fluid $\dot{x}$ should be continuous so is θ_x^{m-1}. However, since problem (4.11), (4.12) does not include the condition that the fluid is incompressible, it is expected that θ_x^{m-1} is generally singular on σ.

Suppose that σ is an acceleration wave, then θ_x can be shown to be generally singular on σ as follows. If σ is an acceleration wave, then from (4.5) a limiting value $(\operatorname{div} \dot{x})^+$ exists and is not zero. From (4.10) we get

$$\operatorname{div} x = -\frac{m}{m-1}\frac{\partial^2\theta^{m-1}}{\partial x^2} - \frac{\partial\theta^{n-1}}{\partial x}$$

$$= -[m\,(m-2)\,\theta^{m-3}\frac{\partial\theta}{\partial x} + (n-1)\,\theta^{n-2}]\frac{\partial\theta}{\partial x} - m\,\theta^{m-2}\frac{\partial^2\theta}{\partial x^2} \qquad x \,\varepsilon\, V^+ \tag{4.15}$$

From (4.15) if $m > 3$, and if both θ_x and θ_{xx} are continuous on σ, then the limiting value $(\operatorname{div} \dot{x})^+$ becomes zero. Therefore either θ_x or θ_{xx} must be discontinuous on σ. This shows that even if σ is an acceleration wave, it is possible that θ_x is discontinuous on σ. Next we examine the condition for θ_x to be discontinuous on σ.

The time derivative of θ with respect to the surface that is moving with a speed u_n is given as

$$\frac{D\theta}{Dt} = \frac{\partial\theta}{\partial t} + u_n\frac{\partial\theta}{\partial x}$$

$$= \frac{\partial^2\theta^m}{\partial x^2} + \frac{\partial\theta^n}{\partial x} + u_n\frac{\partial\theta}{\partial x}\,, \qquad x \,\varepsilon\, V^+ \tag{4.16}$$

From (4.10) and (4.16) we get

$$\frac{D\theta}{Dt} = [u_n - \dot{x} + \frac{m(m-2)}{m-1}\frac{\partial\theta^{m-1}}{\partial x} + (n-1)\,\theta^{n-1}]\frac{\partial\theta}{\partial x} - m\,\theta^{m-1}\frac{\partial^2\theta}{\partial x^2} \tag{4.17}$$

If the limiting values of all terms of (4.17) exist, accounting $\dot{x}^{+}=0$, and $(\theta^{m-1}{}_{x})^{+} = 0$, we get

$$(\frac{D\theta}{Dt})^{+} = u_n\ (\frac{\partial\theta}{\partial x})^{+} - m(\theta^{m-1}\ \frac{\partial^2\theta}{\partial x^2})^{+} \qquad (4.18)$$

Since always $\theta^{+} = 0$, so the left hand side of (4.18) should be zero. Therefore we conclude that if $(\theta^{m-1}\theta_{xx})^{+} = 0$, then θ_x is continuous. If not, θ_x is discontinuous on σ.

Unfortunately the existence of $(\mathrm{div}\ \dot{x})^{+}$ does not directly follow the regularity results by Gilding and Peletier [5]. So far, the present theory does not contradict the regularity results. Therefore, the regularity results neither negate nor confirm the existence of an acceleration wave.

WETTING FRONT OF A COMPRESSIBLE FLUID

We consider a porous medium that contains only one compressible fluid B. We assume that the porous medium $\underset{2}{}$ consists of a part V^{+}, in which the pressure $\underset{2}{h}$ of the fluid is greater than a non-negative number h_o and a stagnant part V^{-}, in which the following conditions hold true:

$$\underset{2}{h} = h_o \qquad (5.1)$$

$$\underset{2}{\dot{x}} = 0 \qquad (5.2)$$

We refer to an interface between V^{+} and V^{-} as σ and assume that the velocity is given as

$$\underset{2}{\dot{x}} = -\ k(\underset{2}{h})\ \mathrm{grad}\ (\underset{2}{h} - \underset{2}{x_3}),\quad \underset{2}{h} > h_c \qquad (5.3)$$

$$= 0 \qquad\qquad 0 \le \underset{2}{h} \le h_c \qquad (5.4)'$$

From (2.6) and (5.3), the spatial equation of continuity for the fluid in V^{+} is given as

$$\partial \underset{2}{d}/\partial t = \mathrm{div}\ (\underset{2}{d}\ \underset{2}{\dot{x}}) \qquad (5.5)$$

The jump conditions of local acceleration and velocity are given as

$$(\frac{\partial}{\partial t}\ \underset{2}{\dot{x}_n})^{+} = -\ u_n\ (\mathrm{div}\ \underset{2}{\dot{x}})^{+} \qquad (5.6)$$

and

$$\underset{2}{d}^{+}\ \underset{2}{\dot{x}_n}^{+} = u_n\ [\underset{2}{d}] \qquad (5.7)$$

(5.7) includes the following three cases

Case 1 $\quad \underset{2}{d}^{+} \neq 0 \quad$ and $\quad [\underset{2}{d}] = 0$

Case 2 $\quad \underset{2}{d}^{+} \neq 0 \quad$ and $\quad [\underset{2}{d}] \neq 0$

Case 3 $\quad \underset{2}{d}^{+} = 0 \quad$ and $\quad [\underset{2}{d}] = 0$

In Case 1, $\underset{2}{\dot{x}_n}$ is continuous across σ, so σ is an acceleration wave. In Case 2, $\underset{2}{\dot{x}_n}$ is discontinuous across σ, so σ is a shock wave. In Case 3, it is not certain whether or not $\underset{2}{\dot{x}_n}$ is continuous on σ. So σ can be either an acceleration wave or a shock wave. The physical description of Case 3 is that a compressible fluid is flowing into a vacuum through the porous medium. Since Aronson [1, 2] studied the Cauchy problem for a special case of this situation, we examine whether or not the above conclusion is consistent with his results.

In his problem, (5.3) takes the form

$$\dot{x} = \frac{-m}{m-1}\ \frac{\partial\theta^{m-1}}{\partial x},\ m > 2 \qquad (5.8)$$

where $\quad x = \underset{2}{x_3}$

$\quad\quad\quad \theta = \underset{2}{d}$

and (5.5) reduces to

$$\frac{\partial\theta}{\partial t} = \frac{\partial^2\theta^m}{\partial x^2} \qquad (5.9)$$

The Cauchy problem is the same as problem (4.11) and (4.12) except that the operator L is defined as

$$L\theta = (\theta^m)x,x - \theta_t \qquad (5.10)$$

Aronson showed that the regularity results (a) ∿ (f) hold true. Moreover he proved that (g) if θ_i^{m-1} is continuous. If the derivative of θ_i^{m-1} with respect to x is absolutely continuous and

$$\text{ess inf } \frac{\partial^2}{\partial x^2} \theta_i^{m-1}(x) \geq -\alpha \qquad (5.11)$$

for some constant $\alpha \geq 0$, then

$$\text{ess inf } \frac{\partial^2}{\partial x^2} \theta^{m-1}(x, t) \geq -\beta \qquad (5.12)$$

for some constant $\beta \geq 0$ and

$$\grave{x}^+(x, t) = \grave{x}(\sigma(t), t) \qquad (5.13)$$

exists for all $t \geq 0$ and

$$\grave{x}^+(u_n - \grave{x}^+) = 0 \qquad (5.14)$$

Again it is easy to find that the regularity results from (a) to (f) are not inconsistent with the present theory. Since the regularity result (g) adds more insight to the problem, we examine it.

(5.14) includes two mutually exclusive situations. If $\grave{x}^+ = 0$, then (5.14) is satisfied. This implies that σ is a shock wave. Also $u_n = \grave{x}^+$ satisfies (5.14). And this implies that σ is a shock wave. Since the problem has a unique solution, it may be interpreted that this unique solution can be either an acceleration wave or a shock wave, depending on other unknown conditions. This conclusion is indeed consistent with the present theory.

CONCLUSION

A new theory that both the water table and the wetting front in Darcy's flow are generally singular surfaces of order 2 is found theoretically sound and consistent with the reported regularity results obtained from a purely mathematical viewpoint. Darcy's law is considered an accurate but approximate description of real porous flow; however, the problems involved in either a water table or a wetting front require an extra boundary condition imposed on these surfaces due to a singularity that is inherent in the porous media described by Darcy's law.

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EVALUATION OF PHOSPHORUS MODELS FOR PREDICTION OF PERCOLATE WATER QUALITY IN LAND TREATMENT

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ABSTRACT

A review of existing literature yields several models proposed to assist a designer in estimating the concentration of phosphorus in the percolate water under land application wastewater treatment systems. The models range in approach from empirical equilibrium reactions to multistepped, hysteretic, kinetic models. The models are classified in three broad groups-- empirical models which are not based on any known theory; two phase kinetic models which assume a solution phase and some adsorbed phase; and multiphase models which include solution, adsorbed and precipitated phases.

Representative models from each of the classes are presented. It is concluded none of the models have been adequately evaluated under field conditions. The ability to predict phosphorus concentrations in percolate waters from land application systems, therefore, has not been demonstrated. Based on the models presented, an interim design approach is suggested. The design approach estimates maximum application rates to achieve a discharge requirement. A method to estimate the capacity of a soil to react with phosphorus is referenced.

INTRODUCTION

The application of wastewaters to soil-water plant systems results in the accumulation of phosphorus in soils. As residence time increases, the amount of water soluble phosphorus decreases. This observation has led some individuals (e.g. Soil Conservation Service , 1975) to the conclusion that the potential for phosphate pollution after land treatment is small. As a general rule, soils do have a large capacity to react with soluble phosphorus. Thus, land application does present an economical and effective means of treating wastewaters for soluble phosphates.

The objectives of this presentation are:
1) Present an equilibrium chemical thermodynamic model as a rationale to determine the potential for phosphate reaction in soil.
2) Review other current modeling approaches to phosphate transport through soils. Brief comments discussing advantages and disadvantages of the approaches are included.
3) Propose a preliminary approach to designing land application wastewater treatment systems to meet a percolate water quality standard.

EQUILIBRIUM THERMODYNAMICS MODEL IN SOILS

The number of phosphate species which might be found under wastewater application are large and highly pH dependent. Thermodynamics governing the transformation between the different forms is extremely complex (Larson, 1967). A careful study of chemical thermodynamics indicates trends for potential reactions.

Given sufficient time for the reaction to proceed, concentrations estimated from a chemical thermodynamic model should be quite accurate. The equilibrium state is dependent on the activity of several ions in the soil solution including phosphorus, iron, aluminum, calcium, sulfur, fluoride, and hydrogen. The activity of thermodynamic concentration is the important parameter to consider, since the activity is a measure of the effective concentration of reactants in a chemical reaction.

To understand the interrelationship between solution concentration reported in the literature and thermodynamic concentrations (activity), six thermodynamic concepts are presented.

Law of Mass Action

This law states if the reaction of b moles of B with c moles of C has equilibrated with d moles of D and e moles of E

$$bB + cC \rightleftharpoons dD + eE \qquad [1]$$

then

$$\frac{[a_D]^d\,[a_E]^e}{[a_B]^b\,[a_C]^c} = K \qquad [2]$$

where K is the thermodynamic equilibrium constant and a represents activity.

Standard State Activity

A standard state is defined for each substance in terms of reference conditions, usually 25°C and one atmosphere pressure. Under the reference conditions, the substance is assigned an activity of unity.

Free Energy of Reaction

The standard free energy change for a reaction is the sum of free energies of formation of the products in their standard state, minus the free energies of formation of the reactants in their standard state.

$$\Delta G^\circ_r = \Sigma\,\Delta G^\circ_f\,(\text{products}) - \Sigma\,\Delta G^\circ_f\,(\text{reactants}) \qquad [3]$$

Chemical reactions will take place to achieve the lowest energy state. Thus, when ΔG°_r is negative, the products of the reaction are more stable than the reactants.

Relation Between Reaction Free Energy and Equilibrium Constant

The free energy of change (ΔG°_r) in a reaction is related to the thermodynamic equilibrium constant (K) by the equation

$$\Delta G^\circ_r = -RT\ln K \qquad [4]$$

where R is the gas constant, and T is the absolute temperature.

Ionic Strength

Ionic strength, I, is defined as

$$I = 1/2\,\Sigma\, m_i Z_i^2 \qquad [5]$$

where m_i is the molality, and Z_i is the charge of the $i^{\underline{th}}$ ion in the solution. The ionic strength of soil solutions can be estimated from the electrical conductivity of a saturation extract by the equation

$$I = 0.013\ EC \qquad [6]$$

where the ionic strength (I) is in moles/liter, and the electrical conductivity (EC) is in millimoles/cm^2 at 25°C.

Activity Coefficient

Ions in electrolytic solutions exert long-range forces on one another with a resulting lowering of the ionic activity. In dilute solutions, the activity coefficient is given by the modified Debye-Hückel expression

$$-\log \gamma_i = \frac{A Z_i^2 \sqrt{I}}{1 - a_i B\sqrt{I}} + \beta I \qquad [7]$$

where

γ_i = activity coefficient of the $i^{\underline{th}}$ ion

I = ionic strength of the solution

Z_i = valence of the $i^{\underline{th}}$ ion

a_i = effective diameter in angstroms of the $i^{\underline{th}}$ ion in solution (3.5 for OH^-, F^-; 4.2 for $H_2PO_4^-$, HPO_4^{-2}, PO_4^{-3}, SO_4^{-2}; 5 for S^{-2}; 6 for Ca^{+2}, Fe^{+2}; and 9 for Al^{+3})

β = 0.257 (Moreno et al, 1960)

m_i = molar concentration of the $i^{\underline{th}}$ ion

A = temperature dependent coefficient 0.508 @ 25°C (Garrels and Christ, 1965)

B = temperature dependent coefficient 0.328 @ 25°C (Garrels and Christ, 1965)

Application of Chemical Thermodynamics to Phosphate Transport Through Soils

With the above introduction to chemical thermodynamic concepts, the reader should be able to apply literature data to the evaluation of treatment cites for land application. Sadiq and Lindsay (1978) have reviewed the literature and compiled a list of free energies of formation for numerous compounds occurring in soils. Table 1 is a partial listing, from Sadiq and Lindsay (1978), of compounds related to phosphate chemistry. As can be seen, there are numerous compounds which might be considered even in this partial listing. For simplification, only one aluminum phosphate (Variscite), one iron phosphate (strengite), and three calcium phosphates (dicalcium phosphate dihydrate, octacalcium phosphate, and hydroxyapatite) will be discussed in relation to the equilibrium chemical thermodynamic model.

TABLE 1

Gibbs Free Energy of Formulation for Selected Compounds Related to Phosphates in Soil Solutions (after Sadiq and Lindsay, 1978)

SPECIES	ΔG_f° (K cal/mole)
Al^{+3}	-117.33
$Al(OH)^{+2}$	-167.17
$Al(OH)_2^+$	-218.02
$Al(OH)_3^\circ$	-266.94
$Al(OH)_4^-$	-312.25
$Al(OH)_3$ (amorp)	-274.21
α-$Al(OH)_3$ (bayerite)	-275.78
γ-$Al(OH)_3$ (gibbsite)	-276.43
$AlPO_4$ (berlinite)	-388.50
$AlPO_4 \cdot 2H_2O$ (variscite)	-505.97
$Al(SO)_4^+$	-299.64
$Al(SO_4)_2^-$	-475.82
$Al_2(SO_4)_3^\circ$	-765.94
$Al_2(SO_4)_3$ (C)	-758.08
$Al_2(SO_4)_3 \cdot 6H_2O$ (C)	-1121.93
Ca^{+2}	-132.52
$CaHCO_3^+$	-274.33
$CaCO_3$ (aragonite)	-269.87
$CaCO_3$ (calcite)	-270.18
$CaCO_3 \cdot 6H_2O$ (ikaite)	-607.52
$CaPO_4^-$	-386.51
$CaHPO_4^\circ$	-398.29
$CaH_2PO_4^+$	-406.28
$CaHPO_4$ (monetite)	-403.96
$CaHPO_4 \cdot 2H_2O$ (brushite)	-516.89
$Ca(H_2PO_4)_2 \cdot H_2O$ (C)	-734.48
α-$Ca_3(PO_4)_2$ (C)	-922.70
β-$Ca_3(PO_4)_2$ (whitelockite)	-927.37
$Ca_8H_2(PO_4)_6 \cdot 5H_2O$ (octacalcium phosphate)	-2942.62
$Ca_{10}F_2(PO_4)_6$ (fluorapatite)	-3094.73
$Ca_{10}(OH)_2(PO_4)_6$ (hydroxyapatite)	-3030.24

TABLE 1 (continued)

SPECIES	ΔG_f° (K cal/mole)
Fe^{+2}	-21.8
Fe^{+3}	-4.02
$FeCl^{+2}$	-37.41
$FeCl_2^\circ$	-84.44
$FeCl_2^+$	-69.67
$FeCl_3^\circ$	-99.19
$FeOH^+$	-69.29
$Fe(OH)^{+2}$	-57.72
$Fe(OH)_3^-$	-148.22
$Fe(OH)_2^\circ$	-113.29
$Fe(OH)_2$ (C)	-117.58
$Fe(OH)_3$ (amorp)	-169.25
$Fe(OH)_3$ (soil)	-170.41
$FePO_4$ (C)	-283.19
$FeHPO_4^\circ$	-288.74
$FeHPO_4^+$	-280.93
$FeH_2PO_4^+$	-297.33
$FeH_2PO_4^{+2}$	-283.28
$FeSO_4^\circ$	-202.75
$FeSO_4^+$	-187.63
FeO (C)	-60.1
α-Fe_2O_3 (hematite)	-177.85
Fe_3O_4 (magnetite)	-243.47
α-$FeOOH$ (goethite)	-117.42
$FePO_4 \cdot 2H_2O$ (strengite)	-398.58
$Fe_3(PO_4)_2 \cdot 8H_2O$ (vivianite)	-1058.36
α-FeS (C)	-23.40
FeS_2 (markasite)	-37.83
FeS_2 (pyrite)	-38.78
PO_4^{-3}	-245.18
HPO_4^{-2}	-262.03
$H_2PO_4^-$	-271.85
$H_3PO_4^\circ$	-274.78
OH^-	-37.594
H_2O	-56.687

Consider first the dissolution of Variscite.

$$AlPO_4 \cdot 2H_2O \rightleftharpoons Al^{+3} + H_2PO_4^- + 2OH^- \quad [8]$$

To predict the phosphate activity from the above reaction, knowledge of the aluminum activity and hydroxyl activity are required. A common approach is to assume the aluminum in the reaction is made available from the solubilization of a naturally occurring mineral such as Gibbsite. The solubility of these minerals are also pH dependent, and little data is readily available on the amounts of such minerals in given soil samples. Thus, predicting equilibrium conditions based on the presence of one of these minerals may lead to significant errors. In cases where the activity of aluminum in the wastewater stream and the activity of aluminum in the soil solution are approximately the same, this aluminum activity should be used. For the purpose of discussion, assume Gibbsite controls the solubility of aluminum then

$$AlPO_4 \cdot 2H_2O + H_2O \rightleftharpoons Al(OH)_3 + H^+ + H_2PO_4^- \qquad [9]$$

From Eq. 9 and Table 1 ΔG_r° is positive; therefore, the reaction should proceed toward the $AlPO_4^- \cdot 2H_2O$ (Variscite). From Eq. 4 we determine log K = -10.54 or

$$10^{-10.54} = \frac{[Al(OH)_3][H_2PO_4^-][H^+]}{[AlPO_4 \cdot 2H_2O][H_2O]} \qquad [10]$$

at atmospheric pressure and 25°C. Assuming standard state conditions for $[Al(OH)_3]$, $[AlPO_4 \cdot 2H_2O]$, and $[H_2O]$, Eq. 10 reduces to

$$10^{-10.54} = [H_2PO_4^-][H^+] \qquad [11]$$

To obtain the total phosphate activity, both $H_2PO_4^-$ and HPO_4^{-2} species must be considered in normal pH ranges found in soils (Enfield et al, 1977).

$$H_2PO_4^- \rightleftharpoons H^+ + HPO_4^{-2} \qquad [12]$$

$$\Delta G_r^\circ = 9.82 \qquad [13]$$

Again from Eq. 4 when the temperature is 25°C log K = -7.20 or

$$[HPO_4^{-2}] = \frac{10^{-7.2}[H_2PO_4^-]}{[H^+]}$$

The total phosphate activity $[P_T]$ is then

$$[P_T] = [HPO_4^{-2}] + [H_2PO_4^-] \quad \text{or} \quad \frac{10^{-17.74}}{[H^+]^2} \quad \frac{10^{-10.54}}{[H^+]} \qquad [15]$$

From Eq. 15, the molar phosphate activity, for a Gibbsite-Varisite system, can be calculated as a function of pH. Before this activity can be correlated to the measured concentration in the soil solution, the ionic strength, Eq. 5 or 6, and the activity coefficient, Eq. 7, must be calculated. Table 2 gives a sample analysis for a non-saline soil extract used in the example development.

TABLE 2
Major Cations and Anions in Sample Problem

	Concentration	
ION	(mg/l)	Molar
Ca^{+2}	88	0.0022
Mg^{+2}	2	0.0001
Na^+	46	0.0020
K^+	31	0.0008
Cl^-	50	0.0014
SO_4^{-2}	77	0.0008
HCO_3^-	177	0.0029

The ionic strength (I) of the solution in Table 2 is then 0.00895, and the activity coefficient for phosphorus is 1.24. Using this activity coefficient and Eq. 15, a solubility diagram in terms of solution concentration and pH can be developed as shown in Fig. 1.

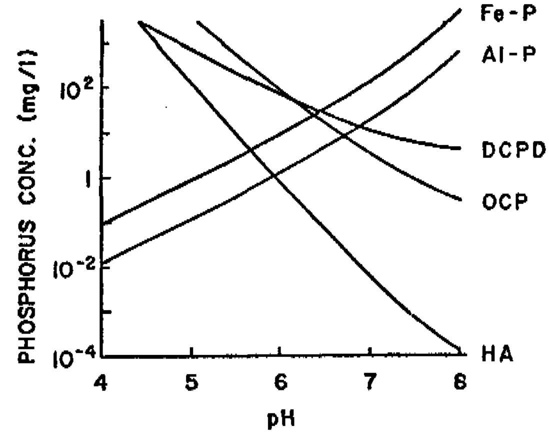

Fig. 1. Equilibrium solubility of selected phosphate compounds vs pH. Fe-P is the disassociation of strengite in the presence of $Fe(OH)_3$ (soil). Al-P is the disassociation of variscite in the presence of $Al(OH)_3$ (amorp). DCPD is the disassociation of brushite (dicalcium phosphate dyhydrate), OCP is the disassociation of octacalcium phosphate, and HA is the disassociation of hydroxyapatite in the sample wastewater Table 2.

In a similar manner, iron and calcium phosphate equilibrium solubility diagrams can be developed as shown in Fig. 1.

The equilibrium chemical thermodynamic model gives no information on how fast a reaction will take place. Capacities can be estimated for given initial conditions. For example, at a pH of 8.0 where calcium phosphate will control the equilibrium phosphorus in solution. The capacity of a soil to react with phosphorus can be estimated from the total calcium in a soil which has not received wastewaters. One problem with this approach is: How much calcium will be in the soil profile after 20 years of wastewater application, and how is this calcium fractionated? Under acidic conditions, a considerable knowledge of the mineralogy is required before accurate estimates can be made. Thus, it will be extremely easy to incorrectly estimate a system's capacity using this approach.

REVIEW OF EXISTING FLOW MODELS

Chemical thermodynamics gives information on the potential for a reaction to take place but does not give any information on how fast one can anticipate being able to apply wastes to the land without having detrimental problems. Most modeling approaches attempt to answer the rate question while predicting capacities of a given soil to react with phosphorus. Several approaches have been proposed beginning about 1970 (e.g. Kuo and Lotse, 1972; Shaw et al, 1975; Harter and Foster, 1976; Enfield and Bledsoe, 1975; Novak et al, 1977; Enfield et al, 1977; and Mansell et al, 1977). The models can be classified in three broad groups--those which are strictly empirical; those based on a sorption theory; and those based on more than one mechanism. Selected models from each class are presented. Some advantages and disadvantages of the models are discussed.

Empirical Models

Harter and Foster (1976) used a sorption isotherm which was a n^{th} order polynomial of the form

$$S = A + BX + CX^2 + \cdots NX^{n-1} \qquad [16]$$

where S = amount of phosphorus sorbed
X = amount of phosphorus added
A-N are constants.

This equation was developed by repeated applications of phosphorus to a soil sample under laboratory conditions until no further sorption was observed. From this sorption isotherm, Harter and Foster predicted breakthrough curves dependent only on the rate phosphorus is applied to the system. This approach is quite similar to that proposed by Taylor and Kunishi (1974), since neither addresses the rate of reaction, and both are primarily concerned with the capacity of a soil to react with phosphorus.

Advantages of either approach are their ease of application. Useful conservative results can be obtained from either approach, provided soil from the proposed treatment site and wastewater are used in the laboratory studies.

Limitations to equilibrium sorption isotherms include: 1) During the past few years there has been a reducing trend in the phosphate concentration in wastewater streams. Since Harter and Foster's procedure was developed for a single concentration, there does not appear to be an adequate way of accounting for changes in applied concentration.
2) The amount of time allowed for the reaction to take place will give different results. This observation has been made by numerous researchers (Coleman et al, 1960). It is, therefore, possible to obtain an infinite number of curves of isotherms for each soil dependent on the time allowed for equilibration. Figure 2 shows a similar type of data for phosphorus sorption by soils taking into consideration contact time. From this data, the importance of contact time can be seen.

Enfield (1974) presented an empirical model which includes reaction rate in its formation. The equation is exponential in form and has characteristics similar to a time dependent Freundlich isotherm, Eq. 20. Enfield proposed describing the reaction rate as

$$\frac{\partial S}{\partial t} = a\, C^b\, S^d \qquad [17]$$

where
$\frac{\partial S}{\partial t}$ = rate of sorption
C = phosphorus solution concentration
S = concentration of phosphorus in the solid phase
a, b, and d are constants.

The equation has been shown to describe time dependent sorption isotherms (Enfield et al, 1976) and do a fair job of predicting phosphate transport through soils (Enfield, 1976). This approach does not, however, take into consideration:

1) Differences in composition of applied wastes.
2) Desorption is not permitted.
3) Soils would have infinite capacity at infinite times.

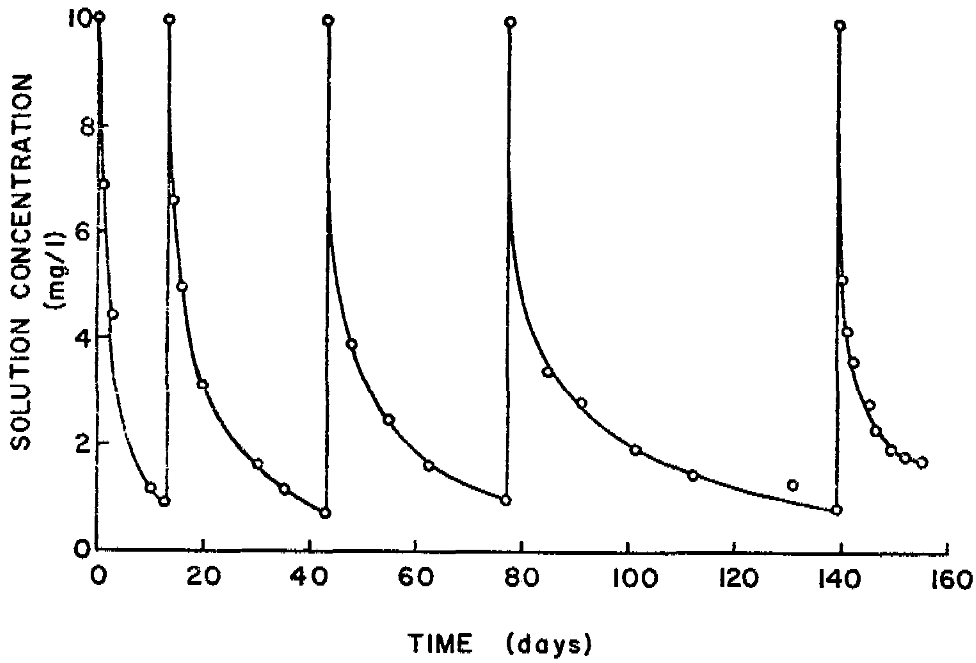

Fig. 2. Change in solution concentration vs time for several applications of Phosphorus.

Sorption Theory Kinetics Models

Sorption theory kinetic models are one step more advanced than equilibrium sorption models or empirical kinetic models. These models assume equilibrium can be described by some adsorption theory such as:

1) linear sorption

$$S = KC \qquad [18]$$

where K is a constant

2) Langmuir adsorption

$$S = \frac{S_m BC}{1 + BC} \qquad [19]$$

where

S_m is the maximum concentration which can be sorbed by the solid phase
B is a constant related to the bonding energy

3) Freundlich sorption

$$S = mC^n \qquad [20]$$

where
m and n are constants.

They further assume the force driving the reaction is the difference between "equilibrium" and the systems current status. Different researchers have used different methods to describe the kinetics. The majority are first order kinetic models (e.g. Novak et al., 1975); others are diffusion limited models (e.g. Enfield et al, 1976).

Novak et al (1975) proposed the equation

$$\frac{\partial S}{\partial t} = K_o A(C - C^*) \qquad [21]$$

where

C^* = equilibrium concentration for langmuir adsorption
A = liquid interface area
K_o = mass transfer coefficient

Enfield and Bledsoe (1975) used a similar approach with equilibrium described by a Freundlich equation

$$\frac{\partial S}{\partial t} = \beta (S^* - S) \qquad [22]$$

where β = soil dependent variable which includes both liquid and solid interface area and mass transfer coefficient of Novak et al, (1977).
S^* = equilibrium sorption described by Freundlich equation.
A linear approach

$$\frac{\partial S}{\partial t} = \alpha (KC-S) \qquad [23]$$

where α is a constant was presented by Enfield and Shew (1975).

An advantage to the linear model is the existence of analytical solutions for simulation (Oddson et al, 1970). There is considerable doubt, however, that linear sorption can adequately describe the equilibrium status for phosphorus. A deficiency common to each of the sorption theory kinetic model is the ability to accurately describe the equilibrium condition. The reaction of phosphorus with soils is much more complex than sorption alone.

Multiple Mechanistic Models

Models with multiple mechanisms essentially assume more than two phases for the phosphorus. Mansell et al (1977) assumed four phases to be present-- a) solution, b) adsorbed, c) immobilized (chemisorbed), and d) precipitated.

The equations describing the rates of transfer between the phases were given as follows:

$$\frac{\partial(\Theta A)}{\partial t} = -\Theta(K_1 A^N + K_5 A)$$

$$+ \rho\ (K_2 B + K_6 D) \qquad [24]$$

$$\frac{\partial(\rho B)}{\partial t} = K_1\ \Theta A^N$$

$$- [K_2 + K_3]\ \rho B + K_4 \rho C \qquad [25]$$

$$\frac{\partial(\rho C)}{\partial t} = K_3\ \rho B - K_4 \rho C \qquad [26]$$

$$\frac{\partial(\rho D)}{\partial t} = K_5\ (\Theta A) - K_6 \rho D \qquad [27]$$

where

Θ = volumetric soil water content
ρ = soil bulk density
t = time
A = concentration of phosphorus in solution
B = amount of phosphorus adsorbed
C = amount of phosphorus immobilized
D = amount of phosphorus precipitated
N = constant representing order of adsorption process
K_1 = rate coefficient for adsorption
K_2 = rate coefficient for desorption
K_3 = rate coefficient for immobilization
K_4 = rate coefficient for mobilization
K_5 = rate coefficient for precipitation
K_6 = rate coefficient for dissolution

As written, the model could also be considered a three phase system where sorption (adsorption and chemisorption) is a two step process. The model is linear first order except for adsorption which could be considered possibly being non-linear when N is not 1. If the equations are all linear, the numerical solution, when combined with a flow model, can be solved analytically rather than some approximating technique. This greatly reduces simulation time. The model has several degrees of freedom. Thus, using curve fitting techniques, it should be possible to very accurately reproduce experimental breakthrough curves. The question becomes : Are the same coefficients adequate for different flow bounddaries? In other words, are the coefficients directly related to physical or chemical properties of the soil? It will take considerable effort to evaluate the validity of the model.

Another multistepped model which conceptually is quite similar to the multistepped model of Mansell et al (1977) was presented by Enfield et al (1977). Enfield et al assumed adsorption was instantaneous and described by a Langmuir equation (Eq. 19) rather than a Freundlich function (Eq. 20) as assumed by Mansell et al. Enfield et al further assumed that precipitation and dissolution were reversible. The precipitation process was assumed first order but limited by chemical thermodynamic equilibrium. They further limited the maximum precipitation or dissolution to the chemical limits of the controlling soil system. Their model follows the equation

$$ST(t) = \frac{S_m bC(t)}{1 + bC(t)} + \sum_{j=1}^{n} S_{xj} \qquad [28]$$

where

$ST(t)$ = theoretical phosphorus sorption at some time t
S_m = Langmuir sorption maximum

$$\sum_{j=1}^{n} S_{xj} = \begin{cases} \int_0^t a_j \,(C(t) - CE_j)\, dt & \text{for } S_{xj} < S_{mxj} \\ S_{mxj} & \text{for } S_{xj} \geq S_{mxj} \\ 0 & \text{for } S_{xj} \; \underline{K} \; 0 \end{cases}$$

where

a_j = a rate constant for the j^{th} species under consideration

CE_j = thermodynamic chemical equilibrium concentration of the j^{th} species

S_{mxj} = the maximum amount of phosphorus which can be formed by a particular ionic species

b = Langmuir constant related to the bonding energy

C(t) = solution concentration at some time t

Again the model can have several degrees of freedom. The number of degrees of freedom will depend on the number of ionic species considered by the model. Enfield et al considered three species of which two influenced the system under acidic conditions, while one was the predominate factor under basic conditions.

They also showed a reasonable correlation between pH and the rate coefficient particularly under basic conditions. If the rate coefficients are not dependent on soil characteristics and dependent only on the pH and the species being formed, the number of dependent variables will be reduced.

MODEL EVALUATION AND APPLICATION TO LAND TREATMENT DESIGN

Most of the models have not been adequately evaluated. Generally the data presented in the literature was developed by "curve fitting". In other words, a preconceived model was regressed to a set of experimental data to obtain a series of regression coefficients. Some models were "fitted" from batch sorption studies while other studies were based on column data. There are some studies such as Enfield and Shew (1975) and Enfield (1976) which used the regression coefficients derived from batch sorption studies to predict the results of column studies. These studies were performed under laboratory conditions with controlled flow and solution compositions. Published data attempting to evaluate under field conditions the above mentioned models are lacking. Fig. 3 reproduces some of the findings of Enfield (1976) and Enfield and Shew (1975) where attempts were made to evaluate models. Four different modeling approaches were evaluated--1) after the empirical model, 2) an instantaneous form of langmuir adsorption, 3) a simplified approximation of langmuir adsorption with first order kinetics similar to Eq. 21, 4) a linear form of adsorption with first order kinetics (Eq. 23). None of the models

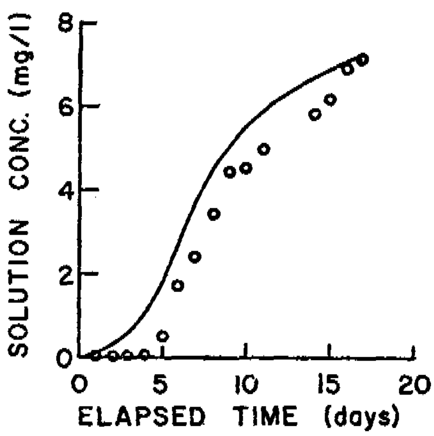

Fig. 3-1. Comparison between predicted phosphorus breakthrough curve and measured breakthrough curve using Eq. 17.

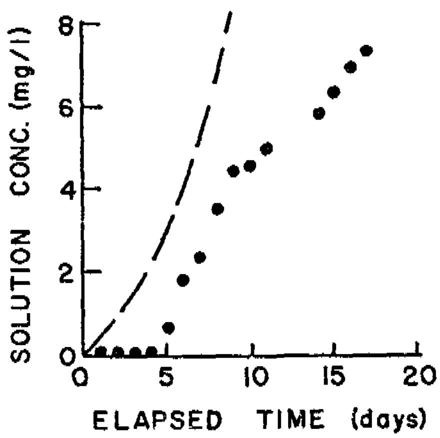

Fig. 3-2. Comparison between predicted phosphorus breakthrough curve and measured breakthrough curve based on Eq. 19.

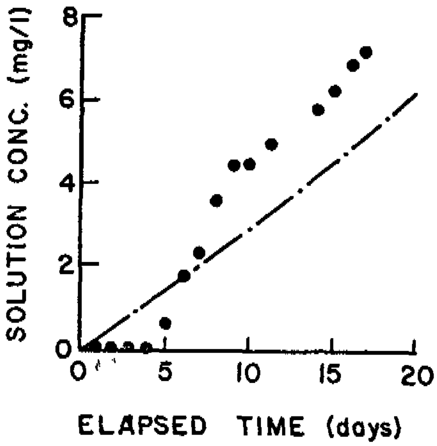

Fig. 3-3. Comparison between predicted phosphorus breakthrough curve and measured breakthrough curve based on Eq. 19 and Eq. 22.

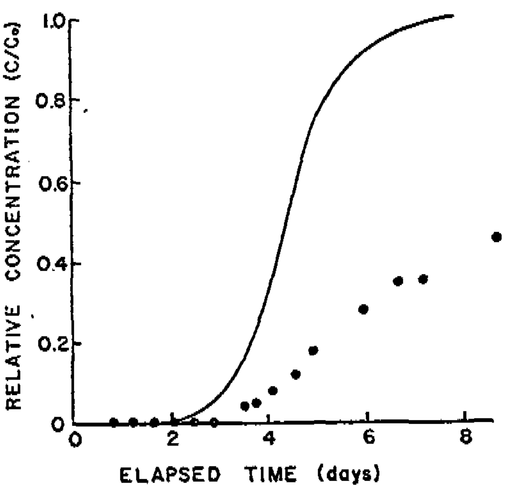

Fig. 3-4. Comparison between predicted phosphorus breakthrough curve and measured breakthrough curve based on Eq. 18 and Eq. 21.

evaluated are totally satisfactory. The closest fit was from the empirical model (Eq. 17). But with limitations previously mentioned, it should not be considered satisfactory for design purposes.

The conclusion that must be made is--no currently available model has been adequately demonstrated to satisfactorily predict phosphorus concentrations in percolate waters resulting from land treatment. This conclusion corresponds to a recent review by Pratt (1977). This conclusion does not mean all of the models are inadequate; only a demonstration of the adequacy or inadequacy has not been provided.

TENTATIVE DESIGN APPROACH

With the uncertainty of current prediction models, the designer has difficulty evaluating systems for specific sites. Pratt's proposed design model (1977, sec. B.4.4) uses long sorption times to estimate sorption capacity and recommends this approach for slow rate systems. The method is quite similar to the models of Harter and Foster (1976) or Taylor and Kunishi (1974), except long equilibrium times are permitted for the reaction to proceed. This may be as good as any other method at the present time to estimate capacities. For high rate systems, Pratt suggests employing Eq. 17. Since design models for the prediction of phosphates in percolates have not been field tested, a more logical recommendation might to be use Eq. 28 to estimate maximum application rates. Crude conservative estimates of maximum application rates can be obtained without time dependent sorption isotherms.

For example, assume a designer is required to estimate the maximum application rate for a high rate system. The designer determines 10 mg/l phosphorus wastewater with pH7.5 ionic strength of 0.009 is to be applied to a soil with pH 7.5 and bulk density 1.5 g/cc. The discharge requirement one meter below the surface is 1.5 mg/l of phosphorus. No plants are to be grown on the system, and it is to operate without seasonal constraints.

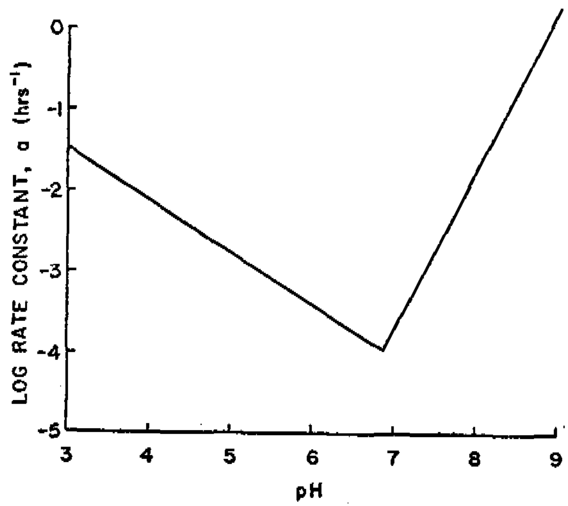

Fig. 4. First order rate constant in Eq. 18 as a function of pH developed from Enfield et al, 1977.

From Fig. 4, the estimated rate coefficient $10^{-2.7}$ h^{-1} for above conditions is obtained. Under the worst possible conditions, or a driving force of 1.5 mg/l, the maximum application rate would be 3×10^{-8} micrograms phosphorus per gram of soil per hour. This is equivalent to applying 1.3 cm of wastewater per day. Under optimum conditions, or a driving force of 10 mg/l, the allowable application rate would increase to approximately 8 cm per day.

This procedure is crude but will give an appropriate range for design purposes without the complexities of computer analysis. Much additional work is required to validate all of the mentioned models and determine their applicability and accuracy before they can be recommended for use in estimating percolate water quality from land application.

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LAND TREATMENT MATHEMATICAL MODELING

EVALUATION OF N MODELS FOR PREDICTION OF NO_3-N IN PERCOLATE WATER IN LAND TREATMENT

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ABSTRACT

Nitrogen simulation models developed to describe one or more processes in agricultural soils can be adopted for land treatment. The most important processes in the simulation of N transformations for prediction of N in percolate water in land treatment are: nitrification, denitrification, plant uptake and exchange of NH_4 with the soil. The N model must be incorporated into a moisture flow model. It was concluded that the Michaelis-Menten type model is the most appropriate, although the first order kinetic may be used to describe the nitrification process. Modeling the denitrification process in slow infiltration must include biodegradable carbon and dissolved oxygen as limiting factors. Although several large models are available to simulate and predict N in leachate in land treatment, a need for a simplified model that can be tested in the field is apparent.

INTRODUCTION

In land treatment, nitrogen is almost always the factor limiting the rate of application. Excessive nitrate concentrations in groundwater are a great health concern. This is due to its association with infant methomoglobinemia (blue baby syndrome) and eutrophication of natural waters. Consideration of land treatment as an alternative to advanced waste treatment has been hampered by the lack of hard scientific data on the fate of N that would allow efficient design of systems without incurring health risks on one hand but avoiding overdesign (cost ineffectiveness) on the other.

Land treatment of wastewater is not a new concept. Systems were built in 1559 in Lower Silesia and Germany (Iskandar, 1978). Present interest, however, is in the design of cost-effective and reasonably safe systems. Thus a quantitative description is needed of the chemical, physical, and biological interactions that occur between the wastewater constituents and the soil system to make possible accurate recommendations on how and when wastewater can be applied and predictions of the quality of the leachate.

Allison (1966) published a review entitled, "The fate of nitrogen applied to soil." While it may be claimed that little that is new has been reported in this field since then, there have been two major trends:

(1) In the past, we were primarily concerned with the problem of getting more fertilizer into a crop. At the present time, and particularly in land treatment, the primary focus is on the fate of applied N not taken up.

(2) Ways of using research data are changing. With recent advances in the use of computers and development of techniques for modeling ecological systems, we are becoming better equipped to describe (or even predict) quantitatively the behavior of nitrogen in soil systems).

The fact that in land treatment systems nitrogen is applied in small amounts repeatedly (most often weekly), in contrast to normal agricultural fertilizing practice, should make significant differences in the nitrogen transformation processes. Also, the soils under land treatment are most often kept wet (above field capacity), and the water flow pattern as well as N transformation processes will vary significantly from those of an agricultural regime.

The objective of this paper is to review the literature in regard to modeling the nitrogen transformation in soil-plant-water systems, with particular emphasis on possible application to land treatment systems. Because of limited space, only those models developed for prediction of percolate quality will be discussed.

TYPES OF MODELS

There are three types of mathematical models. The first are mechanistic type models which were developed to understand the response of a biological system in terms of the mechanisms present. They are constructed by looking at the system, dividing it into components, and acquiring an understanding of the behavior of the whole system in terms of the individual system components and their interaction one with another. The steps that are most often taken are assumptions, mathematical formulation, solution of equations, and finally validation. The last step is carried out to test the accuracy and validity of the assumptions. The second type of model is empirical. These models let us understand the response of a system without going through the structure, making assumptions, and understanding the mathematical consequences of those assumptions. Briefly looking at the experimental data, possibly doing some analysis of the data and trying to make an "intelligent" guess at a (simple) form of equation or set of equations. The third type of model is a mixture of the first two and is the type now being used most of the time.

SUBMODELS FOR N PREDICTION IN PERCOLATE WATER

To predict nitrogen concentration in percolate water from land treatment, at least three major steps need to be considered:

(1) Chemical and/or biological transformation of nitrogen species in soils.
(2) Transport of water and soluble N species of soil water.
(3) Plant uptake of water and N species.

Figure 1 illustrates the major processes that may occur in land treatment of wastewater. The form of N applied to the land depends largely on the pretreatment steps. However, in most cases most of the N applied (about 85%) is in the ammonium form (Iskandar

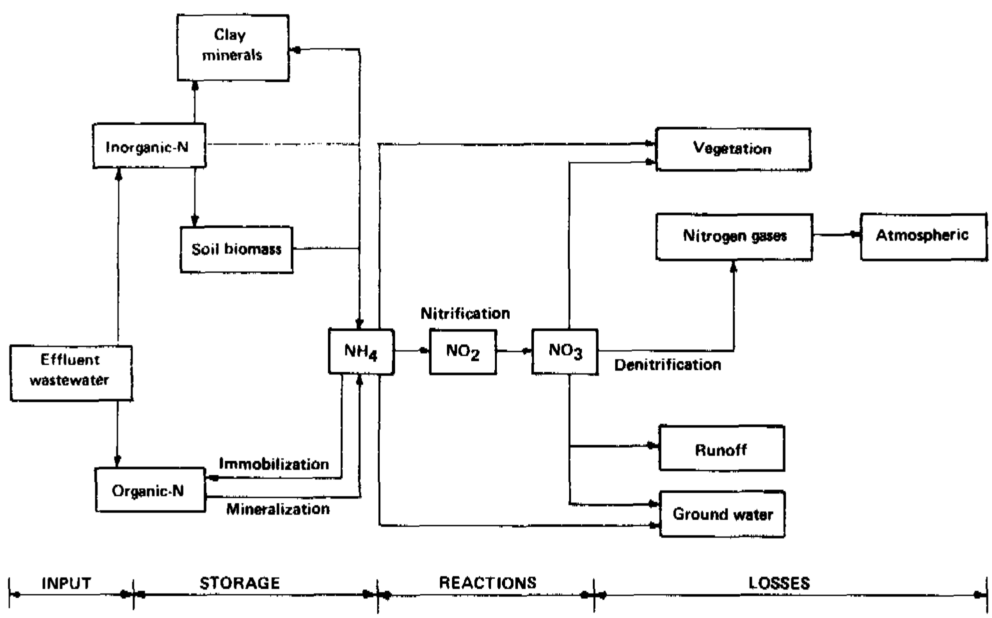

Figure 1. Nitrogen transformations in land treatment.

et al., 1976), while organic N and NO_3-N constitute <15%. Once NO_3 is formed, it moves down as fast as or even faster than the water. Therefore, the first process to be considered in the model must be nitrification. Nitrification is the process whereby ammonium ions are changed to nitrate by biochemical action. Simulation of this process is important for predicting the amount of NO_3 percolating to the groundwater and as a bookkeeping record of plant uptake of N from the NH_4 and NO_3 sources.

Modeling the Nitrification Process

Since the extensive review of the literature on the nitrification and nitrifying bacteria by Painter (1970), a large amount of quantitative and semi-quantitative information on this process has become available (Jenkins, 1977; Focht and Chang, 1975; Sharma and Ahlert, 1977; Duffy et al., 1975; Mehran and Tanji, 1974; Endelman et al., 1973; Dutt et al., 1972; Hagin and Amberger, 1974; Beek and Frissel, 1973; Ardakani et al., 1974; Starr et al, 1974; Cameron and Kowalenko, 1976; McLaren, 1970). There have been contradictory descriptions of the simulation process of nitrification. Cameron and Kowalenko (1976), Duffy et al. (1975), Mehran and Tanji (1974), Misra et al. (1974) and Starr et al. (1974) considered the process as first order kinetic while it has been considered zero order kinetic by Beek and Frissel (1973) and Sabey et al. (1969). Other descriptions such as sigmoid (Hagin et al., 1976; Lees and Quastel, 1946), logarithmic (Quastel and Scholefield, 1951; Stojanovic and Alexander, 1958) and Michaelis-Menten (Ardakani et al., 1973, 1974; Laudelout et al., 1977; McLaren, 1970; Nishio and Furusaka, 1971) have also been reported. Recently, Leggett and Iskandar (1978) reviewed the literature on the simulation of the nitrification process in soils amended with ammonium and concluded that the Michaelis-Menten type is the most reasonable approach. They also developed and validated an improved model for simulating the nitrification process in soils under land treatment conditions. Although several investigators have previously used Michaelis-Menten's equations to fit their data, very little work has been done on the effect of environmental factors such as pH, temperature and oxygen concentration on the rate of nitrification. Leggett and Iskandar included in their model the effect of temperature and pH, but due to the complexity involved, felt that oxygen could not be included at this time. However, they concluded that oxygen would not be as critical in land treatment because of the small periodic nature of the nitrogen increment. In other words, the kinetics may truly be nitrogen-limited as is required for the single substrate Michaelis-Menten model to be valid.

Modeling the Denitrification Process

Nitrate reduction by bacteria was documented as early as 1886 (Gayon and Dupetit, 1886). Since then many publications have become available; however, there is very little quantitative description of this process which may be due at least partially to the lack of complete understanding of the effects of the environmental factors and to the difficulty in identifying the responsible species. Nommik (1956) and Bremner and Shaw (1958) studied the factors affecting denitrification in soils and more recently Focht and Chang (1975) reviewed the process as it is related to wastewater effluent. They found that pH, temperature, water content, oxidation status, and organic carbon are the most important factors. The rate of denitrification was very slow at a pH below 4.8 and increased very rapidly at pH 8.0-8.6. Denitrification is a relatively temperature-tolerant process, with an optimum temperature close to 60°C (Nommik, 1956). It is very slow between 5 and 10°C and stops completely at temperature <5°C. The data of Nommik show a linear response to temperature ranging from 10 to 60°C.

The degree of water saturation of soil has a significant influence on the rate of denitrification. Bremner and Shaw (1958) concluded that if all other variables are favorable for denitrification, little loss of N occurs if the moisture content is less than 60% of the water holding capacity of the soil. It seems that the effect of moisture content on denitrification is indirect--it affects the oxygen content and the Eh (oxidation reduction potential) of the soil. Focht and Chang (1975) concluded that the critical factor governing denitrification in sewage water is the dissolved oxygen concentration, not the composition of the atmosphere. The critical oxygen concentration proposed by Wuhrmann (1964) was 0.1 mg O_2/l. At any higher rate, denitrification could be described as of zero order due to the small Michaelis constant ($K_m \approx 10^{-6}$ m) for respiration rates of microorganisms (Painter, 1970). In this case the reduction of organic substrate is the limiting step. Focht and Chang (1975) related the aeration status of the wastewater to the Eh. Patrick (1961) reported that denitrification in soils occurs at a potential of 300-350 mV or less. The aeration status is important not only in influencing the rate of denitrification but also in determining whether any intermediate products might accumulate.

Temperature is undoubtedly the most important and most difficult environmental parameter to control in soils and causes the greatest problems, particularly during the winter months. The optimum temperature for denitrification is higher than that for nitrification, ranging from 65 to 75°C, and denitrification ceases at 85°C (Nommik, 1956). Dawson and Murphy (1972) showed that denitrification conformed to Arvenius kinetics from 3 to 28°C, although most other studies showed that the rate is affected proportionally more below the 10-15°C range (Nommik, 1956; Bremner and Shaw, 1958). Changes in temperature may also influence the composition of the products formed during denitrification. At lower temperature NO is detected in greater quantities in soils (Nommik, 1956). Several investigators (Nommik, 1956; Bailey and Beauchamp, 1973) have observed no reduction of NO_3 after 22 days at 5°C; however, they did detect NO as the only gas formed at 5°C. Novak (1974) considered the effects of temperature upon the Michaelis constant K_m and the maximum rate constant V_{max} and modified the standard Michaelis-Menten equation to consider exponential coefficients for both K_m and V_{max}.

The kinetics for denitrification are far more complex than for nitrification and have not been well developed. Because reduction of NO_3 is coupled with the oxidation of a carbon substrate, a kinetic model must involve at least a dual substrate enzyme(s) complex. Such equations have been developed (Bray and White, 1966) for idealized systems, but denitrification is far more complex. The nitrogenous intermediates presumably have different saturation constants and may be competitively inhibited at specific redox potentials, and the association complex of reductant and oxidant may involve one or several enzymes (Bremner and Shaw, 1958). Unfortunately, most of the studies have not even considered whether Michaelis-Menten kinetics are applicable and have taken a grosser, simplistic outlook by attempting to describe denitrification as a zero or first order reaction. Investigation with soil showed that the rate of denitrification was independent of nitrate concentration (Wiljer and Delwiche, 1954; Nommik, 1956; Bremner and Shaw, 1958; Cooper and Smith, 1963), but this is apparently due to the use of a high NO_3 solution concentration (usually 100 mg N/l or greater), which appears to be much lower when considered on a dry weight basis. Thus, apparent first order kinetics were reported by Stanford et al. (1975) in soil where solute concentrations were less than 32 ppm NO_3-N. Bowman and Focht (1974) showed that the apparent zero order and first order are the extreme ends of the standard Michaelis-Menten curve in describing denitrification provided the concentration of reductant or oxidant, held constant while the other was varied, was not limiting.

The discrepancies between reported zero-order (Wuhrmann, 1963; Dawson and Murphy, 1973) and first-order (Johnson, 1968; Balakrishnan and Eckenfelder, 1969; Mulbarger, 1971) kinetics observed in wastewater treatment were explained by Focht and Chang (1975) based on whether the system is carbon or N-limiting. Secondary treated sewage and soil systems are usually carbon limiting, so that addition of more NO_3 should not increase the rate of denitrification.

The rate of denitrification generally proceeds maximally at a C/N ratio of 2 to 3 (Dawson and Murphy, 1973; Wuhrmann and Mechsner, 1973) in sewage plants. This same ratio has also been found to apply to soils, which have much higher C/N ratios than sewage when exogenous carbon is added (Bremner and Shaw, 1958; Bowman and Focht, 1974). This would suggest that, like sewage, the bulk of the carbon is used for respiratory rather than assimilatory purposes. Increasing the C/N ratio by additions of methanol or other exogenous substrates beyond that needed for denitrification does not significantly reduce the effluent NO_3-N concentrations and increases the effluent BOD as Wuhrmann and Mechsner (1973) have shown.

Other Processes

Mineralization of organic-N, immobilization of inorganic-N, and plant uptake of N are the remaining three processes considered in N modeling. Mineralization of organic-N is a long-term process which has been simulated and studied for many years (Beek and Frissel, 1973; Hagin et al., 1976; Hagin and Amberger,1974). Studies on immobilization and mineralization will be discussed in the next section. However, for prediction of NO_3-N in percolate water from land treatment, it is possible to assume that the net change in soil N due to those two processes is equal over a short period of time (2-4 months).

Plant uptake of N is very important in land treatment, particularly the slow infiltration and overland flow types. On a yearly basis Iskandar et al. (1976) found that it constitutes up to 70% of the N removal mechanisms in a slow infiltration field experiment. Miller and Stuart (1978), however, reviewed the literature on plant uptake models for predicting water quality in land treatment (this volume). Therefore, it is probably more constructive to discuss complete N simulation models.

SIMULATION MODELS FOR N TRANSFORMATION AND TRANSPORT

So far, most N models have been developed to simulate one or two processes to provide an understanding of the basic mechanisms involved. Recently, efforts have been made to integrate the major N processes into a complete soil system by making use of data accumulated on the individual basic mechanisms incorporated into a moisture simulation model (Frere et al., 1970; Dutt et al., 1972; Beek and Frissel, 1973; Mehran and Tanji, 1974; Hagin and Amberger, 1974; Hagin et al., 1976; Cameron and Kowalenko, 1976; Reuss and Innis, 1977).

Mass balance modeling approaches have been proposed by Morel (1969) and Reuss and Innis (1977). The main feature in the model by Reuss and Innis (1977) is the introduction of a simple producer/ decomposer submodel. In this submodel, relative rates appropriate to the system being simulated are set externally. The model includes the state variables NH_4, NO_3, live roots, dead roots, soil organic N, live tops and litter. Temperature and soil water, driving variables, were supplied from external sources. The model has been tested in the field and has a potential for wider utilization in large-scale systems.

Greenwood et al. (1974) developed a dynamic crop response model with functions to describe leaching, nitrate uptake by plant roots, and growth. These models, however, have to be incorporated into larger simulation models which include water submodels in order to predict the quality of the percolate. Campbell et al. (1975) obtained reasonably good results by using regression analysis to relate change in NO_3-N concentration to soil temperature and moisture under both laboratory and field conditions. Multiple regression analysis was used by Dutt et al. (1972) to develop an empirical model for N transformation and transport in soils. The model was tested recently by Gupta and Schaeffer (personal communications), using data from an existing land treatment site in Minnesota. The predicted N data did not match the experimental data, which could be due to the assumptions used in the model (Dutt et al., 1972). The models by Beek and Frissel (1973), Hagin et al. (1976) and Hagin and Amberger (1974) were focused on simulation of organic matter decomposition and oxidation of mineralized NH_4 to NO_3. Hagin et al. (1976) assumed first order kinetics for the nitrification and denitrification rates. The rate constants were taken from the literature and were adjusted according to environmental factors such as pH, moisture, oxygen and water content. No validation of the model was given.

In summary, these large models are useful in aiding our understanding of the complete N cycle but they are very complicated due to the complex nature of the soil processes. The models must be simplified in order to be useful tools in managing or controlling nitrogen behavior in soils under land treatment. In addition, they should be modified to land treatment conditions, as they were developed for agricultural land where nitrogen is being applied in large quantities (fertilizer applications) but less frequently. In general, the models available lack the evaluation (validation) and field testing of model parameters.

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LAND TREATMENT MATHEMATICAL MODELING

NITROGEN BEHAVIOR IN LAND TREATMENT OF WASTE WATER: A SIMPLIFIED MODEL

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A simplified mathematical model was developed to describe transformations and transport of nitrogen under transient soil water flow conditions. Kinetic reactions were assumed to govern the nitrification and denitrification processes. A macroscopic approach was used to incorporate plant uptake of water as well as NO_3-N and NH_4-N from the soil solution. The sensitivity of the model to changes in rate of N transformation, N uptake by plants, and schedule and amounts of N application were also investigated. The model can be used as a tool to predict the fate of nitrogen in land treatment systems. The model is flexible and can be adapted to incorporate various nitrogen transformation mechanisms as well as layerings in the soil profile.

INTRODUCTION

Nitrogen models such as those described by Beek and Frissell (1973), Hagin and Amberger (1974) and Selim et al. (1976b), among others, provide a mechanistic description of the fate of nitrogen in the soil system. However, several disadvantages are inherent in the use of such models. These models require an extensive number of parameters. In addition, independent measurements of many of these parameters are difficult to achieve. Another disadvantage is that these models, because of their complexity, are difficult to validate. They also require an extensive amount of computer (CPU) time if predictions for one year or even a few months are desired.

In this paper, a simplified nitrogen model is presented (Fig. 1) for the purpose of describing the transport and transformations of soil nitrogen during saturated-unsaturated water flow conditions. The main feature of the proposed model is that it requires a minimum number of independently measured soil parameters. Furthermore, the model is simple to use and easy to adapt to incorporate soil and plant parameters.

MODEL DEVELOPMENT

Two major approximations were made in the development of the simplified nitrogen model. First, the nitrification process was considered as a single step, i.e. $NH_4 \rightarrow NO_3$, rather than a two-step process ($NH_4 \rightarrow NO_2 \rightarrow NO_3$). Such an assumption is considered adequate, since NO_2, in most soils with a neutral pH, is rapidly oxidized to NO_3: The second major simplification is that the organic-N phase was not incorporated in the model. Thus, in the simplified model it is assumed that the organic-N content is small and/or the rates of nitrogen mineralization as well as immobilization are extremely slow or equal to each other. Such an assumption is not valid if considerable mineralization and/or immobilization of nitrogen is encountered in the soil. However, if the average amount of mineralization approximately equals the amount of N immobilized, over an extended period (a few months or a year), the simplified model can provide an adequate description of the fate of nitrogen in the soil.

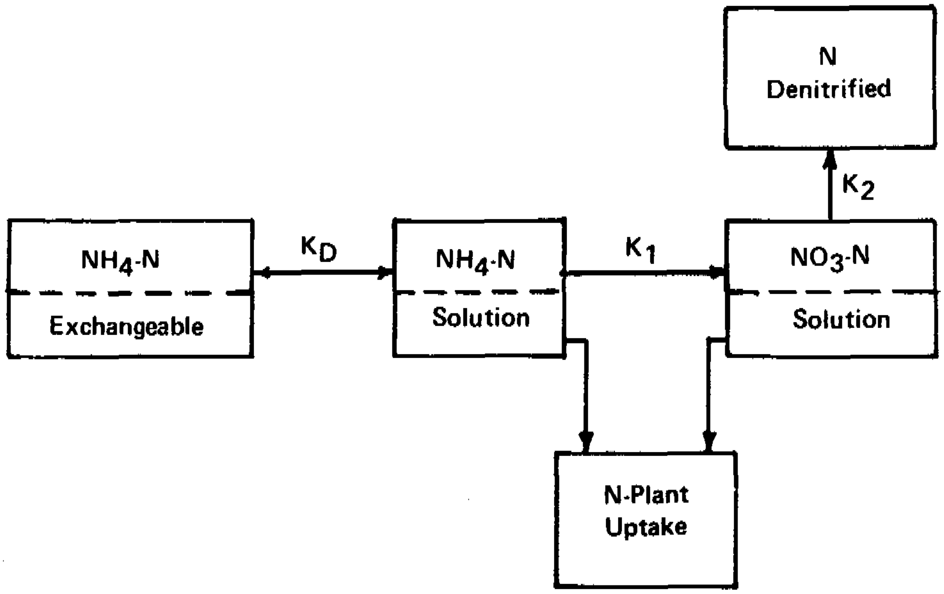

Figure 1. Schematic of nitrogen transformations in the simplified nitrogen model where $\bar{K}_1$ and $\bar{K}_2$ are rate coefficients for nitrification and denitrification, respectively, and K_D the Freundlich distribution coefficient for exchangeable NH_4-N.

The nitrogen transformation processes considered in the simplified model were: nitrification of NH_4 to NO_3, denitrification of NO_3, and ion exchange of NH_4. The ion-exchange process was considered instantaneous, whereas the nitrification and denitrification processes were of the first-order kinetic type (Mehran and Tanji, 1974; Hagin and Amberger, 1974). In future, a Michaelis-Menten kinetic type may be incorporated (Leggett and Iskandar, 1978).

The transport of NH_4-N and NO_3-N in the soil solution occurs as a result of molecular diffusion, mechanical dispersion, and convection or mass flow. Molecular diffusion results from the random thermal movement of molecules, whereas mechanical dispersion results from the velocity distribution of water in the soil pore space. A single dispersion coefficient (D) is commonly used which combines mechanical dispersion and diffusion. Thus, the convective-dispersive equation for NH_4-N and NO_3-N transport may be expressed as (Selim et al., 1976a)

$$\partial(\theta C)/\partial t = (\partial/\partial z)(\theta D \ \partial C/\partial z) - \partial(vC)/\partial z - \theta k_1 C - \rho \partial S/\partial t - q_{NH_4} \quad (1)$$

$$\partial(\theta Y)/\partial t = (\partial/\partial z)(\theta D \ \partial Y/\partial z) - \partial(vY)/\partial z - \theta k_1 C - \theta k_2 Y - q_{NO_3} \quad (2)$$

where

θ = soil water content (cm^3/cm^3)
C = concentration of NH_4 in soil solution ($\mu g/cm^3$)
Y = concentration of NO_3 in soil solution ($\mu g/cm^3$)
D = solute dispersion coefficient (cm^2/hr)
v = soil water flux (cm/hr)
t = time (hr)
z = distance in soil, positive downward (cm)
S = amount of NH_4 in exchangeable phase per gram soil ($\mu g/g$)
ρ = soil bulk density (g/cm^3)
k_1 and k_2 = kinetic rate coefficients for nitrification and denitrification (hr^{-1}), respectively

q_{NH_4} and q_{NO_3} = rate of plant uptake of NH_4-N and NO_3-N per unit soil volume ($\mu g/cm^3$ hr), respectively.

The first two terms on the right side of equations (1) and (2) account for solute transport, and are usually referred to as the dispersion and mass flow terms, respectively. The third and fourth terms of equation (1) account for nitrification and ion exchange of NH_4-N, respectively. Similarly, the third and fourth terms of equation (2) represent the nitrification and denitrification processes, respectively. The ion exchange process governing NH_4-N adsorption-desorption was assumed to be of the linear Freundlich type, i.e.

$$S = K_D C, \text{ or } \partial S/\partial t = K_D \partial C/\partial t \qquad (3)$$

where K_D, which is commonly referred to as the distribution coefficient (cm^3/g), represents the ratio of the amount of NH_4 adsorbed to concentration in the soil solution.

The terms q_{NH_4} and q_{NO_3} in equations (1) and (2) account for the rate of uptake of NH_4-N and NO_3-N respectively. Several models have been proposed to describe the flux of nitrogen to plant roots. These models utilize two distinct approaches: a microscopic approach, where the nitrogen flux to a single root is considered (Nye and Marriot, 1969; Claassen and Barber, 1976), and a macroscopic approach, where the entire root system as a whole is considered (Dutt et al., 1972; Davidson et al., 1977). In the simplified model proposed here the macroscopic approach for nitrogen uptake is used. Here the Michaelis-Menten approach was used to determine the total N uptake as a function of NH_4-N and NO_3-N concentrations,

$$Q_{NH_4} = \frac{I_{max}\, C}{K_m + (C + Y)} \qquad (4)$$

$$Q_{NO_3} = \frac{I_{max}\, Y}{K_m + (C + Y)} \qquad (5)$$

where Q_{NH_4} and Q_{NO_3} represent the rate of NH_4-N and NO_3-N uptake for the entire rooting system per unit area of the soil surface ($\mu g/hr\ cm^2$), I_{max} the rate of uptake at infinite concentration ($\mu g/hr\ cm^2$), and K_m the Michaelis constant ($\mu g/cm^3$). From equations (4) and (5) the rate of NH_4-N and NO_3-N uptake per unit volume of soil was calculated as a function of the root density distribution R(z) in the soil profile. Therefore, q_{NH_4} and q_{NO_3} of equations (1) and (2) may be expressed as

$$q_{NH_4} = Q_{NH_4} R(z) \Big/ \int_0^L R(z)\, dz \qquad (6)$$

$$q_{NO_3} = Q_{NO_3} R(z) \Big/ \int_0^L R(z)\, dz \qquad (7)$$

where L is the depth of the root zone (cm) in the soil and R(z) is the effective root density distribution, which may be regarded as a measure of the root density (volume or length of roots per unit volume of soil) as a function of soil depth. Such a relationship is an approximate one since root density does not reflect accurately the nitrogen adsorption capacity of individual roots. For our purposes, R(z) will be considered proportional to the root density.

From equations (1) and (2) the transport of NH_4-N and NO_3-N is dependent on the water content θ and water flux v, both of which are variables under transient water flow conditions. The water flow equation for unsaturated soils is

$$Cap(h)\ \partial h/\partial t = (\partial/\partial z)\ (K(h)\ \partial h/\partial z) - \partial K(h)/\partial z - A(z,\theta) \qquad (8)$$

where

h = soil water pressure head (cm)
K(h) = soil hydraulic conductivity (cm/hr)
Cap(h) = soil-water capacity (cm^{-1})
A(z,θ) = rate of water extraction ($cm^3/hr\ cm^3$).

In equation (8) the soil-water capacity, Cap(h), is a measure of the change of soil-water content with suction ($Cap(h) = \partial\theta/\partial h$). This was determined using soil-water characteristic relationships (θ versus h). Equation (8) was chosen over the water diffusivity form (Selim et al., 1976b) because it allows for saturated-unsaturated flow as well as soil stratification. The root extraction or sink term A(z,θ) represents the volume of water uptake per unit volume of soil per unit time ($cm^3/hr\ cm^3$). In this study, the macroscopic approach for the extraction term is used (Molz and Remson, 1970), where

$$A(z,\theta) = T\,R(z)\,K(h) / \int_0^L R(z)\,K(h)\,dz \qquad (9)$$

where T is the evapotranspiration rate per unit area of soil surface (cm^3/cm^2 hr).

The initial and boundary conditions used in conjunction with the water and nitrogen transport equations (1), (2) and (8) were

$$h = h_o(z) \qquad 0<z<\ell,\ t=0 \quad (10)$$

$$h = 0 \qquad z=\ell,\ t>0 \quad (10a)$$

$$v=-K(h)\ \partial h/\partial z+K(h) \qquad z=0,\ t<T \quad (11)$$

$$C = Y = 0 \qquad 0<z<\ell,\ t=0 \quad (12)$$

$$v\ C_s=-\theta D\ \partial C/\partial z+vC \qquad z=0,\ t<T \quad (13)$$

$$0=-\theta D\ \partial Y/\partial z+vY \qquad z=0,\ t<T \quad (14)$$

$$\partial C/\partial z=0,\ \partial Y/\partial z=0 \qquad z=Z,\ t>0 \quad (15)$$

Equations (10) - (15) describe water and solute transport through a soil profile initially ($t = 0$) devoid of nitrogen and having a non-uniform suction head $h_o(z)$ with a water table at depth ℓ from the soil surface. It is further assumed that an ammonium solution having a concentration C_s was applied at the soil surface for a time T. A water flux boundary condition at the soil surface was used (equation (11)), where the water flux v was a function of time, in order to simulate water infiltration as well as redistribution in the soil profile. Equations (1), (2), and (8) are nonlinear partial differential equations and cannot be solved analytically. Therefore, equations (1), (2) and (8) subject to boundary conditions (10) through (15) were solved using numerical analysis techniques. The method of solution was the explicit-implicit finite difference approximation which was successfully used by Selim et al. (1976a) for transient water flow conditions. Finite difference approximations provide distributions of the various nitrogen species as well as water content or suction at incremental distances Δz in the soil and at discrete time steps Δt. A computer program which was written in FORTRAN language was developed to solve the water and nitrogen transport and transformation equations simultaneously. The program is valid for (1) nonuniform initial water and nitrogen content distributions, (2) uniform and layered soil profiles, (3) variable water flux (infiltration or evaporation) or water head at the soil surface, (4) variable kinetic rate coefficients, and (5) variable water. The computer program is flexible and can be adapted to incorporate other nitrogen transformation processes and uptake patterns which influence water and nitrogen transport in the soil.

INPUT SOIL AND PLANT PARAMETERS

Simultaneous transport, transformation, ion-exchange, and uptake of nitrogen in wastewater land treatment systems were simulated using the simplified nitrogen model presented here. For the purpose of model simulation, applied wastewater was assumed to contain 25 ppm of NH_4-N. The wastewater applied per application was 5.0 cm and the schedule of application was either weekly or twice weekly (see Iskandar et al., 1976). Thus 5 or 10 cm of wastewater was applied every week. NH_4-N and NO_3-N distributions in the soil profile, in percolate water, and the amounts taken up by plants were obtained for a simulated period up to 100 days. The soil chosen for this study was a loam soil having $\theta(h)$ and $K(h)$ relationships similar to those proposed by Gardner (1958):

$$\theta(h) = \theta_s \ / \ [1 + (-h/a)^b]$$

$$K(h) = K_s \ \exp\ (\alpha h)$$

where θ_s and K_s are the soil water content and hydraulic conductivity at saturation, respectively. The values chosen for θ_s, K_s and the arbitrary constants a, b and α were 0.4, 0.5, 300, 1.0 and 0.03, respectively. The soil profile was assumed uniform with a bulk density $\rho = 1.6$ g/cm^3 where a water table was located at 150 cm below the soil surface. Initially ($t = 0$), the soil was assumed to be in equilibrium with zero flux condition (i.e. $h = 0$ at $z = 150$ cm and $h = 150$ cm at $z = 0$). Furthermore, the soil was initially devoid of nitrogen. The dispersion coefficient D was 2.5 cm/hr^2 and the Michaelis-Menten constant $K_m = 10$ μg/hr cm^2. The value of I_{max} used ranged from 0.5 to 2.0 μg/cm^2 hr. The rooting depth was 60 cm and the root density distribution $R(z)$ used was that given by Danielson (1967) where 40, 30, 20 and 10% of the total uptake is supplied, respectively, from each quarter (15 cm) of the root zone. A daily average evapotranspiration demand of 0.3 cm/day was assumed. The distribution coefficient K_D was 0.25 cm^3/g. Finally, the kinetic rate coefficients for nitrification and denitrification, k_1 and k_2, were used as a function of soil water suction (h) or water content (θ) at any depth in the soil profile such that $k_1 = \bar{k}_1 F_1$ and $k_2 = \bar{k}_2 F_2$. The functional relationships, F_1 and F_2, were similar to those used by Hagin and Amberger (1974). The values of $\bar{k}_1$ and $\bar{k}_2$ used in this study ranged from 0.01 to 0.1 hr^{-1}.

MODEL SENSITIVITY

Model sensitivity to changes in nitrogen transformation rate coefficients, rate of nitrogen uptake, and amounts and scheduling of wastewater application were examined. The purpose of this sensitivity analysis is to illustrate the dependence of percolate water quality as well as the concentration distributions of NH_4-N and NO_3-N in the soil profile with time on the soil and plant parameters used in the model. Furthermore, a wide range of soil and plant parameters is needed because of their dependence on the soil type as well as environmental factors (pH, temperature, organic matter content, etc.) and plant species.

Figure 2 shows NH_4-N and NO_3-N distributions in the soil profile at selected times. In this case, $\bar{k}_1$ and $\bar{k}_2$ were 0.1 and 0.01 hr^{-1} and I_{max} was 1.0 $\mu g/cm^2$ hr. Following the application of 5 cm of the wastewater, which contained 25 ppm NH_4-N, we see that NH_4-N concentration continued to decrease with time from 0.5 day to 2 days. In addition, NH_4-N did not penetrate beyond 30 cm in the soil profile. After 7 days, i.e. prior to the second wastewater application, only negligible NH_4-N was found in the soil profile. It should be emphasized that for the second application (times 7.5, 8 and 9 days) the NH_4-N distribution with time was similar to that following the first application. Similar NH_4-N patterns following each application were due to the disappearance of all NH_4-N applied within each application schedule (7 days).

The total disappearance of NH_4-N was primarily due to its conversion to NO_3-N through nitrification and to a lesser extent to uptake by plants. The NO_3-N contents shown in Figure 2 result totally from the nitrification of NH_4-N, since the soil was initially (t = 0) assumed devoid of NO_3-N. Due to the continued conversion of NH_4-N to NO_3-N, the concentration of NO_3-N continued to increase in the profile with time, as illustrated by the curves for 1, 2 and 3 days after wastewater application. After 7 days, however, the NO_3-N distribution indicated additional transport at lower depths in the soil profile as well as a decrease in concentration in the top portion (20 cm) of the profile. This NO_3-N decrease was due to uptake by plants, denitrification, and transport in the soil profile.

The NO_3-N distributions of Figure 2 show that unlike NH_4-N there was a substantial amount of NO_3-N in the soil profile prior to the second application of wastewater (7 days). As a result of the second application, NO_3-N concentration with depth continued to increase with time. More important, NO_3-N continued to penetrate to lower depths in the soil profile. This transport of NO_3-N in the soil is illustrated in Figure 3 for a time period up to 14 weeks (98 days), during which time the weekly application of 5 cm wastewater was maintained. Due to this weekly application of wastewater, the NO_3-N concentration continued to increase in the soil profile. However, such an increase became extremely small after 10 weeks of wastewater application. These results would suggest that the NO_3-N distribution is at quasi-steady-state conditions. More significant is the quality of the percolate water seeping to the groundwater table at 150 cm. After 14 weeks, the NO_3-N concentration at the water table was extremely low, less than 1 ppm. Such a low NO_3-N concentration at the groundwater table was a result of NO_3-N uptake as well as denitrification. These two mechanisms of nitrogen removal from the soil solution accounted for the majority of the weekly applied nitrogen after 10-14 weeks of application.

Figure 4 shows the influence of a ten-fold decrease in the rate coefficient for nitrification $\bar{k}_1$, from 0.1 to 0.01 hr^{-1}, on the transport and transformation of NH_4-N and NO_3-N in the soil profile. Due to the decrease in the rate of nitrification, substantial amounts of NH_4-N remained in the soil profile. Moreover, the NH_4-N concentration continued to increase with time throughout the soil profile. As a result, NH_4-N reached the groundwater table depth (150 cm) with a concentration of 0.5 ppm after 14 weeks. In contrast, the NO_3-N distributions with time were substantially lower than those where k_1 was 0.1 (Fig. 3). Such low concentration levels of NO_3-N were a direct result of reduced rate of nitrification.

The influence of a tenfold increase in the denitrification rate coefficient $\bar{k}_2$, from 0.01 to 0.1 hr^{-1}, on NO_3-N distribution is shown in Figure 5. The NO_3-N distribution shows a substantial decrease in the concentration levels through the soil profile. Furthermore, NO_3-N did not penetrate substantially beyond 100 cm depth. NH_4-N concentration distributions (not shown) were similar to those shown in Figure 2.

Plant uptake patterns of NO_3-N and NH_4-N for the various cases considered are shown in Figure 6. With one exception, the NO_3-N uptake was always higher than the NH_4-N uptake. Higher NO_3-N is a direct result of a high concentration of NO_3-N in the soil solution in comparison with NH_4-N. Only when the rate of nitrification was small ($\bar{k}_1$ = 0.01 hr^{-1}) did the NH_4-N uptake with time exceed that of NO_3-N. The low rate of nitrification resulted in lower NO_3-N amounts available for plant uptake in comparison with NH_4-N.

The results shown in Figures 2-6 indicate that the maximum rate of plant uptake I_{max} was taken as 1.0 $\mu g/cm^2$ hr. Figures 7 and 8 show the influence of doubling the value of I_{max} on NO_3-N distribution and the cumulative N uptake with time. A comparison of Figures 7 and 3 shows a substantial reduction in NO_3-N concentration at all depths when I_{max} was 2.0 $\mu g/cm^2$ hr. Figure 8 also shows that for higher I_{max} the total N uptake was consistently higher at all times. After 35 days, 38 kg N/ha was taken up for an I_{max} value of 1, whereas 56 kg N/ha was taken up when the value of I_{max} was doubled.

So far, we have considered the transformations and transport of NH_4-N and NO_3-N when wastewater (5 cm) was applied weekly. In addition, the transformation rate coefficients $\bar{k}_1$ and $\bar{k}_2$ were considered uniform (constant) throughout the soil profile. Figures 9 and 10 show NO_3-N distribution and plant uptake of N, respectively, where two applications per week were considered rather than a weekly application. Here 5 cm of wastewater was applied every 3.5 days. In addition, the coefficients $\bar{k}_1$ and $\bar{k}_2$ were considered as an exponential decay function with soil depth where $\bar{k}_1 = \bar{k}_{1\ max} \exp(-0.01z)$ and $\bar{k}_2 = \bar{k}_{2\ max} \exp(-0.10z)$. The values of $\bar{k}_{1\ max}$ and $\bar{k}_{2\ max}$ were 0.1 and 0.01 hr^{-1} and represent the maximum values of $\bar{k}_1$ and $\bar{k}_2$ which are encountered at the soil surface. Furthermore, I_{max} was taken as 0.5 $\mu g/cm^2$ hr. Exponential decay functions for the rate coefficients K_1 and K_2 were used in order to illustrate the decrease of these parameters with soil depth (Ardakani et al., 1974). As a result of doubling the weekly rate of wastewater application, the concentration of NO_3-N in the soil profile continued to increase drastically with time. Moreover, significantly high NO_3-N concentrations reached the groundwater table only after 4 weeks of application. It is suggested that the low rate of denitrification, especially at soil depths below 20 cm, also contributed to the high effluent NO_3-N concentrations.

SUMMARY AND CONCLUSIONS

A simplified mathematical model was developed to describe transformations and transport of nitrogen under transient soil water flow conditions. Kinetic reactions were assumed to govern the nitrification and denitrification processes. A macroscopic approach was used to incorporate plant uptake of water as well as NO_3-N and NH_4-N from the soil solution. The sensitivity of the model to changes in rate of N transformation, N uptake by plants, and schedule and amounts of N application were also investigated. The model can be used as a tool to predict the fate of nitrogen in land treatment systems, estimation of N application load, plant uptake of N, and as a research or management model for N behavior in land treatment. The model is flexible and can be adapted to incorporate various nitrogen transformation mechanisms as well as layerings in the soil profile.

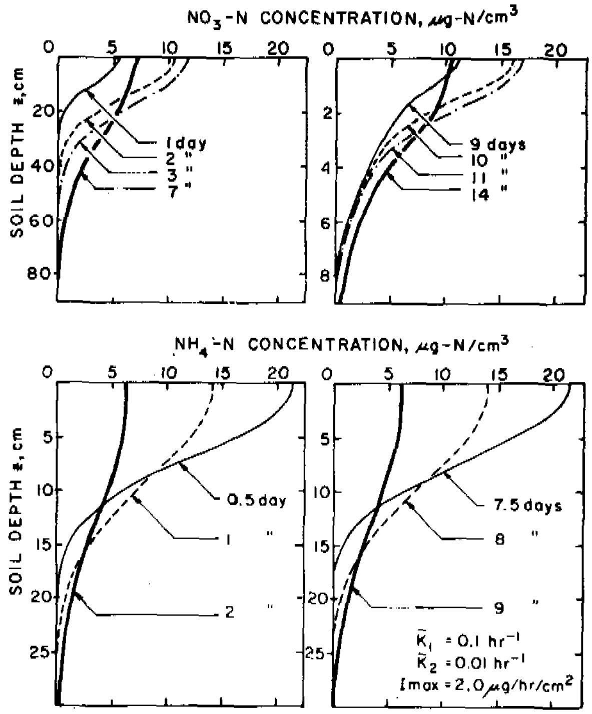

Figure 2. Simulated concentration distributions of NO_3-N and NH_4-N in a loam soil profile with a water table at 150 cm. Soil and plant parameters were $\overline{K}_1$ = 0.1 hr^{-1}, $\overline{K}_2$ = 0.01 hr^{-1}, and I_{max} = 1.0 µg/hr cm^2.

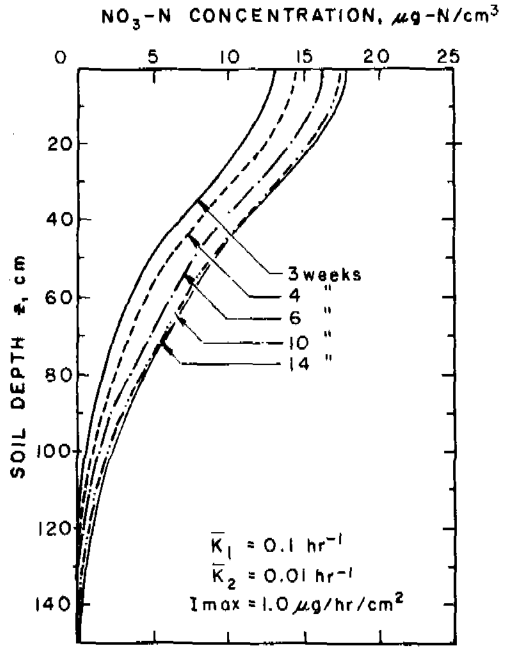

Figure 3. Simulated concentration distributions of NO_3-N in the soil profile where weekly applications of 5 cm of NH_4-N solution were maintained.

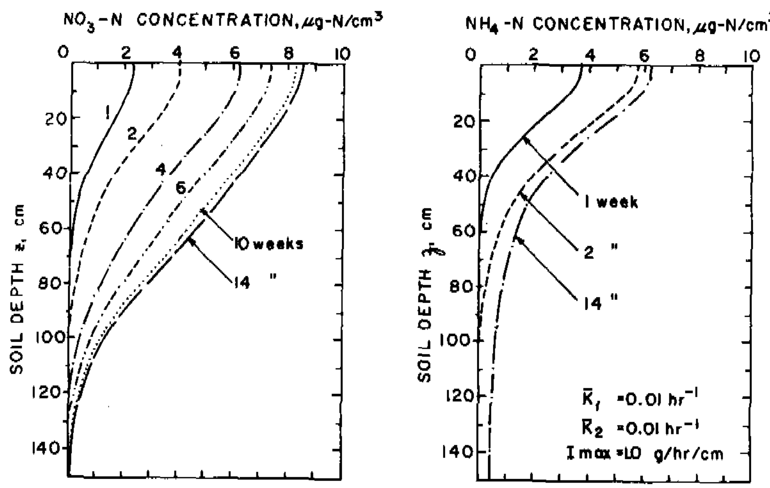

Figure 4. Simulated concentration distributions of NO_3-N and NH_4-N in the soil profile for the case where $\overline{K}_1$ and $\overline{K}_2$ were 0.01 and 0.01 hr^{-1}, respectively.

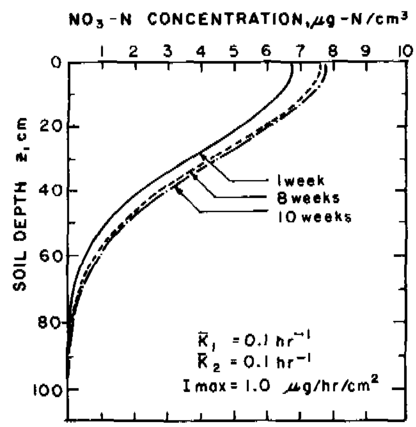

Figure 5. Simulated concentration distributions of NO_3-N in the soil profile for the case where $\overline{K}_1$ and $\overline{K}_2$ were 0.1 hr^{-1}, respectively.

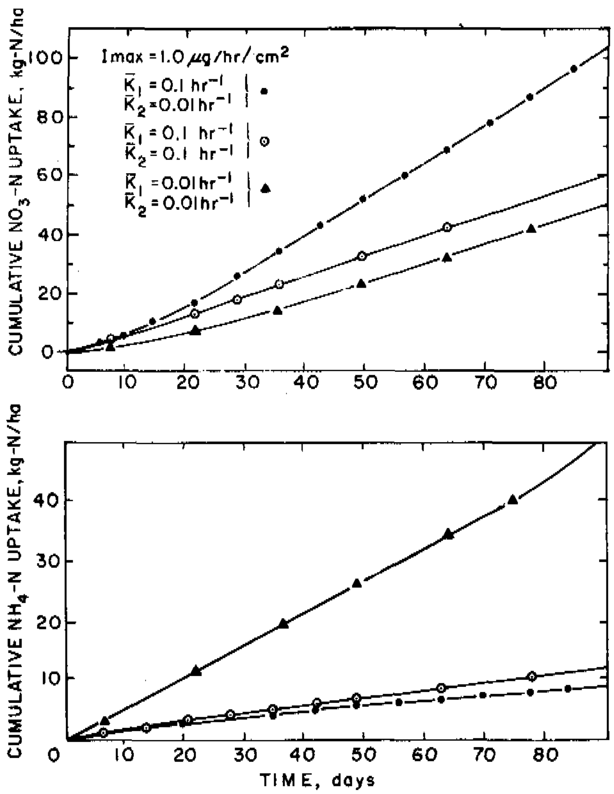

Figure 6. Cumulative NO_3-N and NH_4-N uptake with time for three cases where I_{max} was 1.0 μg/hr^3 cm^3.

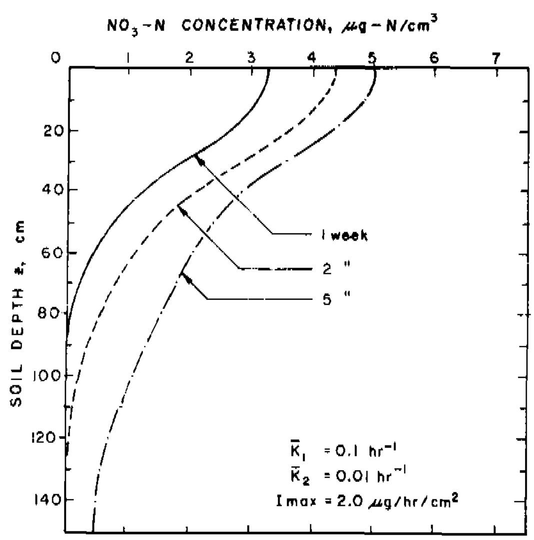

Figure 7. Concentration distributions of NO_3-N in the soil profile where $\bar{K}_1$ = 0.1 hr^{-1}, $\bar{K}_2$ = 0.01 hr^{-1} and I_{max} = 2.0 μg/hr cm^2.

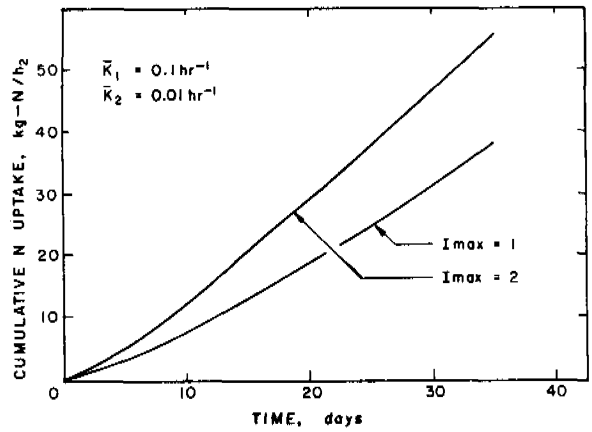

Figure 8. Total N uptake with time where I_{max} = 2.0 μg/hr cm^2.

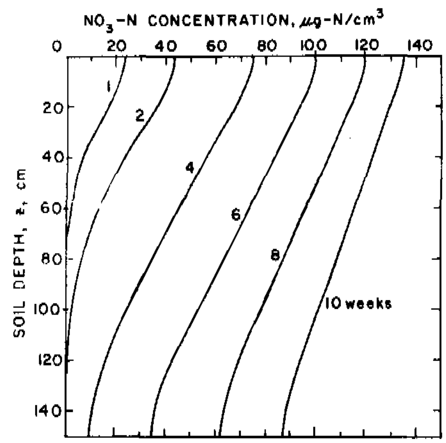

Figure 9. Concentration distributions of NO_3-N in the soil profile where two applications of NH_4-N solution per week were maintained.

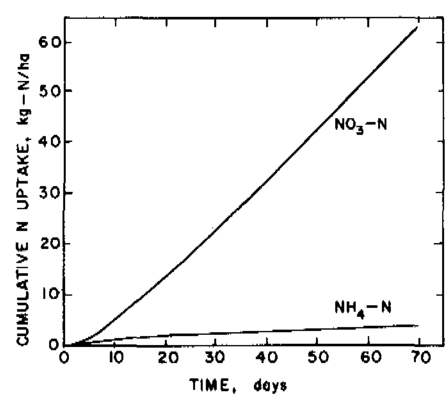

Figure 10. Cumulative NH_4-N and NO_3-N uptake with time where two applications of NH_4-N per week were maintained.

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Evaluation of Plant Uptake Models for Prediction of Water Quality of Land Treatment Sites

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ABSTRACT

Models are developed to meet various objectives; therefore the first problem of modeling is to define objectives clearly. Models have been developed to simulate diffusion and mass flow of nutrients to plant roots, the uptake of nutrients by roots or micorrhizae, root growth, translocation of nutrients from roots to above ground plant parts, plant growth and incorporation of nutrients into plant tissue, and harvesting strategies. These models can be adapted for land based waste water treatment problems. After modeling objectives have been established, the required precision must be balanced against cost, time, and personnel constraints.

INTRODUCTION

Models are developed to meet various objectives. Possible objectives are: to expedite discussion among researchers and to guide research, to keep track of carbon, water, or nutrients in an ecosystem, to predict events based on past behavior, to design new equipment, or to test management alternatives. The first problem of modeling is to define the objectives clearly. The situation being considered here involves models for bookkeeping of water and nutrients. The question is still, on what system are we keeping books? We propose that two themes involving water and nitrogen are implicit in discussion of land treatment sites: (1) keeping books on the amount of water and nitrogen in the soil or soil water and (2) keeping track of the amount of water and nitrogen in the plant-soil system. For the first objective plant uptake can be considered merely as a sink, removing nutrients at some rate. For the second objective, the bookkeeping is more complex and involves first the transfer of nitrogen from the soil to the aboveground plant parts and second the removal of nitrogen from the system at the most rapid rate possible by harvesting the plant tops, the whole system operating in some root-benefit balance.

Management of nutrient levels can be considered in several stages: the first is the supply and delivery of the nutrient to the plant which involves mass flow, diffusion, and root growth; the second stage is the uptake of the nutrient by the root or mycorrhizae; the third is the translocation of the nutrient from roots to above ground parts; and the fourth is the incorporation of the nutrient into plant tissue. The last stage involves harvesting strategies to maximize nutrient removal for the duration of the land treatment system. Models exist for all stages. The time scales of these models range from minutes to years. Some are simple enough to allow analytical solutions, others are complex and require numerical solutions. As the system becomes more complex, numerical solutions are more commonly used.

Models of Diffusion and Mass Flow to Plant Roots

Models organized around solutions of diffusion and mass flow differential equations have been reviewed by Olsen and Kemper (1968). The equation solved by the models, in radial coordinates, is generally of the form:

$$(B+\theta)\frac{\partial c}{\partial t} = \frac{1}{r}\frac{\partial c}{\partial r}\left[rD_p\frac{\partial c}{\partial r} + va\theta C\right] \qquad (1)$$

where: B = slope of the adsorption isotherm ($\mu g\ cm^{-3}$ soil/$\mu g\ cm^{-3}$ soil solution), θ = volumetric water content of the soil ($cm^3\ cm^{-3}$ soil), C = concentration of the nutrient in solution ($\mu g\ cm^{-3}$ soil solution), t = time (sec), r = radial distance (cm), D_p = porous diffusion coefficient ($cm^2\ sec^{-1}$), v = average macroscopic velocity of soil water at the root surface ($cm\ sec^{-1}$), and a = radius of the root (cm).

The first term in the parenthesis of the right hand side of eq. 1 represents the contribution of diffusion to nutrient supply, the second represents supply by mass flow of water. The equation is solved subject to the boundary conditions that at time t = 0, $C = C_0$ where C_0 is a constant at all distances from the root. For all t>0, there is no flux of nutrient across the outer boundary described at the edge of the effective nutrient absorption zone of the root. The inner boundary condition is usually described by one of the following conditions: 1) Nutrient flux across the root surface is proportional to nutrient concentration at the surface: $F = \alpha C$ where α is termed the absorption capacity of the root (Passioura, 1963; Nye, 1966; Marriott and Nye, 1968; Nye and Tinker, 1969; Bar-Yosef et al., 1972; and NaNagera et al., 1976). 2) Concentration of the nutrient at the root surface is 0, i.e. the root is a perfect sink, (Zartman et al., 1976) or constant. 3) The rate of nutrient uptake is constant: $F = k$ (Olsen and Kemper, 1968). 4) The concentration at the root surface decreases exponentially with time (Lewis and Quirk, 1967). Comparative results based on different boundary conditions and assumptions are given explicitly in Olsen and Kemper (1968) and Zartman et al., (1976). Solutions for the analogous heat flow equations are given in Carslaw and Jaeger (1959).

The following assumptions are implicit in the diffusion-mass flow models for which analytical solutions have been derived. Many of the assumptions can be relaxed considerably when numerical methods are used. 1) Plant roots are long cylinders with measurable radius. 2) Roots are uniform along their entire length for nutrient uptake; α is not affected by the age of the root or the presence of other ions. 3) Roots are far enough apart that there is no overlap of absorption zones. 4) The soil is an homogeneous isotopic medium with uniform initial nutrient content, water content, and with soil water velocity slow enough that hydrodynamic dispersion does not occur. 5) D_p and B are constant for the time and conditions being modeled. 6) The concentration at the root surface can be approximated by concentration in the bulk solution when the flux equations are derived from formal solutions. 7) Reaction with the solid phase if it occurs, is instantaneous, reversible and linear.

Zartman et al., (1976) have demonstrated that models for nitrate uptake based on boundary condition 1 underestimate nutrient accumulation, and those based on boundary condition 2 with the root as a perfect sink overestimate accumulation. A constant uptake rate may be applicable for young active roots in the initial stage of uptake, but it is likely that for longer term absorption the appropriate boundary condition is that of a constant concentration at the root surface.

In models developed by Anderssen et al., (1969), Passioura (1963), and Gardner (1968), growth of roots into new areas of soil has been included as a third mechanism for supplying nutrients. Essentially the method involves portraying the root as a moving line sink, or a group of roots as a moving front sink. Anderssen et al. solve for surfaces of equal concentration around an extending root; Gardner offers solutions in one, two, and three dimensions for equations of a moving sink; and Gardner and Passioura give solutions similar to those by Carslaw and Jaeger (1959) for the same boundary conditions.

The choice of equation and necessary boundary conditions again depends on the objective of the modelling effort. Is the interesting question what happens around an individual root, or does the real issue lie in what happens over the entire profile--what drains out the bottom, what is moved to plant tops, and what is left in the soil? If the objective is to define regions of nutrient depletion around a root, the theoretical diffusion and mass flow equations are suitable. However, when approached from a wider viewpoint, i.e. the whole plant system, it is obvious that supply and delivery are only the first steps in nutrient acquisition and such models must be coupled with models dealing directly with plant properties in relation to nutrients.

In addition it is necessary to assess the appropriateness of the mathematical formulation to the site at hand. Are diffusion and mass flow of equal importance, or is one clearly more important and the other able to be ignored? Phillips et al. (1976), show that when the ratio ν/α is < 0.2 the importance of diffusion with respect to mass flow is large, but at ratios $\nu/\alpha > 1.0$ mass flow is the dominant process. At low evapotranspiration the problem clearly becomes one of diffusion. A second question is the relationship of mass flow to transpiration. Soil water velocity and transpiration rate are often coupled in the equation $Ts = 2\pi a \nu \theta$. The equation essentially states that the only driving force for soil water movement is transpiration. In a nearly saturated soil such as the tussock or wet meadow tundra, or in systems where irrigation is practiced, a significant amount of nutrients may be carried in vertical or horizontal movement of water through the profile.

A third question to consider is the trade-off between an elegant mathematical treatment and the acceptance of theoretically questionable conditions such as linear absorption, constant root uptake properties and instantaneous, reversible reaction with the solid phase. A greater flexibility in approach may be achieved with systems models which do not rely on formal solution but which allow control of transfers by rates which may change continuously as a result of the interaction of plant or soil characteristics.

Models of Root Uptake

Studies on nutrient uptake by excised roots in solution have demonstrated that nutrient uptake as a function of external concentration often follows a Michaelis-Menton relationship (Epstein and Hagan, 1952; Carter and Lathwell, 1967). Uptake is then expressed as a function of solution concentration, maximum velocity of uptake and a half reaction constant.

$$U = \frac{V_{max}C}{k_m+C} \qquad (2)$$

The equation can be adjusted for competition with other ions as shown in Epstein and Hagan (1952) and Frere and Ashley (1964). In addition it is possible to extend the Michaelis-Menton equation to include more carrier systems simply by determining the appropriate constants and summing the uptake by each carrier.

Michaelis-Menton uptake equations are used in ecosystem level models (Reuss and Innis, 1977; Cole et al., 1977) in which delivery by the soil system is handled in a separate set of equations. Both the above models incorporate two carrier systems. To prevent excess nutrient buildup in roots during the simulation reverse flow equations are utilized at high root nutrient concentrations. Coefficients are applied to reduce the effective concentration of the soil solution as a result of reaction of the nutrient with the solid phase.

The chief limitation of using Michaelis-Menton equations in field conditions lies in extrapolation of results from solution experiments. Frere and Ashley (1964) did demonstrate however that uptake of cations is largely from the solution phase and that the solid phase ions may be envisioned as a nutrient reservoir.

Michaelis-Menton kinetics may be incorporated as the inner boundary condition for the mass flow and diffusion models. At low nutrient concentration the single carrier equation reduces to boundary condition 1 of the preceeding section, allowing an analytical solution. It is more likely however that nutrient uptake is not governed by a single carrier system, and in a waste water treatment area it is questionable whether the reduced form of the kinetic equation is appropriate due to the frequent charging of the soil profile with nutrients.

Models of Root Growth

Models related to net assimilation rate (Williams, 1946) have been developed by Loneragan (1968), Elgawhary et al. (1972), Mengel and Barber (1974), and Jungk and Barber (1974). The models assume that root length (or weight) and uptake are linearly related over time. The nutrient flux for the entire plant is calculated for a particular nutrient by:

$$I_m = \frac{1}{W_r}\frac{dM}{dt} \qquad (3)$$

where: I_m is the nutrient flux, W_r is the weight of roots, and dM/dt is the change in nutrient content of M in the plant with time. The mean uptake rate may be calculated by:

$$I_s = \frac{\ln W_2 - \ln W_1}{t_2 - t_1} * \frac{M_2 - M_1}{W_2 - W_1} \qquad (4)$$

where subscripts refer to harvest, and M is the amount of a particular nutrient in the plant.

Mengel and Barber (1974) and Jungk and Barber (1974) use the equation with root length substituted for root weight.

Equation (4) is most suitable for roots in the stage of exponential growth. As roots die back, mean root length between successive harvests is used instead of the actual length at harvest. Growth models demonstrate the changing nature of uptake characteristics as roots age and are most suitable for annual crops where there are no substantial nutrient reserves other than in the seed. If significant reserves exist, as in perennial species, the relationship between uptake and flux is more complex.

The major limitation of models based on net assimilation rate is that uptake is not directly coupled to soil processes. The growth models place the emphasis on the plant part of the plant-soil system. Waste treatment models focussed on growth models would visualize the plant as a gross sink for nutrients. Microprocesses of supply or kinetic uptake would only be expressed by a different level of approach.

Models of Translocation

The mechanisms of translocation have been studied for a long time but are still not understood. Models of translocation of carbon and nutrients in relation to plant growth and development have been either detailed and mechanistic (Horowitz, 1958; Dainty, 1965) or descriptive (Warren Wilson, 1964, 1966; Blackman, 1968; Kvet et al., 1971). Thornley (1972a, b), Miller et al. (1977c), and Stoner et al. (1977c) considered translocation to be driven by a concentration gradient impeded by resistances. Brouwer and de Wit (1969), de Wit et al. (1970), and Fick et al. (1975) bypass the problem by pooling all reserves into one plant pool which is drawn upon for growth by each plant part. All formulations ignore the possibility of sugars being transported against a concentration gradient which has been observed.

Models of Water Utilization

The factors which define the interaction of the vegetation and the aboveground environment, especially involving water and CO_2 gas transfer, are most clearly understood. These include radiation, turbulent transfer, plant water relations, and canopy photosynthesis. Radiation models which have a broad base of usage in temperate zone crops have been developed from concepts of light penetration in continuous vegetation canopies (Monsi and Saeki, 1953; Davidson and Philip, 1958; de Wit, 1965; Duncan et al., 1967; Anderson and Denmead, 1969; Yim et al., 1969; Acock et al., 1970; Idso and de Wit, 1970). The model of Miller et al. (1976) and Miller and Stoner (1978) calculates infrared radiation within the canopy, a component not found in other models which is significant in affecting leaf and ground surface temperatures and evaporation rates, and includes the effects of stems on the radiation distribution within the canopy, a factor which has rarely been considered but is important in woody vegetation. This model was developed in the red mangrove vegetation of south Florida (Miller, 1975), and applied to the canopies of the arctic and alpine tundra (Tieszen et al., 1975; Miller et al., 1976) and to the canopy of the chaparral (Lawrence, 1975; Miller and Mooney, 1974). A model of radiation in non-homogeneous canopies and individual shrubs was developed for the chaparral (Roberts and Miller, 1975).

Models of the turbulent transfer of heat, water vapor and CO_2 in the canopy have been proposed by Denmead (1964), Waggoner and Reifsnyder (1968), Waggoner et al. (1969), Murphy and Knoerr (1970, 1972), and Stewart and Lemon (1972) and tested with data from red clover, millet, pine, and corn canopies with reasonable success. In Miller et al. (1976) and Miller and Stoner (1978) profiles of air temperature and humidity between the top of the canopy and a reference height above the soil surface are calculated with algorithms proposed by Waggoner and Reifsnyder (1968) and Waggoner et al. (1969). The canopy microclimate predictions have been tested on a preliminary basis on chaparral, mangroves and arctic tundra with fair success (Miller et al., 1974; Miller, 1975; Miller et al., 1976; Tieszen et al., 1976).

In Miller et al. (1976) and Miller and Stoner (1978), leaf and stem temperatures are calculated from the energy budget equation for individual leaves (Gates, 1962, 1965). The model estimates the physical processes of solar and infrared radiation absorption, infrared loss from the leaves, convectional exchange, and transpirational exchange. Leaf temperatures have been tested against measured temperatures in mangroves (Miller, 1972) and chaparral (Lawrence, 1975). Energy exchange processes and temperatures of plant parts are calculated for sunlit and shaded parts at different levels in the canopy.

Models of plant water relations have been proposed by Honert (1948) and Rawlins (1963), and described in non-mathematical terms by Jarvis and Jarvis (1963). As concepts have become refined, the formulation has become more complex

(Cowan, 1965; Philip, 1966). These models, drawn as electrical analogues, attempt to explain the regulation of water movement from the soil through the plant and into the atmosphere by plant and environmental variation. These models have been tested only partially because of the difficulty in measuring simultaneously all of the necessary parameters. The model of plant water relations of Miller et al. (1976), and Miller and Stoner (1978) allows for non-equal rates of water uptake and loss and was first developed for the mangroves (Miller, 1975) and evaluated and used in the alpine tundra (Ehleringer and Miller, 1975a, b), the arctic tundra (Stoner and Miller, 1975) and the chaparral (Fig. 1) (Miller and Mooney, 1974; Miller and Poole, in litt.).

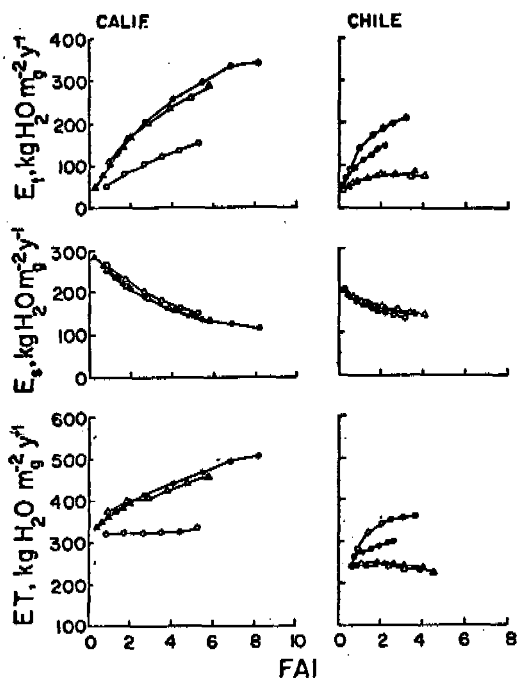

Figure 1. Transpiration, soil evaporation, and evapotranspiration calculated for pure stands of four species in California and Chile, with increasing foliage area indices.

Models of canopy photosynthesis are usually based on the photosynthetic relations of a single leaf within the canopy. Monsi and Saeki (1953) and Davidson and Philip (1958) used a simplified model to calculate stand photosynthesis. de Wit (1965) used a more realistic model to calculate the annual course of potential production for different regions. Monteith (1965), Anderson (1966), Duncan et al. (1967) and Lemon (1967) proposed a series of refinements to the basic concepts. Miller and co-workers developed a canopy photosynthesis model for the mangroves (Miller, 1972) and used it to indicate the daily and seasonal progression of production and water use efficiency in chaparral, mangroves, and tundra (Miller and Mooney, 1974; Miller, 1975; Miller et al., 1976; Miller and Stoner, 1978).

Energy exchange processes at the soil surface have been modeled by Goudriaan and Waggoner (1972) and Denmead (1973), using simplifying assumptions about the moisture status of the soil. A more complex version of the energy budget model has been used for work in chaparral, mangroves and arctic tundra (Miller et al., 1974; Ng and Miller, 1975, 1977). Soil temperatures may be simulated from a known surface temperature (Goudriaan and Waggoner, 1972; de Wit and Goudriaan, 1974), or directly from air temperatures (Hasfurther and Burman, 1974).

The movement of water in the soil was calculated by Hanks and Bowers (1962), Liakapoulos (1965), Amerman (1971), and de Wit and Goudriaan (1974). A soil water model based on the same principles was tested and developed in chaparral to estimate evaporation, deep percolation and transpiration losses (Ng, 1974).

Models of Plant Growth and Nutrient Incorporation

Models providing an integrative framework of plant growth processes are still rudimentary (Loomis, 1970), even for plants with agricultural importance. Monsi and Murata (1970) showed by simple calculations how the allocation of carbon to productive tissues, rather than to absorptive tissues, can affect production. Brouwer and de Wit (1969) proposed a model of plant growth in which allocation to shoots or roots was controlled by the availability of carbohydrate reserves and water. de Wit et al. (1970) elaborated upon this model. The model was developed for agricultural crops and compared against data on corn in particular. Thornley (1972a, b) proposed general models describing the growth of shoots and roots by logistic equations. Bodkin (1975) developed a model of the growth of several forest trees, describing the process of succession in a northeastern hardwood forest using as simple formulation as possible to describe the responses to light and water. Fick et al. (1975) describe a model for the growth and carbohydrate allocation in sugar beets. Penning de Vries et al. (1976) present a model of nitrogen dynamics and growth of loblolly pine over a multi-year time interval. Van Keulen (1975) describes a model of the growth of annual grasses in an arid region, which was extended to include nitrogen (van Keulen et al., 1975). Harpaz (1975) included the nitrogen dynamics for an arid region in a model similar to van Keulen's. However, these models include only a superficial treatment of nitrogen dynamics in the plant.

Three models of growth and allocation, with different levels of sophistication, were developed for the single-shooted graminoid, during the U.S. IBP Tundra Biome Research (Miller, 1972; Miller et al., 1978c; Stoner et at., 1978c; Lawrence et al., 1978). The models are driven by the seasonal patterns of growth in various structures. Growth is affected by temperature and plant sugar, nitrogen, phosphorous, and calcium status. Carbon and inorganic nutrients associated with the growth of new tissue are calculated from the composition of young mature tissue. Growth respiration is calculated from the respiratory costs of creating different biochemical constituents (Penning de Vries, 1972, 1973, 1974; Penning de Vries et al., 1974). A similar model has been developed for an evergreen shrub (Ledum palustre), a deciduous shrub (Salix pulchra), and a tussock graminoid (Eriophorum vaginatum) (Stoner et al., 1977b).

Miller et al. (1978) describe a model of plant and soil processes in mediterranean scrub ecosystems. The processes include the movements of and controlling influences on carbon, water, and nitrogen. The model has been parameterized for four shrub species and two grass species. The species are grouped singly or together in various combinations to simulate the seasonal progression of water loss from the soil surface and plant tissue; carbon uptake; growth of leaves, stems, and roots; death of leaves, stems, and roots; nitrogen uptake, mineralization, and decomposition (Fig. 2). The model will also estimate the annual totals of various quantities of interest to the investigator and search for combinations of

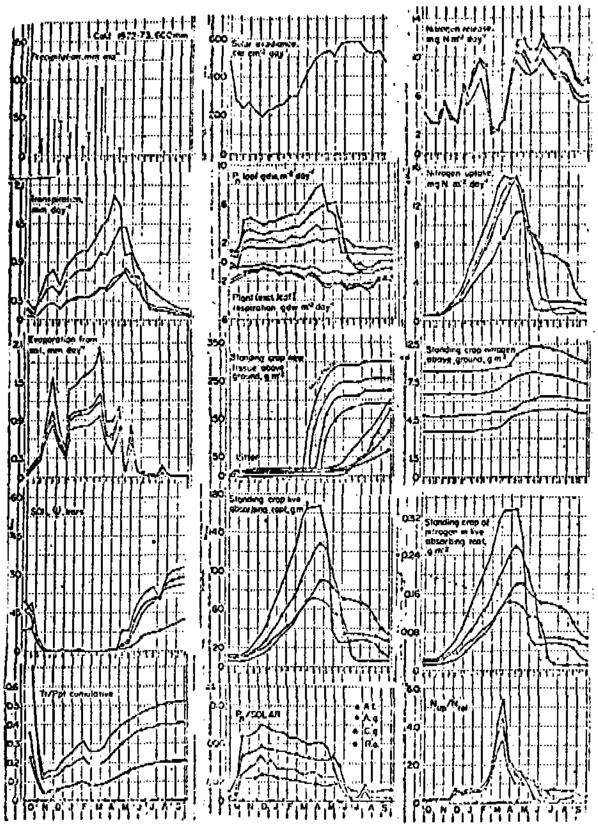

Figure 2. Seasonal progressions of the resources available to the vegetation, precipitation, solar irradiance, and nitrogen released by mineralization; capture of the resource by the vegetation; transpiration; net photosynthesis, and nitrogen uptake; other related processes; and resource use efficiencies; transpiration over precipitation, net photosynthesis over solar irradiance, and nitrogen uptake over nitrogen release.

species and plant growth forms which use light energy, water, and nitrogen maximally (Fig. 3) (Miller et al., 1978). The plant process models have been parameterized and combined with soil thaw and decomposition models to simulate the saturated and nearly saturated conditions of the arctic tundra (Miller et al., unpubl.).

Models of Harvesting Strategies

Davidson and Phillip (1958) exploited their model to define harvesting strategies to maximize production. The tradeoffs are between how much and how often a crop is removed by harvesting, maintaining enough leaf area for rapid regrowth, and allowing too much leaf area for high production rates.

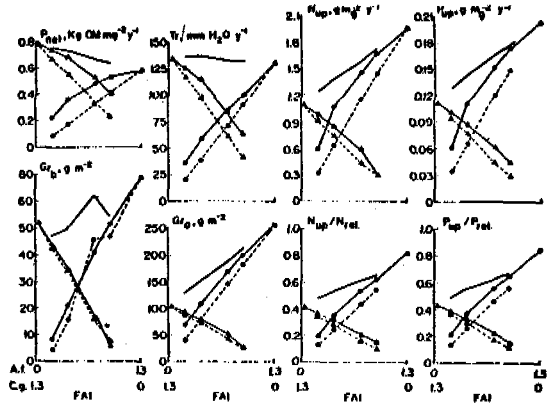

Figure 3. Resource use and resource use efficiencies for two species replacement series using Adenostoma fasciculatum (A.f.) and Ceanothus greggii (C.g.). The abscissa gives the foliar area index (FAI) for species mixes used in the simulation.

SUMMARY

Models of land based waste water treatment can be constructed for any one aspect of the problem, from microprocesses in the soil to a harvesting scheme to maximize nitrogen harvest. Models constructed to predict long range trends may not be suitable for small time spans. Conversely, models suitable for small time spans may not be applicable to long range use. A model can be built to monitor nitrate per se in the system, or the model can be expanded to include phosphorous or heavy metals. A model can be designed to describe soil processes in detail with the plant as a black box sink, or the biochemistry of nutrient uptake and allocation can be modelled with the soil a black box source. A model can incorporate details of both the plant and the soil systems in an integrated, interrelated manner. In short, models exist or can be developed for most applications. The major problem in model development is establishing the objectives and necessary precision, balanced by cost, time and personnel constraints. Once the objectives and required precision are defined, the adequacy of the data base and research plan can be evaluated.

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EVALUATION OF EXISTING SYSTEMS

OVERVIEW OF EXISTING LAND TREATMENT SYSTEMS

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ABSTRACT

This paper reviews existing systems of land application of wastewater. Particular emphasis is placed upon the historical philosophy of the utilization of the natural soil-plant system for purifying wastewater, reasons for the success or failure of the older systems, and experience gained from their design, construction and operation.

The first documented systems were built in 1559 in lower Silesia, Poland, and in Buzlau, Germany. The practice of applying wastes to land then spread throughout Europe. Early systems were constructed to improve stream water quality and to fertilize the soil. The first system in the U.S. was built in Augusta, Maine, in 1872.

The number of sewage farms began to decline at the start of the 20th century, mainly because of urbanization and the increase in the cost of transporting wastewater. The decline was not due to any documented case of health problems associated with the utilization of wastewater in agriculture.

Today, the purposes of land application of wastewater are to treat the wastewater, to utilize its nutrients in agriculture, to provide for its reuse as a water resource for arid and semi-arid regions, to help solve the energy shortage, and to eliminate the pollution of natural waters.

Existing systems can be utilized as demonstration sites, for public acceptance purposes, for gaining experience in design criteria, management, and operation practices, and for assessment of the long-term effects on the soil-plant systems.

INTRODUCTION

Population growth and urbanization in recent years have resulted in increased production of municipal and industrial wastes which must undergo treatment and disposal. Also, demands for drinking and irrigation water have increased in arid and semi-arid regions. At the same time, more stringent water quality legislation has been enacted to prevent eutrophication of lakes and streams, and to protect human health from an increasing number of potentially toxic chemicals being released into the environment. Concurrently, the realization that our present supply of potable water and energy resources is not infinite has led to a reevaluation of wastewater treatment practices. Emphasis has been placed on reclaiming this water resource, while utilizing a much more cost effective method.

Land treatment systems, given the proper design and management, can achieve a level of treatment equal to or even higher than an advanced waste treatment plant, at a lower energy and cost level. Although there are many benefits to the reutilization of wastewater, describing them is not the main objective of this paper. Its purpose is to summarize the historical philosophy of the utilization of the natural soil-plant system for purifying wastewater. Reasons for the success or failure of these old systems are documented, and experience gained from the design, construction, and operation of new land treatment systems is discussed.

HISTORICAL OVERVIEW

The application of wastewater to land is not a new concept; it has been reported as far back as ancient Athens (Markland et al., 1974). The first documented case occurred in 1559 when Boleslawiec, in lower Silesia (Wierzbicki, 1949), became the first European city to forbid the direct discharge of sewage into nearby waterways. During the same year, a sewage farm was established in Bunzlau, Germany (Pound and Crites, 1975). The practice of applying wastes to the land spread slowly throughout Europe. Not only was it found to be the simplest method for the treatment and disposal of wastewater, but it also proved to increase the fertility of the land when used for crop irrigation. This, in turn, increased both crop yields and the value of the land. For a detailed historical survey of land application of sewage effluent, the reader should consult Iskandar and Keefauver (1978), Jewell and Seabrook (1978), Sullivan et al. (1973), and the Center for the Study of Federalism (1972). The report by Jewell and Seabrook was not available for evaluation at the time this paper was submitted for publication. However, the senior author (Dr. W. Jewell) gave the reference for citation and indicated its availability in the near future as an EPA publication (personal communications).

The growth of industry in England during the early 18th century, and the subsequent pollution of waterways, forced the English to consider the necessity of wastewater treatment. Early land application systems were established in order to reach two goals: first, to improve stream water quality, and second, to fertilize the soil (Ewert et al., 1973). The first facility was a slow infiltration system built outside London in 1829 (Hyde, 1929). In 1843, sewage farms were established in Edinburgh, Ashburton and Devon.

Treatment of sewage in the expanding European cities became an increasing health problem. The outbreak of such infectious diseases as typhus provided the impetus for the establishment of a treatment system in Berlin (Weiland, 1955). Studies conducted in the early 1870's demonstrated the effectiveness of land treatment in the removal of pathogens. Tests conducted by the Veterinary College of Berlin showed that no ill effects resulted from feeding sewage-irrigated fodder to cattle. The system, which initially consisted of 2,000 hectares, began operating in 1878. At first, because no pretreatment was given to the wastewater, the suspended solids in the wastewater affected the crops. Settling basins were installed to alleviate this problem. Since its initiation, the Berlin sewage farm has played a major role in the development of land application techniques.

Following the English design, a sewage farm was established in the 1890's at Braunschweig, Germany (Tietjen, 1977). This early system encompassed 350 hectares. Raw sewage from the city was piped to the farm and applied by flooding. Over several decades, Braunschweig expanded to the point where the farm could no longer handle all of the city's wastewater. Recognizing the high degree of purification that could be obtained from the sewage farm, the citizens enlarged it to twice its original size, and it began to serve two additional nearby cities, Wolfenbüttel and Salzgitter.

Another major land treatment system was begun in the 1890's. Located outside Melbourne, Australia, the Werribee Sewage Farm was destined to become one of the oldest and largest operating land treatment systems (Seabrook, 1975). The idea arose from a recommendation made by an English engineer, James Mansergh. In fact, out of eight proposals for wastewater treatment and disposal, five involved land treatment. The original farm consisted of 3,580 hectares. Operations began in 1897, when the first raw sewage flooded the fields by the overland flow mode. Today, approximately 2,000 hectares consist of a fine sandy loam with an infiltration rate of 25 mm/day. The remaining land has a much lower permeability (2.5 mm/day) and a gentle slope.

Sewage farms became common in the second half of the 19th century. In 1872, the first one in the United States was established at Augusta, Maine (Pound and Crites, 1975), and by the end of the century, there were systems in 20 states.

Land application of sewage effluent has been practiced in many other countries as well. It has been reported in

India (Arceivala, 1977), Israel (Shelef, 1977), Japan (Kubo and Sugiki, 1977), South Africa (Hart and Van Vuuren, 1977), China (Tsu-Yuan, 1965), Canada (EPS, 1976), Mexico (Aguirre and Eloy Urroz, 1977), New Zealand (McDowali, 1958) and the Netherlands (Beck and De Haan, 1973). For a complete listing the reader should consult Iskandar and Keefauver (1978).

At the start of the 20th century, increasing urbanization began to cause a decline in sewage farms in both Europe and America. With the growth of cities, land not only became more expensive, but there was less suitable land available. This made it necessary to move further from the cities, which, in turn, increased transportation costs. As populations increased, old fields became overburdened by the increasing amounts of waste. This resulted in poor treatment of the sewage, as well as poor crop yields. The combination of these factors led to a decline in the number of sewage farms. By 1900, the method was considered outdated, and nutrients from wastewater were found to be more costly than commercial fertilizer (Center for the Study of Federalism, 1972).

Although some sewage farms continued to operate successfully, a shift was made toward physical-chemical and biological treatment. This shift was clearly demonstrated by the geographical distribution of land treatment systems in the United States. By 1935, there were well over 100 systems in the U.S. (Markland et al., 1974). The vast majority of these systems were located in the western portion of the country. Denser populations, higher land values, and little need for crop irrigation provided small incentive for the continuation of such systems in the east. Water supplies in the west, however, were limited, and the demand for irrigation was high. Here wastewater represented an additional supply of water. There were over 100 communities in the west which utilized wastewater for irrigation, 67 of which had municipally owned sewage farms (Hutchins, 1939). By 1971, the total number of land application systems in the United States exceeded 500 (Center for the Study of Federalism, 1972). Again, most of them were in the western and southern regions of the U.S.

There was concern over the possibility of disease arising from the irrigation of crops with raw sewage. This led states to require pretreatment, even though it was recognized that many streams which were utilized for irrigation purposes were also the dumping sites for untreated wastes. The direct discharge of raw sewage into streams and lakes was not uncommon (Lawton, 1960). In regions where stream flow was naturally intermittent, raw sewage composed a major portion of the stream flow during certain times of the year. For example, in 1934 at Ogden, Utah, Hutchins (1939) found that:

> The Odgen sewage, which was discharged untreated into the Ogden River, ..., represented more than one-half of the water being pumped by two irrigation companies...

Even taking into account the potential health risk and the negative public attitude, it was felt that the rising demand for irrigation water would and could be alleviated by increased use of treated wastewater. In general, there is a good similarity between the reasons that forced the British Royal Commission on Sewage Disposal to press for the universal revival of land application of sewage in 1865 and today's pressure from U.S. regulatory agencies for land treatment as a viable alternative to advanced waste treatment. Nevertheless, it is improbable that land treatment will attain universal application in the United States due to the high variability in climate, soil type, hydrology, topography, and state regulations.

EXISTING SYSTEMS AND HEALTH ASPECTS OF LAND APPLICATION OF WASTEWATER

One of the major issues raised in the consideration of land treatment is the health risk involved. Concern over the danger of disease at sewage farms and the possible contamination of irrigated crops has existed from the start. This public fear, however, has had little evidence to support it. When a major typhoid epidemic struck Berlin in 1889, not one case was reported on the sewage farm (Mitchell, 1931). Boleslawiec, site of the first land treatment system, was considered one of the healthiest European cities in the 19th century (Wierzbicki, 1949). Kutepov (1968) stated:

> ... no direct evidence that the irrigation of fields with sewage increases the danger of epidemics ... that where this was thought to be the case, careful investigation invariably dispelled the suspicion.

The use of wastewater for the irrigation of crops and the possibility of contamination has aroused great public concern. This concern led to restrictions concerning the types of crops used for wastewater irrigation.

California was one of the first states to adopt legislation concerning the use of sewage for irrigation (Wolman, 1977). This bill, which was passed in 1918, prohibited the application of untreated wastes to garden produce which was consumed raw, but permitted the irrigation of vegetables which were normally cooked, as long as the irrigation ceased thirty days prior to harvest (Ongerth and Jopling, 1977). Since then, new regulations have been passed, the most recent ones in 1975. This legislation, which is very comprehensive, not only covers the conditions under which wastewater reclamation is permitted with regard to pretreatment, application mode, crop, and bacteriological standards, but also regulates the design and operational procedures of such systems.

Germany also has regulations concerning the use of domestic sewage for irrigation and a restriction prohibiting its use for the irrigation of vegetables (Müller, 1977). The regulations, especially with regard to pretreatment, are much less specific than the California legislation. For example, in Germany the only pretreatment required is settling and some degree of biological treatment is termed "desirable." The only restriction placed on wastewater utilized for spray irrigation is that it is to be chlorinated. The California regulations, on the other hand, specifically state pretreatment steps for both flood and spray irrigation for varying crop types.

Kornder (1976) asks:

> Is our concern justified? I think the first assumption has to be that there really are health concerns associated with spray irrigation of sewage. The degree of risk is, of course, most difficult to quantify. However, the risk the public seems prepared to accept is quite small, in spite of the fact that many do not use seat belts, many smoke and drink to excess, and most are obese. Those personal choices of health abuse somehow seem to be viewed with much less alarm than the infinitely smaller risk associated with spray irrigation of sewage.

It seems rather ironic that so much concern is focused on spray irrigation while at the same time relatively little is said about the indiscriminate use of surface waters which are often polluted. Many land treatment systems utilize wastewater which has undergone secondary treatment.

There are several ways in which land treatment systems may spread disease. They can be a direct source, as in the cases of crop contamination, bodily contact, or airborne dispersal of pathogens, or they can be an indirect source, as in the case of underground water contamination. Although the possibility of any of these occurrences is very real, with proper design and management such risks may be rendered minimal. In evaluating the degree of risk associated with land treatment, consideration must be given to the health risk involved with alternative treatments.

The transmittance of pathogenic organisms to humans by means of sewage-irrigated crops has been documented (Kornder, 1976). This danger, however, can apply equally as well to conventionally treated wastes. This is especially true in arid regions where sewage plant effluent composes a major portion of stream flow which is often used for irrigation.

The biological purification obtained from conventional treatment is not complete. There is evidence that greater bacterial and viral removal results from land treatment (Kutepov, 1968; Ewert et al., 1973; Tietjen, 1977; Center for the Study of Federalism, 1972; Bouwer, 1976). Concerning the subsurface drainage of the land treatment system at Braunschweig, Germany, the local public medical board has stated (cited by Ewert et al., 1973):

> The discharge of the Okertrench at Neubruck... is in its quality very definitely better than discharge of unobjectionably operating mechanical-biological treatment plants.

Although conventional treatment does remove most pathogenic organisms, it does not remove viruses. Soil, on the other hand, has been found to remove viruses (under proper management) as well as bacteria (Tietjen, 1977; Bouwer, 1976; Center for the Study of Federalism, 1972). Viral particles are removed by adsorption onto negatively charged clay particles or organic matter, or by fixation by surface active materials, such as iron and aluminum oxides. There are many factors, such as pH, cation concentration, and the soil cation exchange capacity, which greatly affect virus removal (Bouwer, 1976).

The Santee, California, system was the site of a three year study of virus removal. Five monitored viruses, Reovirus, Adenovirus, Polio, Echo, and Coxsackie, were found to be present in both primary and secondary treated waste, but absent after land treatment. To determine the efficiency of removal, an experiment was performed where concentrations of Type 3 Polio virus were applied directly to the soil. Complete virus removal was achieved in the first 60 meters (Center for the Study of Federalism, 1972).

Bodily contact with wastewater poses hazards which can be easily avoided. When wastewater is applied to recreational areas such as parks and golf courses, pretreatment, which includes disinfection, renders the wastewater just as safe as the streams and lakes into which sewage effluent is discharged. Fishing and other recreational use is made of many such streams. Land treatment also provides additional purification of the water. For example, effluent from the Santee system is used to create a series of lakes, the last of which was found safe enough for public swimming, and was so used for several years (Ongerth and Jopling, 1977).

Spray irrigation of wastewater does pose a risk because both bacteria and viruses can travel in the air (Bouwer, 1976). To decrease the possibility of spreading disease by spray drift or droplet infection, buffer zones can be established or rows of hedges or trees erected around the fields. Other options include flooding the wastewater onto the fields, or disinfecting the wastewater before spraying.

At locations where there is a potential contamination of groundwater from a land treatment system, there are two possible options. In the case where the contamination is the result of just one or two substances, it is possible to pretreat the waste specifically for the removal of those substances.

In summary, the potential health hazards from land treatment are no greater and probably less than those from conventional treatment. Land treatment, under proper management, is more effective, reliable and predictive in the removal of toxic trace metals and organics, viruses, and pathogens than conventional sewage treatment. The degree of acceptable risk has to be defined and compared with the risk associated with other alternatives of wastewater treatment and disposal.

PRESENT INTEREST IN LAND TREATMENT

Recent interest in land treatment of wastewater has been a result of the increased pollution of natural water as a consequence of the discharge of untreated or partially treated wastewater into open water. Eutrophication and a high level of toxic chemicals in the food chain were the immediate signs of water quality deterioration. As a result, governments have set standards for the discharged effluents which can not easily be met on a cost-effective basis by mechanical treatment methods. In the meantime, they passed laws and guidelines encouraging local governments and companies to consider treating wastewater by application on land. The most important ones are PL 92-500 and PL 95-217.

Recently the U.S. Environmental Protection Agency (EPA) emphasized two basic criteria: namely, the cost effectiveness and the best practicable treatment (BPT). These require any grant applicant to evaluate three alternatives for each project: 1) treatment and discharge, 2) land treatment, and 3) reuse, such as for industrial waters, the basic requirement being secondary treatment. For land treatment, the primary goal is treating the wastewater while simultaneously protecting the groundwater. In this regard, three separate criteria have been established, based on the ultimate use of the renovated water. If the groundwater is to be used for drinking, the aquifer must continue to meet the chemical, biological, and pesticide criteria established in the drinking water standards. If the groundwater has a potential to be used as a source of potable water, even though it may not be used for that purpose at the present time, the same chemical and pesticide standards must be met, though the biological standards do not apply. If the renovated water would clearly never be used as potable water, then regional EPA and state regulations would apply.

Although land treatment, like all known waste treatment systems, has potential health hazards associated with it, these risks can be kept to a minimum. Considering that land treatment has been found to remove nitrogen, heavy metals, phosphorus, bacteria, and viruses better than conventional treatment (Bouwer, 1976; Ewert et al., 1973; Tietjen, 1977; Kutepov, 1968; and Center for the Study of Federalism, 1972), it is rather odd that the former instills so much more public fear than the latter.

LEARNING FROM THE PAST

There are many things that we can learn from past experience in land disposal of wastewater. Information on purpose, feasibility, design, performance, management, monitoring, and public acceptance may all be extrapolated by proper evaluation of existing systems. Several review articles (Shuval, 1977) have recently been published on the reuse of wastewater. The feasibility of land treatment should be evaluated on a cost effective basis.

Experience gained in design, performance, management and monitoring of land treatment systems may be utilized during the planning and design stages of new systems. Because of the large number of systems available, a computer file of both foreign and domestic land application systems has been established (Iskandar, unpublished) at the U.S. Army Cold Regions Research and Engineering Laboratory in Hanover, N.H.

Information on 11 design parameters as well as available papers is included. Parameters include system name, location, system type, flow rate, waste type, purpose of the system, and year of initiation. When information on a certain type of system is desired, the program can retrieve all stored information which fits the given parameters as well as a listing of all related papers. It is hoped that by the use of such information, past experience can be used and successful techniques employed so that further improvements in the land treatment of wastewater may be achieved.

Long-term heavy metals and phosphorus accumulation in land treatment may be estimated by examination of soils and vegetation obtained from old disposal systems and compared with the levels in adjacent similar soils which have never received wastewater. This technique has been utilized successfully by Iskandar et al. (1977), Baillod et al. (1977) and Uiga et al. (1977), and may also be utilized in working demonstration projects. An operating example is the best response to objections raised by the public concerning odor, unsightliness, depreciation of land value, or treatment efficiency. In California, the state containing the largest number of land application systems (Bouwer, 1976), a public survey was made of the attitudes toward different uses of reclaimed water. It was found that opposition was greatest for uses involving the greatest amount of human contact and that the reason for this opposition was mainly psychological repugnance (Ongerth and Jopling, 1977).

In summary, there is much to gain from visiting and evaluating existing land treatment systems for wastewater. Not only can past performance characteristics be determined, but also information about the design parameters, cost estimates, health aspects, and public acceptance may be gained for present and future construction purposes.

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EVALUATION OF EXISTING SYSTEMS

RENOVATION OF WASTE WATER BY LAND TREATMENT AT MELBOURNE BOARD OF WORKS FARM WERRIBEE, VICTORIA, AUSTRALIA.

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Wastewater from the City of Melbourne has been renovated by land treatment method at the Board of Works Farm, Werribee since 1896. On arrival the raw sewage is distributed through a network of concrete lined channels. Of this sewage, 30% of flow and 75% of the organic loading (B.O.D.5) is contributed by tradewaste discharges, the balance comes from domestic sources. Average flow is 440 Ml/day, B.O.D.5 540 mg/l and S.S. 588 mg/l. Three methods of purification are used depending on the season and the rate of flow - Land Filtration, Grass Filtration and Lagooning.

The soils and climate conditions at Werribee favour land treatment methods for wastewater renovation. The performance of both land systems shows removal rates of 95% B.O.D.5 and S.S., whilst Total Nitrogen and Total Phosphorus removal under grass filtration, the winter system, is much less effective than land filtration.

During the year approximately 23% of flow is land filtered, 34% grass filtered with 43% going to lagooning. The hydraulic loadings imposed are 1.10 Ml/ha/day for land filtration and 0.18 Ml/ha/day for grass filtration.

Analysis of herbage shows considerable increases in N.P.K., with very little increase in Ca and Mg levels. Heavy metals are within the acceptable range for most common grasses and legumes. Analysis of kidney and liver tissues have shown concentrations of Zn, Cd etc. to be within the expected range for mammalian tissue.

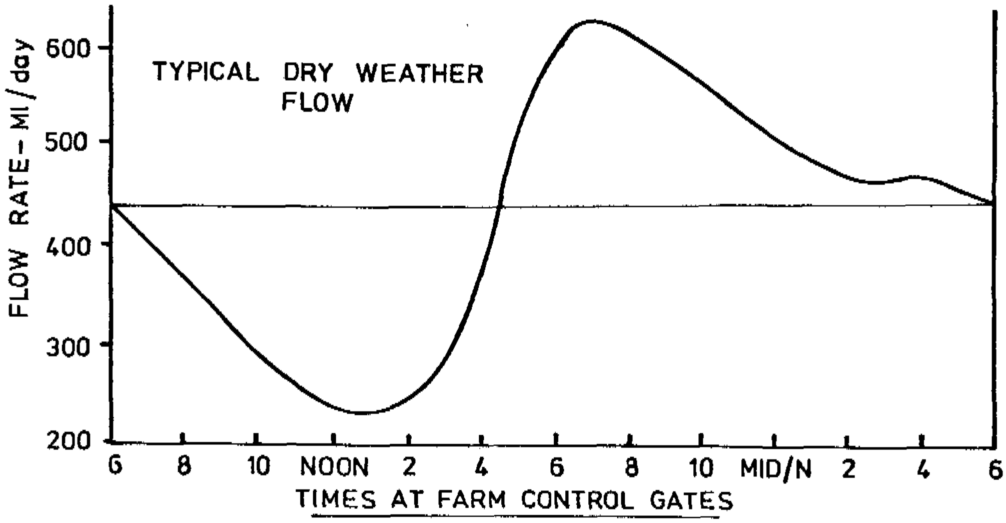
The Farm carries 22,000 head of beef cattle and 30,000 sheep. Revenue from the sale of livestock offsets some of the net cost of sewage purification.

Investigations are now in progress to assess the future potential for the re-use of this reclaimed wastewater.

INTRODUCTION

Over the past 80 years, sewage from the City of Melbourne (population 2,689,000) has been purified by land treatment at the Melbourne and Metropolitan Board of Works Werribee Farm located 35 kilometres south west of the city and occupying an area of 10,850 hectares. This area is bounded to the south by a frontage of 21 kilometres to Port Phillip Bay, and to the north by a major highway. The Werribee River adjoins the Eastern Boundary for 10 kilometres, whilst to the west the boundary abuts general farmland and a major airfield.

The waste water arrives at the Farm as raw sewage and is distributed by a network of channels. Of the flow 30% is trade waste, the balance coming from domestic sources. Figure 1 shows typical dry weather flows. The average flow is now approximately 440 Ml/day, five day B.O.D. 540 mg/l and suspended solids 588 mg/l.



Variation Of Flow Rate Figure 1.

PURIFICATION PROCESSES

Three main methods of purification are used depending on the season and the rate of flow: (Figure 2).

Land Filtration

Land filtration, for the periods of high evaporation during late spring, summer and autumn (mid September - April), which is the normal irrigation season in Victoria.

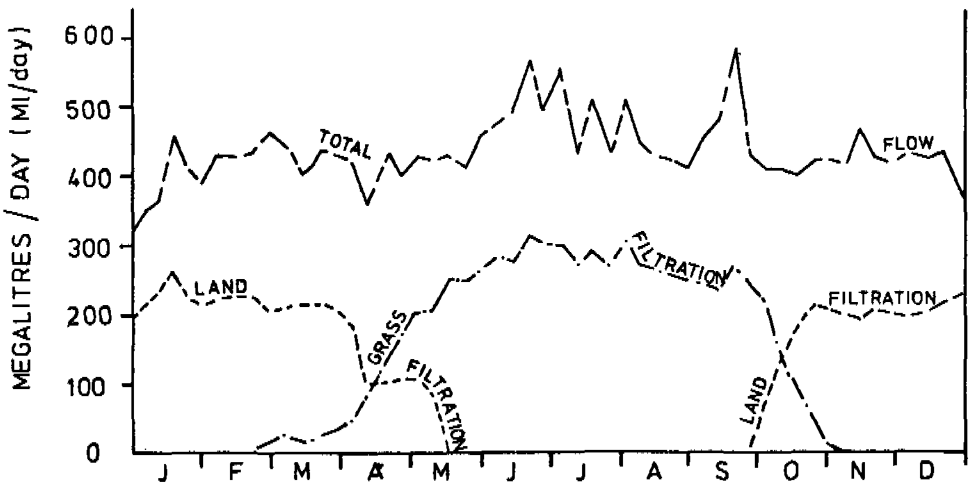
Grass Filtration

Grass filtration, also known as overland flow, for periods of low evaporation (May - August), when irrigation is not practical.

Lagooning

Lagooning (anaerobic and aerobic) for peak daily and wet weather flows.

In general, the more permeable soils adjacent to the Werribee River are used for land filtration, the heavier clays and loams on the western side of the Farm for grass filtration and the lower foreshore areas for lagoons.



Average Daily Flow Figure 2.

PRIMARY SEDIMENTATION AND SLUDGE PROCESSING

All of the sewage applied to the grass filtration areas, and part of the flows to the land filtration and lagoon areas, is first passed to sedimentation tanks to remove settleable solid matter.

Sedimentation is performed in shallow concrete lined basins having a total area of 3.3 ha. The solids deposited, usually referred to as sludge, are transferred periodically to sludge digestion lagoons.

The sludge is retained here for an extended period, during which it ferments to a "digested", or stabilized condition.

The sludge lagoons average about 4 m in depth and have an overall area of approximately 13 ha each.

Digested sludge is discharged by gravity flow from these lagoons to drying areas. It remains drying over a period of at least 2 years after which it is cleared by earth moving equipment, and used as earth fill around the property.

LAND FILTRATION

Purification by land filtration involves irrigation of permanent pasture bays, with mainly raw sewage, during the periods of high evaporation between October and April.

Each bay is given approximately 1 to 2 days irrigation, followed by 5 to 8 days drying and 10 to 14 days livestock grazing, making a total operating cycle of 18 to 21 days.

Depending on the terrain, the bays are prepared, either as flat plots surrounded by banks, or as graded areas 180 m long by 10 m wide, which receive sewage at their higher end. The first system described is referred to as square check irrigation, whilst the latter is the well known border check irrigation system.

In both cases, banks prevent overflow of the applied sewage directly into the collecting drainage system, except where series treatment in adjacent bays is adopted.

The irrigation of each bay is adjusted according to the prevailing circumstances, the average application being some 100 mm.

The irrigant is purified by filtration through the soil, most of the nutrient and heavy metal compounds being removed in the process.*

Some 50 to 60 percent of the applied irrigant percolates in this manner and is collected in a system of open drains 1.2 to 1.8 m deep, which are constructed throughout the area at 180 m intervals.

The remainder is lost in evaporation, transpiration or seepage into the lower subsoil.

Wastewater irrigation promotes a high rate of pasture growth and this plant material must be regularly removed to maintain areas in proper condition, both from the point of view of continuing growth and irrigation. Removal is carried out by grazing animals.

GRASS FILTRATION

These areas are used during the months from May to September, when evaporation is low and land filtration is impracticable.

Use of this process began in about 1930, to provide purification of winter flows which had previously been held in shallow lagoons along the foreshore to Port Phillip Bay.

In the grass filtration process, sedimented sewage is continuously passed over graded areas planted with Italian rye grass. As the sewage seeps through the vegetation, the suspended matter is filtered out and the organic matter is removed by a biologically-active film built up on the vegetation. The purified effluent is then discharged to the drainage system.

The areas are prepared in a similar manner to those for the graded land filtration. The grass filtration areas then dry out, the grass goes to seed and cattle are then admitted to feed on the dry vegetation. The grass seed remaining on the ground is germinated by rain or irrigation in the next autumn to provide for the following operational season.

LAGOON TREATMENT

Lagoons, although not a form of land treatment, were introduced in 1937, to provide purification throughout the year for daily peak flows and wet weather flows which exceed the capacity of the land or the grass filtration processes.

In the lagoon process, the sewage is passed through a series of 8 to 12 lagoons, each having an area of between 4 to 8 ha. The average depth of the lagoons is just over a metre. The first lagoon in each series being deeper than the remainder to accommodate sludge accumulation.

Sewage is purified by the natural processes of sedimentation, aeration and by bacterial and algal activity in the lagoons. Because of the high organic loading which is imposed, the first few lagoons of each series are anaerobic. The organic loading progressively decreases as the sewage passes through the succeeding lagoons, which remain aerobic. Development of algal growth in the lagoons assists the purification process by the photo-synthetic production of oxygen.

PRODUCTION FROM WERRIBEE FARM

Irrigated pastures are grazed by up to 22,000 head of cattle throughout the year and during spring and summer 20,000 to 50,000 sheep are fattened.

Annually, 7,000 cattle are sold and replaced by natural increase with calves born during the year.

Sheep are purchased during spring and summer and sold in autumn.

Livestock thrive on the Farm and require only the same care and attention as livestock elsewhere. Sales of Farm cattle are subject to the provisions of the State Health Act 1958 and the Abattoirs and Meat Inspection Act 1973, which requires all cattle on leaving the Farm, to be immediately slaughtered at a registered abattoir controlled by the Victorian Department of Agriculture, whose staff ensure all carcasses are subjected to rigid inspection.

These restrictions are imposed under the Victorian Health Act Regulations to detect whether cattle which have grazed on land irrigated with sewage have developed the disease commonly known as "beef measles". This is caused by the ingestion of grass containing eggs of the tape worm Taenia saginata which may develop into cysts (Cysticercus bovis) in the body of the animal. Viable cysts in the beef are destroyed by cooking or freezing.

Under such rigid meat inspection, condemnations of carcasses of Farm animals from all causes run at about 0.02% - approximately the same as for animals over the rest of the State.

Sheep are not subject to restrictions concerning disposal as in the case of cattle.

HEALTH AND AMENITY

During the whole of its existence, the Farm has had a resident population varying from 500 at times to 40 at present. The work force has exceeded 500 on occasions and is now 325. The health of these people has been as good as that of the community generally and no epidemics of disease have occurred.

No special precautions have been taken other than normal hygiene practices, and at one time, it was the custom for residents to water their gardens with raw wastewater.

There have been complaints of odours, but it is only infrequently that these extend beyond the Farm boundaries, and it is felt that the degree of offence or inconvenience has been minimal.

The area of Port Phillip Bay adjacent to the Farm is a popular fishing ground, and many small boats operate there, obtaining good catches.

ECONOMICS OF FARM OPERATION

The costs incurred in operating and maintaining the Farm can be divided into the following categories:

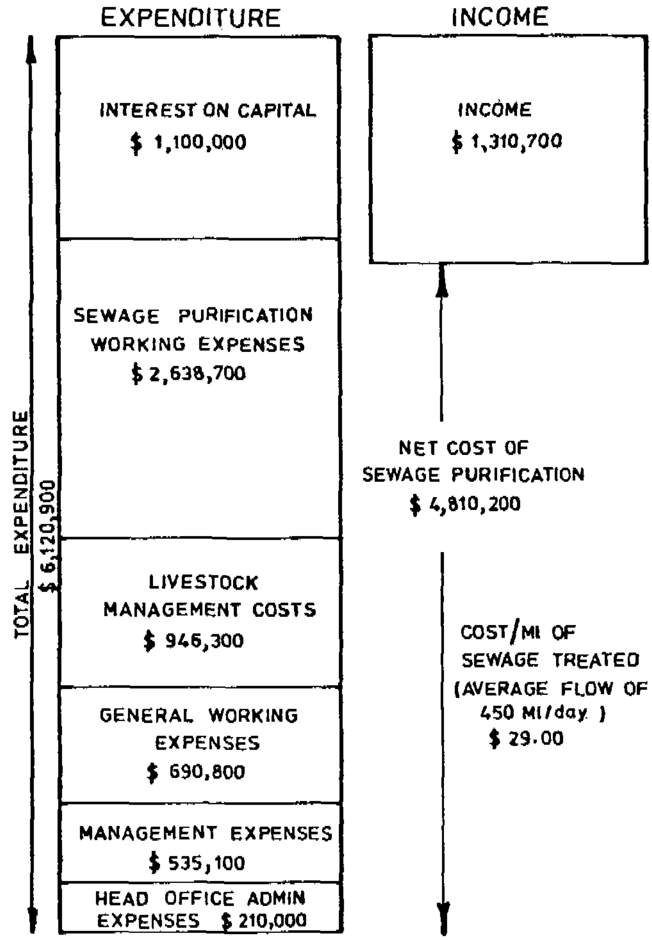
- sewage purification working expenses;
- general working expenses;
- management expenses;
- livestock management costs;
- interest charges; and
- head office administrative charges.

These costs are partly offset by marketing the Farm's livestock.

The annual gross return from the sale of livestock is in the order of $1.3 million.

After crediting revenue, the net cost of Sewage Purification is some $4.8 million. This represents treatment costs of about 2.9 cents per 1,000 l.

Fig. 3 is a diagrammatic representation of the Farm's budgeted income and expenditure for the year 1977/78. Income from livestock fluctuates seasonally depending on weather and market conditions but always exceeds that expended on livestock management.



Farm Estimated Expenditure And Income For 1977/78 Figure 3.

CLIMATE

The average annual rainfall (83 year av.) is 488 mm, whilst evaporation (4 year av.) is 1,447 mm. The rainfall is distributed evenly throughout the year with just a slight winter evidence.

Temperatures are never low enough to completely stop plant growth, although crops such as lucerne (alfalfa) make very little winter growth. Frosts may occur from April to October, but owing to the proximity to Port Phillip Bay, they are not severe.

SOILS

Two main soil types are encountered, a delta soil - red brown loam (15 - 30 cm) overlying a medium clay. This soil has a moderate permeability and a pH 6.9. The second soil is of basaltic origin, consisting of a red brown light clay (0 - 10 cm) or variable coloured hard setting loams (0 - 15 cm) over medium to heavy clay. This soil has low permeability and a pH 6.9.

RESULTS OF TREATMENT PERFORMANCE

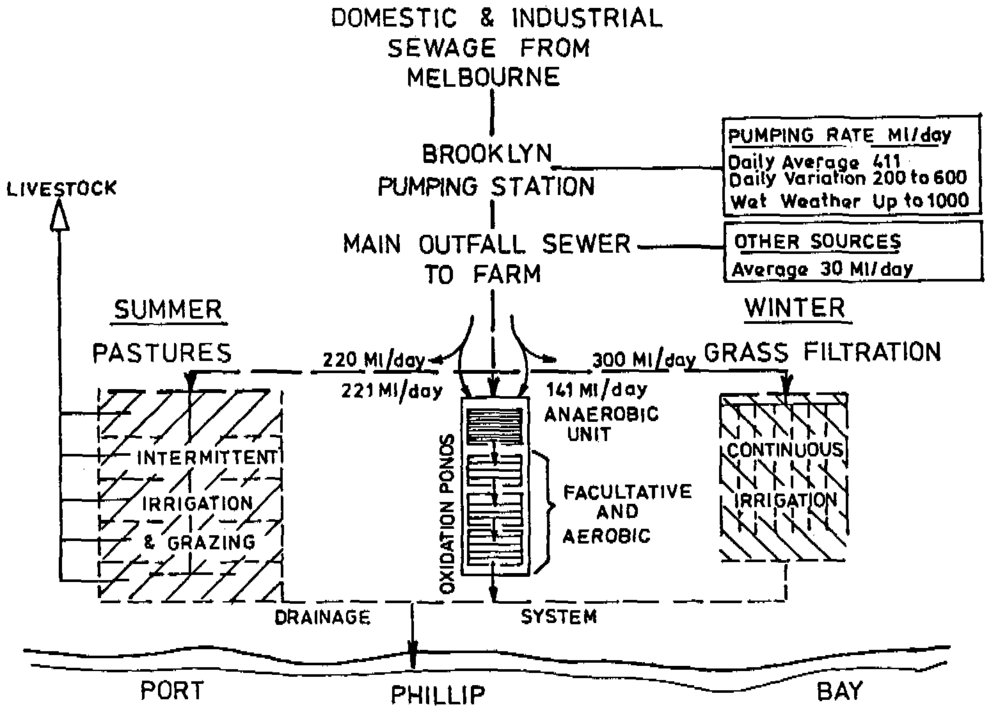
Samples of effluent from all drains discharging from the various treatment processes are taken twice weekly, and submitted for laboratory examination.

Results of the various land treatment processes for 1977 are summarised in the following Table 1.

PARAMETER	INFLOW RAW SEWAGE	LAND FILTRATION		SEDIMENTATION		GRASS FILTRATION PLUS SEDIMENTATION	
	mg/l	OUTFLOW EFFLUENT mg/l	PERCENT REMOVAL %	OUTFLOW EFFLUENT mg/l	PERCENT REMOVAL %	OUTFLOW EFFLUENT mg/l	PERCENT REMOVAL %
5 DAY B.O.D UNFILTERED	540	6	95	396	25	21	95
SUSPENDED SOLIDS	588	15	95	172	70	28	95
VOLATILE SUSPENDED SOLIDS	476	10	95	140	70	13	95
TOTAL DISSOLVED SOLIDS	1100	2200	—	—	—	1075	—
TOTAL ORGANIC CARBON	336	22	95	241	35	37	90
COLOUR Pt/Co UNITS	270	50	—	270	—	300	—
pH UNITS	6·9	7·2	—	6·9	—	7·7	—
DISSOLVED OXYGEN	—	*6·0	—	—	—	*2·9	—
NITRITE AS N	0·06	0·24	—	0·06	—	0·07	—
NITRATE AS N	0·12	2·02	—	0·04	—	0·34	—
AMMONIA AS N	26·1	1·6	—	27·2	—	25·9	—
ORGANIC NITROGEN	25·0	2·7	—	16·5	—	4·8	—
TOTAL NITROGEN	51·3	6·6	85	43·8	15	31·1	40
ORTHOPHOSPHATE AS P	5·1	0·9	—	6·9	—	6·5	—
TOTAL PHOSPHORUS	7·8	1·1	85	7·3	5	6·5	15
ANIONIC SURFACTANTS	3·5	0	100	3·2	10	0·7	80
COPPER	0·32	0·02	95	0·18	45	0·03	90
CHROMIUM	0·29	0·03	90	0·20	30	0·06	80
CADMIUM	0·010	0·002	80	0·007	10	0·002	80
IRON	3·8	—	—	2·4	40	1·9	50
LEAD	0·28	0·01	95	0·19	25	0·02	95
MERCURY	0·0030	0·0004	85	—	—	0·0005	85
NICKEL	0·11	0·04	65	0·08	10	0·06	45
ZINC	0·79	0·04	95	0·51	35	0·08	90

Table 1.

Note: Removal rate is on a concentration basis. Percent removal is to the nearest 5%.



Sewage Purification 1977 Figure 4.

HYDRAULIC LOADING

Figure 4 sets forth the sewage distribution over the past 12 months period.

To effect these performance figures, the inflow of raw sewage was 441,000 m^3/day (i.e. 441 Ml or 116 MG(U.S.)/day).

The land filtration process is used to purify up to an average of 214 Ml/day of raw sewage over an approximate peak operational season which is equivalent to a hydraulic loading rate of 1.10 Ml/day/ha (Table 2) as only 5% of the total area available is under irrigation at any one time.

The flow to the grass filtration area is continuous and relatively constant with only small diurnal fluctuations. The water losses by infiltration to ground water and evapotranspiration are relatively insignificant and largely balanced by precipitation. The process is used to treat an average flow of up to 271 Ml/day of settled sewage at a hydraulic loading rate of 0.18 Ml/day/ha.

Sewage Treatment Facilities

		LAND FILTRATION	GRASS FILTRATION	LAGOONS	SEDIMENTATION AREAS
AREA	$10^6 m^2$	42·81	14·63	14·99	0·60
	ha.	4 281	1 463	1 499	60
	ac.	10 578	3 615	3 704	148
PERCENT FLOW TREATED		23	34	43	
PEAK OPERATING AVERAGE DAILY FLOW					
	$10^3 m^3$	214	271	191	
	Ml.	214	271	191	
	M.G. (IMP.)	47	59	42	
	M.G. (U.S.)	56	72	50	
PEAK OPERATING HYDRAULIC LOADING					
	Ml/day/ha.	1·10	0·18	0·14	
	M.G.(IMP)/day/ac.	0·097	0·016	0·013	
	M.G.(U.S.)/day/ac.	0·117	0·019	0·016	

Table 2.

NUTRIENT REMOVAL

Nitrogen

Nitrogen is removed from the sewage by land processes in the following ways:

- by incorporation into the pasture;
- by volatilization of ammonia from the irrigant, from the soil and from the faecal matter of the grazing animals;
- by diffusion through the soil to the atmosphere of molecular nitrogen resulting from denitrification of nitrate; and
- by incorporation into microbial biomass in the soil.

In the land filtration process, with the present method of irrigation and at the current hydraulic loading rate, the first two removal mechanisms are most important. Ammonia can be temporarily bound in the soil under the anaerobic conditions occurring immediately after each raw sewage irrigation and can be converted to nitrate during the drying period when aerobic conditions develop. Nitrate ions cannot be bound in the soil matrix and will either be utilized by the vegetation or leached out to the underground water table which is intercepted by the effluent drains. Presently the nitrogen removal rate, for the six month operational season is some 85%.

In the grass filtration process the main mechanism for the removal of nitrogen is by incorporation into the pasture. Presently the nitrogen removal rate, for the six month operational season, is some 40%.

In both these processes real nitrogen removal with the pasture is achieved by grazing with cattle and sheep, even though some recycling of nutrients occurs via animal faecal matter.

Phosphorus

Phosphorus is removed from sewage by the land processes in the following ways:

- by incorporation in the pasture; and
- by chemical precipitation and adsorption to the soil matrix.

In the land filtration process the latter mechanism is the principal means of phosphorus removal. Phosphorus precipitated in the soil is far in excess of the normal requirements for pasture. Total phosphorus removal averages some 85% for these areas at the present rates of loading.

In the grass filtration process the former mechanism is the primary means of phosphorus removal. As there is little leachate flow through the soil matrix to the underground water table, a significant removal mechanism for phosphorus is not available. Total phosphorus removal averages some 15%.

Nutrient Accumulation in the Soil

There has been significant accumulation of nitrogen, phosphorus and carbon in the soil in the land filtration areas, as a result of long-term irrigation with unsedimented sewage. Detailed information is provided in the report prepared for the U.S. Department of the Army, Corps of Engineers on "Selected Chemical Characteristics of Soils, Forages and Drainage Water from the Sewage Farm Serving Melbourne, Australia". It can be seen from Table 3, P. 17 of this report that the accumulation of nitrogen is confined to the top layers of soil up to 23 cm below the surface whereas the building up of phosphorus has extended to depths of 46 cm or more.

It is considered that the soil in the land filtration areas will not become "clogged" with phosphorus for many years and that it will still provide a significant phosphorus removal medium for at least the next 50 years.

In the grass filtration areas nutrients are only accumulated in a relatively thin surface layer of some 75 mm.

TRACE METALS

Removal

Agents for trace metal removal throughout the Farm treatment processes include sedimentation and adsorption to organic matter (sludge) and soil clay particles (with subsequent uptake by vegetation and grazing animals); precipitation as sulphides; together with bioflocculation and accumulation in treatment biota (slimes algae and zooplankton).

In general high levels of removal are obtained by the three processes (83% ± 12%).

Relationship Between Soil, Plant and Animal

Current investigation of land filtration areas indicates a general linear relationship between accumulation in the soil (some 60% in the top 10 cm) and total volume of sewage applied over the years. Retention in the soil is therefore non uniform even over quite small areas, due to variation in terrain, grade, soil type and application techniques.

Vegetation likewise (within particular soil types) correlates with input loads and bears a general direct relationship in trade metal levels to the soil on which it grows.

Current investigation into the liver and kidney levels of Farm cattle grazed generally on sewage irrigated pastures indicate no increase with age. The levels, whilst generally higher than those of Farm cattle grazed solely on non irrigated pastures, were marginally lower than a random sample (26) from non Farm cattle.

The picture is further clouded by metal interaction, where such significant metal level increases as for example zinc, cadmium and particularly copper in herbage consumed, are not reflected in the grazing animal tissue.

CURRENT INVESTIGATIONS FOR FUTURE DEVELOPMENT

A number of investigations are currently underway to determine the practicability and economics of using treated effluent for irrigation of high value agricultural crops.

The areas of immediate interest are:

Forestry

Varied species of trees have been planted to evaluate those best suited for:

- Commercial production of structural timber, wood chip, pulp and matchwood.
- Ornamental varieties for esthetic purposes, shade and shelter.
- Attention is being given to those most suited for the establishment of dense growth wood lots.

Agriculture

A continuing program is underway to assess the potential of various grain and forage crops.

- Winter and spring sown cereal crops.
- Oilseed crops.
- Summer grain and forage crops.
- Vegetable rotations and production.
- Varied pasture mixtures for protein extraction.

As would be expected in the forestry section, several eucalypt species have made dynamic growth, whilst the growing of summer grain crops requires close attention to variety and crop type. Sunflowers have given excellent yields, as has winter sown barley for grain production. With most summer grown crops, seed maturity and harvesting have presented many difficulties.

Vegetable production appears to provide a high degree of success, but studies are continuing to observe growth rates, product quality, health status and heavy metal accumulation.

Animal Health Investigations

The problem of "beef measles" in cattle has been investigated with the production of a vaccine and the application of chemotherapy.

Vaccination of calves against Taenia saginata infection using antigens collected during in vitro cultivation of larval cestodes has created world interest. This work commenced in 1974 and has given complete protection to calves subjected to artificial challenge of tapeworm eggs. This vaccine is now undergoing field test evaluation.

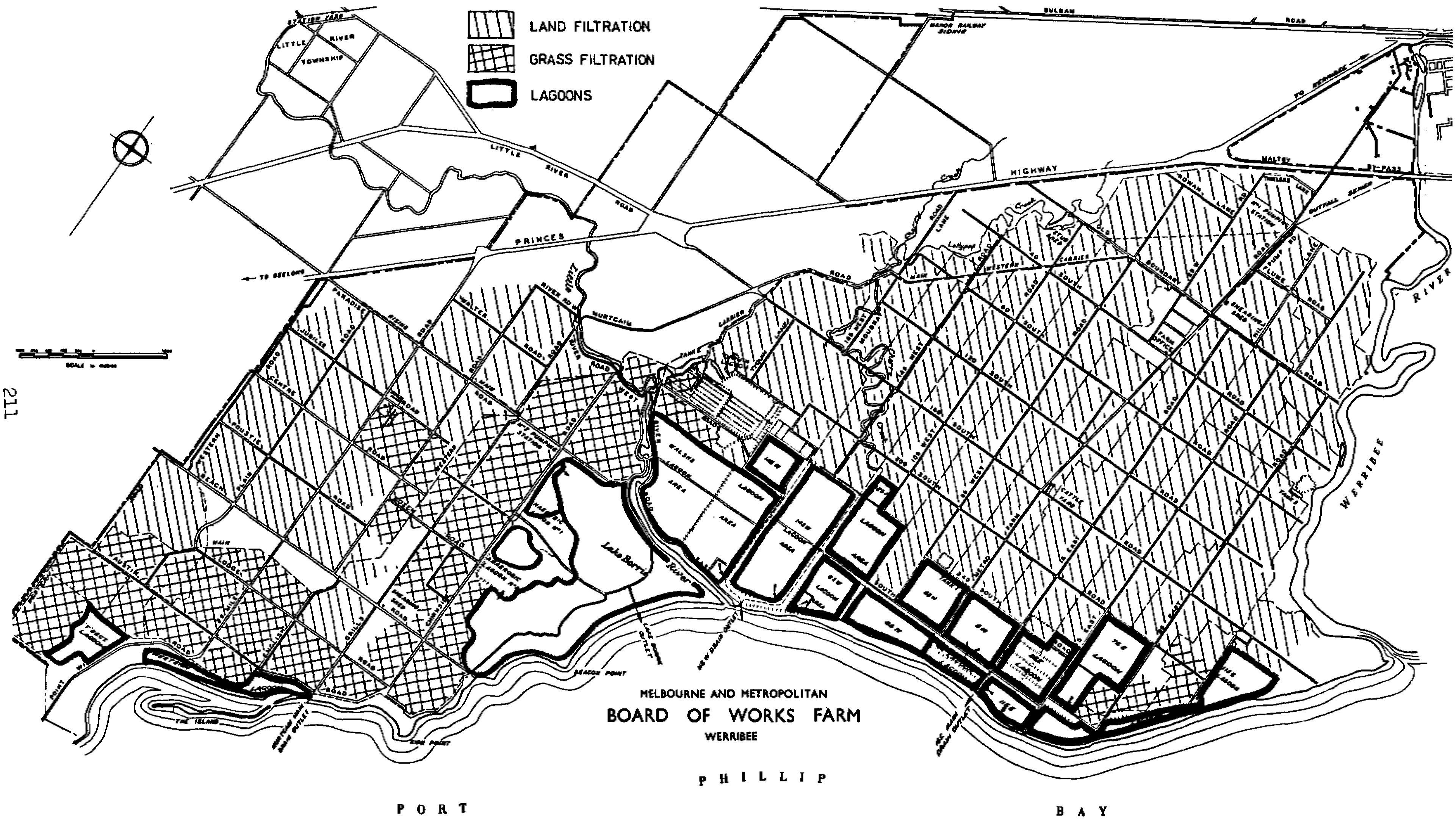
Another important activity under investigation is the use of Taenia hydatigina derived vaccine rather than Taenia saginata. Earlier experiments showed a significant degree of cross protection with this parasite and work is in progress to record whether the degree of cross protection given is sufficient to hold field acquired infections to a low level.

Facts About The Board Of Works Farm Werribee
(As at 30th June, 1977)

Area	10,849 hectares (26,809 acres)
Number of Employees	325
Road Constructed	231 km
Fencing Erected	Approx. 2,024 km
Channels Constructed	853 km
Drains Constructed	666 km
Annual Rainfall at Farm Office	488 mm
Sewage Purified and Disposed of during Year Ended 30/6/77	163,026 Ml
Median Daily Flow	438 Ml
Average Daily Flow	441 Ml
Area used for Purification of Sewage -	
Land Filtration	4,281 hectares
Grass Filtration	1,463 hectares
Lagoon Treatment	1,499 hectares
Sedimentation Areas	60 hectares

LIVESTOCK

Cattle on Farm as at 30/6/77	15,651
Cattle Bred on Farm during Year	6,665
Cattle sold during Year	6,951
Sheep on Farm as at 30/6/77	6,208
Sheep sold during Year	19,591
Number of Sheep Shorn	4,333


LAND FILTRATION
GRASS FILTRATION
LAGOONS
LITTLE RIVER TOWNSHIP
LITTLE RIVER ROAD
PRINCES
HIGHWAY
TO GEELONG
Lake Borrie
WERRIBEE
RIVER
MELBOURNE AND METROPOLITAN
BOARD OF WORKS FARM
WERRIBEE
PORT
PHILLIP
BAY

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EVALUATION OF EXISTING SYSTEMS

THE FLUSHING MEADOWS PROJECT

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After 10 years of operation of the Flushing Meadows Project west of Phoenix, Arizona, in which 760 m of secondary effluent infiltrated into the soil, there has been no reduction in renovation efficiency and hydraulic capacity of the system. Cycles of 2-3 weeks flooding and 10-20 days drying produced maximum hydraulic loading rates of about 90 to 120 m per year, but relatively low nitrogen removal (< 30%). Nine-day-flooding and 12-day-drying periods, on the other hand, yielded loading rates of 60 to 75 m/year, but about 60% of the nitrogen in the effluent was removed in the soil by denitrification. Under these loading conditions, the renovated water in 1977 contained 8.6 mg/l NO_3-N, 1.7 mg/l NH_4-N, and 0.55 mg/l organic N. Phosphate removal was 50 to 80% in the first 10 m of downward movement below the basins, but increased to more than 90% after 60 m lateral movement of the renovated water through the aquifer, where phosphate apparently continued to precipitate. The renovated water also contained about 1 mg/l fluoride (2 mg/l in the effluent) and 0.5 mg/l boron (boron is not removed by the soil). Metal concentrations in the effluent were already below maximum limits for irrigation. Viruses could not be detected in renovated water sampled at 6 to 9 m depth below the basins. Fecal coliforms were absent after the renovated water had moved laterally through the aquifer for about 60 m. Total organic carbon of the renovated water averaged about 4 mg/l. Possible toxicity of trace organics is the major concern in potable reuse of renovated wastewater. Operational systems should be designed and managed so that only a portion of the aquifer is used as a natural, advanced treatment system, protecting the rest of the aquifer and high quality indigenous groundwater against encroachment by renovated wastewater.

PURPOSE AND DESCRIPTION OF SYSTEM

The Flushing Meadows Project was installed in the Salt River bed west of Phoenix, Arizona, in 1967 to study the feasibility of renovating conventional secondary sewage effluent by groundwater recharge with rapid-infiltration basins. Renovation of the effluent is desirable because it enables use of the water for unrestricted irrigation, recreation, and other purposes with a relatively high economic or social return. The sewage flow from Phoenix and surrounding cities presently is about 0.45 X 10^6 m^3/day (120 mgd) and increases by about 0.02 X 10^6 m^3/day (5 to 6 mgd) each year due to the increasing population. Since the Salt River Valley is a water-short area where groundwater is being depleted, reuse of the effluent would reduce groundwater overdraft and slow down water table declines, which have been as much as 3 m (10 ft) per year in some areas.

The Flushing Meadows Project is located on the north side of the Salt River bed about 2.5 km (1 1/2 miles) downstream from the 91st Avenue Sewage Treatment Plant in Phoenix. This is an activated sludge plant which presently

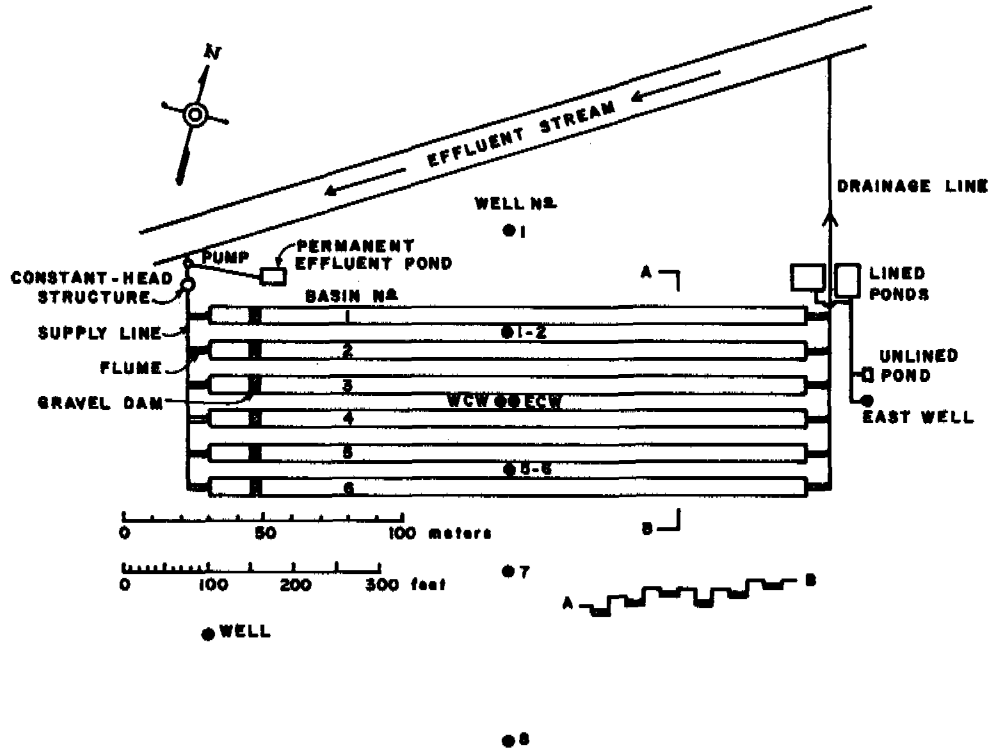

Figure 1. Schematic of Flushing Meadows Project.

discharges about 0.28 X 10^6 m^3/day (75 mgd) into the normally dry Salt River bed. The Salt River floodplain is about 0.8 km (1/2 mile) wide and consists mostly of sand and gravel. While coarse gravel and boulders are exposed in the main channels of the floodplain, the soil profile of the rest of the riverbed typically consists of about 1 m of fine, loamy sand underlain by coarse sand and gravel layers to great depth (Bouwer, 1970). This is a desirable profile for rapid-infiltration systems because the top layer is sufficiently fine to filter out suspended material, to provide an adequate surface area for bacterial growth, and to have enough cation exchange capacity (clay content about 3%) for adsorbing ammonium from the infiltrating sewage effluent. On the other hand, the top soil at the Flushing Meadows Project still is sufficiently permeable (hydraulic conductivity about 1 m/day) to yield high infiltration rates (0.3-1 m/day). The underlying sands and gravels have such a high hydraulic conductivity (85 m/day in horizontal direction; Bouwer, 1970) that the renovated water is readily transmitted laterally through the aquifer without build-up of high groundwater mounds beneath the infiltration basins.

The height of the groundwater mound below the Flushing Meadows Project normally does not exceed 1 m above the static water table.

The Flushing Meadows Project consists of six parallel basins, 6.1 by 213 m each, and 6.1 m apart (Figure 1). Secondary effluent is pumped from the discharge channel into the basins. The inflow rate into each basin is controlled with an alfalfa valve and measured with a critical-depth flume (Replogle, 1975 and 1977). The water depth in the basins is controlled with an overflow structure at the outflow end of the basins, where the rate of outflow is measured with the same type flumes as at the inflow end. Water depths in the basins normally are held at 18 or 33 cm. The first 15 m of each basin was excavated deeper and separated from the rest of the basin by a 0.5-m high gravel dam to serve as a presedimentation reservoir for removal of large suspended solids. Observation wells consisting of 15-cm cased holes open at the bottom were installed on a line across the project half-way down the basins. The wells are located between pairs of adjacent basins and at various distances from the basins outside the project area (Figure 1). The East

Center Well (ECW) is 9.1 m deep, the West Center Well (WCW) 30.5 m, and the other wells 6.1 m. The water table normally is at a depth of about 3 m, but declined to 4.5 m and even deeper in dry years when groundwater pumping in the valley intensified.

HYDRAULIC LOADING AND NITROGEN REMOVAL

The primary objective of the Flushing Meadows studies in the first six years was to obtain maximum hydraulic loading. This was achieved with flooding periods of 2 to 3 weeks alternated with drying periods of 10 to 20 days, depending on rate of drying as affected by weather conditions. Regular drying periods are necessary to restore infiltration rates and to bring oxygen into the soil. At a water depth of 33 cm in the basins, these flooding and drying cycles produced hydraulic loading rates of 90 to 120 m per year. Growing grass in the basins or covering the bottom with a 15-cm layer of fine gravel offered no particular advantages. It was concluded that bare soil with volunteer vegetation was the most desirable bottom condition from a standpoint of economics of basin management and hydraulic loading. At a loading rate of 100 m per year, 12.8 ha of basins are needed to infiltrate the sewage effluent from 100,000 people (assuming a sewage production of 350 liters per person per day).

The main effect of maximizing hydraulic loading on the quality improvement of the effluent water as it moved through the soil below the basins was a relatively low rate (< 30%) of nitrogen removal, and possibly a slight reduction in PO_4-removal. Since PO_4-precipitation continued in the aquifer, however, the ultimate PO_4-concentrations in the renovated water about 100 m away from the infiltration basins were not significantly affected by hydraulic loading rate. Removal of other substances such as suspended solids, biodegradable organics (as expressed by the BOD), and fecal coliform bacteria was equally effective at high as at reduced loading rates.

Total-nitrogen concentrations in the secondary sewage effluent normally ranged between 20 to 40 mg/l. Most of this nitrogen was in the ammonium form. The major process whereby nitrogen can be removed from the effluent water as it seeps through the soil in rapid-infiltration systems is biological denitrification. Most of this denitrification occurs in the top 40 cm of the soil and during the beginning of a drying period (Gilbert, et al., 1974), as ammonium that was adsorbed in the soil during the previous flooding period is nitrified in the aerobic portion of the top soil. The resulting nitrate is then denitrified in microanaerobic zones in the otherwise aerated top soil. Not all nitrate is denitrified this way and when flooding is resumed, the remaining nitrate is leached out. This produces a nitrate peak in the renovated water from a certain well when effluent water that has infiltrated at the beginning of a flooding period reaches that well. For the East Center Well, these peaks occurred about 5 days after the start of a new flooding period (September-December period of Figure 2). This is the time it takes for the water to travel from the bottom of the basin to the intake of the well. As the renovated water continues to move through the aquifer, the nitrate peaks from successive flooding periods become flatter and eventually disappear.

If flooding periods are relatively short (for example, less than 2 or 3 days), the upper soil remains sufficiently aerobic for complete conversion of ammonium and organic nitrogen in the effluent to nitrate in the renovated water. When this occurs, nitrogen removal is insignificant (July-August, Figure 2). If, on the other hand, flooding periods are relatively long (more than 2 weeks for example), more ammonium is adsorbed in the soil during flooding than can be nitrified during drying. This reduces the ammonium adsorption during subsequent flooding periods and eventually leads to a gradual increase in the ammonium content of the renovated water. After 4 years of essentially maximum hydraulic loading, for example, NH_4-N concentrations in the renovated water had increased from low initial values of about 2 mg/l in 1968 (Figure 2) to almost 20 mg/l in 1971 (Figure 3).

At the maximum hydraulic loading rates, N-removal initially was about 30% but decreased as NH_4-N concentrations in the renovated water gradually increased. Since laboratory studies had indicated that the percent N-removal could be increased by reducing the hydraulic loading rate (Lance et al., 1976), studies were initiated at the Flushing Meadows

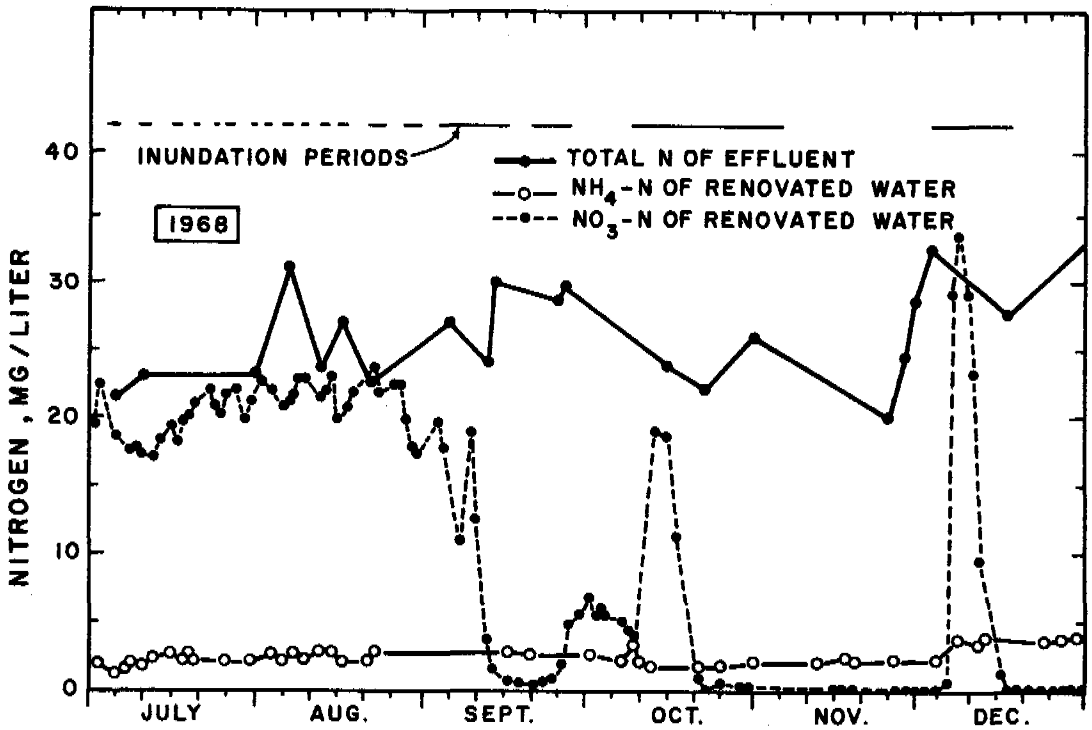

Figure 2. Nitrogen in sewage effluent and renovated water from East Center Well, 1968.

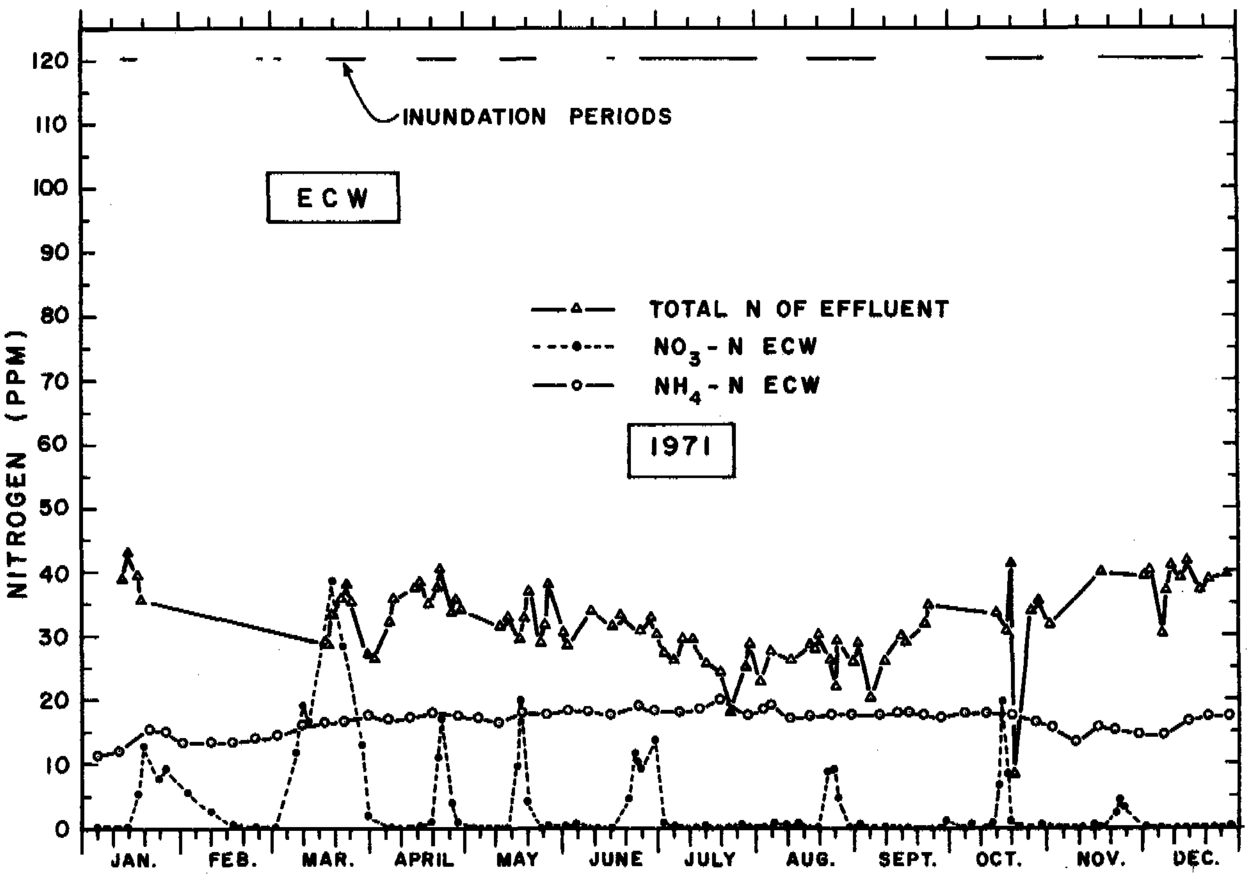

Figure 3. Nitrogen in sewage effluent and renovated water from East Center Well, 1971.

Project in 1973 to see if a reduced water depth (18 cm) and shorter flooding periods (9 days flooding and 12 days drying) would increase the removal of nitrogen from the effluent water. These changes reduced the hydraulic loading rate to about 60 m per year, but increased nitrogen removal to about 60%. The shorter flooding periods and lower infiltration rates apparently produced a better balance between ammonium adsorption during flooding and nitrification of adsorbed ammonium during drying, because the NH_4-N content of the renovated water began to show a gradual decline and reached an essentially steady level of about 1.7 mg/l in 1977. The average NO_3-N concentration of the renovated water in 1977 was 8.6 mg/l, while the organic-N content averaged 0.55 mg/l. The percentage nitrogen removal was calculated from infiltration rates and nitrogen concentrations in effluent and renovated water during flooding only, assuming piston flow from the infiltration basins to the well intake and static conditions during drying. The hydraulic loading rate in 1977 averaged 75 m/year for all six basins. This is not much lower than the maximum rates of 90 to 120 m obtained in the first 5 years of the project when hydraulic capacity was maximized. Nevertheless, nitrogen removal in 1977 had more than doubled, from 30% to 63%.

PHOSPHORUS

Most of the phosphorus in the secondary effluent at the Flushing Meadows Project is in the orthophosphate form, which precipitates as complex calcium phosphates (apatite, for example) in the soil and aquifer. PO_4-P concentrations in the effluent were 15 mg/l and more in 1969 (Bouwer et al., 1974), but averaged about 8 mg/l in 1977. This decrease is probably due to increased use of low-phosphate detergents. While PO_4-P removal in the first 9 m of downward movement (3 m above the water table and 6 m in the aquifer) below the Flushing Meadows Project generally ranged between 50 and 80%, the removal after additional movement of 30 m laterally through the aquifer exceeded 90%. After 10 years of operation of the project, in which a total of 760 m of effluent infiltrated into the soil (average for all six basins), there is no evidence of a decrease in the phosphate removal from the effluent water. Since phosphate may be the only substance to accumulate in significant amounts in the soil and aquifer materials, it may eventually restrict the hydraulic conductivity of the soil materials and hence the hydraulic capacity of the system. Preliminary calculations indicated, however, that this may not occur until after about 100 or 200 years of operation of the project.

FLUORIDE

The removal of fluoride from the effluent tended to parallel the removal of phosphate, indicating precipitation of calcium fluoride and fluorapatite. Fluoride levels in the effluent were around 4 mg/l in the beginning of the project (Bouwer et al., 1974) but gradually decreased to an average of 2.1 mg/l in 1977. F-concentrations in the renovated water in 1977 averaged 1.7 mg/l for the wells between basins, and 1.1 mg/l for Well 1 which is 30 m from basin 1 (Figure 1).

BORON AND METALS

Boron is not removed from the effluent water as it passes through the clay-poor sands and gravels of the Salt River bed. Boron concentrations in the effluent were around 0.7 to 0.8 mg/l in 1970-1972 (Bouwer et al., 1974). Fortunately, these values have dropped in recent years to around 0.5 mg/l, which is below the maximum limit for irrigation of citrus and other boron-sensitive crops (National Academy of Sciences and National Academy of Engineering, 1973). Of the heavy metals, the concentrations of zinc and copper were reduced as the effluent percolated through the soil. However, heavy metals are not a problem because the concentrations in the Phoenix secondary effluent are already below the maximum limits for irrigation water (Bouwer et al., 1974; National Academy of Sciences and National Academy of Engineering, 1973).

BACTERIA AND VIRUSES

Fecal coliform concentrations in the secondary effluent, which is not chlorinated, were on the order of 10^5 to 10^6 per 100 ml. Sometimes, fecal coliforms could be detected in renovated water sampled beneath the basins, especially after the start of a new flooding period when newly infiltrated effluent water had just reached the intake of one of the observation wells between the basins (Bouwer et al., 1974). Peaks of several hundred fecal coliforms per 100 ml have been observed at these times. As flooding continued, however, the fecal coliform concentrations in the renovated water sampled beneath the basins decreased to zero. This was probably due to increased clogging of the bottom and to increased activity and antagonistic effects of other microorganisms in the soil. Further removal of fecal coliforms took place with additional lateral movement of the renovated water in the aquifer. Thus, fecal coliform bacteria have never been detected in observation wells 60 m away from the basins. Viruses were present in the secondary sewage effluent at an average concentration of 21 PFU's per liter, but they could not be detected in the renovated water from the various observation wells. These results were obtained in a 1974 study in which viruses from large volumes of water samples were concentrated and assayed at 2-month intervals for the entire year (Gilbert et al., 1976).

ORGANIC COMPOUNDS

Biodegradable organic compounds in the effluent water were essentially completely removed by soil bacteria. However, although the BOD of the renovated water was about zero (compared to about 20 mg/l of the secondary effluent), the total organic carbon (TOC) concentration of the renovated water averaged about 4 mg/l, indicating presence of refractory or trace organics. These organics now are the main concern for possible potable use of renovated water, since their identity is not completely known and carcinogenicity or other toxicity has been implied (Shuval, 1977). Several studies are under way at various laboratories and universities to identify these trace organics and to evaluate their toxicity. Activated carbon adsorption before and/or after rapid infiltration and ozonation may be effective in removing the trace organics.

LONG-TERM PERFORMANCE

After 10 years of operation of the Flushing Meadows Project, there is no indication of a loss in hydraulic capacity or in renovation efficiency of the system. The 12-day drying periods are sufficient to restore infiltration rates, which generally decrease during flooding. If the suspended solids content of the secondary effluent is fairly low (less than 30 mg/l, for example), the basins can be operated for years without cleaning them to remove accumulated solids.

For deep, unconfined aquifers such as the one below the Flushing Meadows Project, the height of the pseudo-equilibrium groundwater mound during infiltration should vary linearly with infiltration rate. Thus, the height of the mound per unit infiltration rate should be constant, unless there is a change in the hydraulic conductivity of the aquifer. Since this ratio at the Flushing Meadows Project has remained essentially constant at about 0.8 days (0.8-m rise of water table per 1 m/day infiltration rate), clogging of aquifer materials and resulting reduction in hydraulic conductivity apparently has not occurred.

Future plans for the Flushing Meadows Project include studies on trace organics, enhancement of denitrification, fate of viruses in soil, phosphate accumulation in soil, and efficiency of renovation.

INSTITUTIONAL, AGRONOMIC, AND AESTHETIC REQUIREMENTS

The use of sewage effluent for crop irrigation is governed by state rules, which are based on public health concerns and not on agronomic suitability of the water. The strictest standards are for unrestricted irrigation, which includes sprinkler irrigation of lettuce and other crops consumed raw by humans. Arizona standards for unrestricted irrigation[1] require an effluent that has had primary and secondary treatment followed by additional treatment to produce a BOD and suspended solids content both of less than 10 mg/l, and a fecal coliform concentration of less than 200/100 ml. Such an effluent is then also suitable for use in recreational lakes with

[1]Arizona Department of Health Services Rules and Regulations, Chapter 20, Title 9, Article 4.

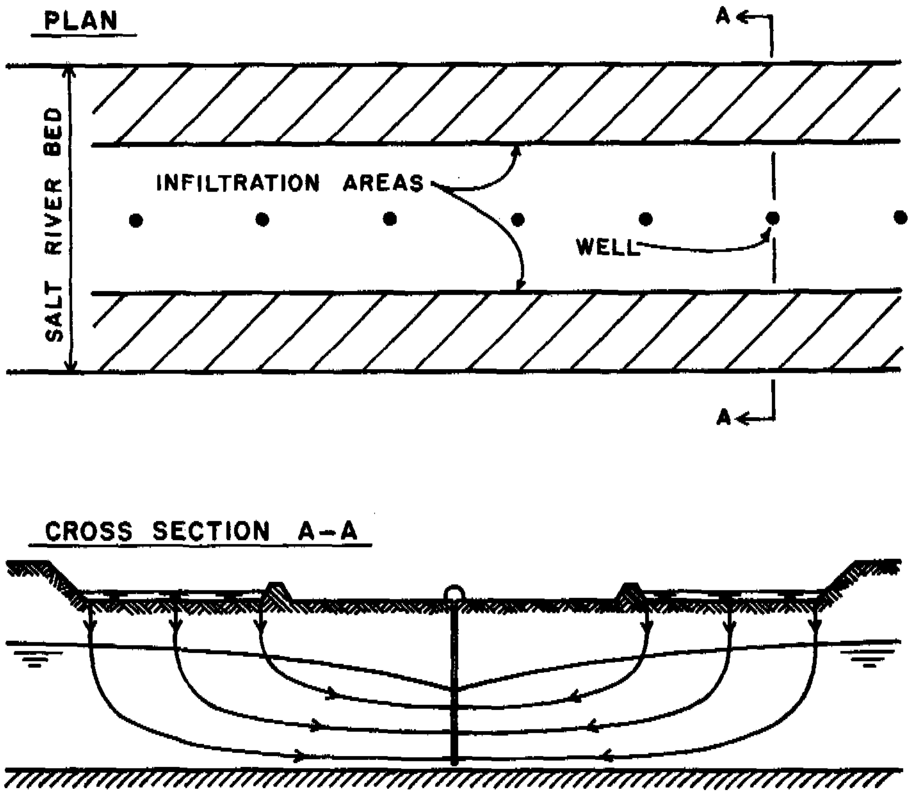

Figure 4. Schematic of system of infiltration strips and wells for collecting renovated water.

primary-contact activity. California regulations[2] are much stricter and require for unrestricted irrigation an adequately disinfected effluent (7-day median coliform count not in excess of 2.2/100 ml and a 30-day maximum not exceeding 23/100 ml), that is well oxidized (organic matter stabilized), coagulated (colloidal and finely divided suspended material removed), clarified (clarification of oxidized, coagulated effluent by further settling), and filtered (clarified wastewater which has passed through soils or filter media).

In addition to the public health aspects, there are also agronomic and aesthetic aspects to consider when using sewage effluent for irrigation. For example, the nitrogen content of the effluent should be reduced, preferably to the range of 0-5 mg/l where it will cause "no problems," but certainly to the low part of the 5-30 mg/l range where it may cause "increasing problems" (Ayers, 1977). Of course, if irrigation with sewage effluent is restricted to grasses or other forage crops, relatively high nitrogen concentrations can be tolerated and may actually be beneficial. Problems due to excessive nitrogen application to other crops may range from delay in crop maturity and harvest (cotton) to a lower sugar content (sugarbeets, sugar cane) and impaired quality of fruit and vegetable crops (Baier and Fryer, 1973). Public acceptance of the use of wastewater for irrigation or other purposes in populated areas probably is easier for renovated water from rapid-infiltration projects than for other upgraded secondary effluents. This is because the former is clear, odor free, and comes from wells, which at least mentally removes the connotation of purified sewage.

SYSTEM LAYOUT AND AQUIFER PROTECTION

When renovation of sewage effluent by rapid infiltration and groundwater recharge is practiced on an operational scale, it may be necessary to design and manage the infiltration and collection

[2] California Administrative Code, Chapter 4, Title 22, Division 4, Environmental Health.

systems so that all water that infiltrates as sewage effluent will be removed from the aquifer as renovated water. This can be accomplished by systems as in Figure 4, where infiltration and pumping rates are controlled so that the water table beneath the outer edges of the infiltration areas is not affected by the infiltration and the pumping, and, hence, remains at the same level as the water table in the aquifer adjacent to the infiltration system. In that case, no renovated water moves outside the system of infiltration basins and wells, and no native groundwater is drawn into the renovation system and pumped from the wells. Systems as in Figure 4 would avoid legal problems regarding withdrawal of native groundwater or loss of control over the renovated water after it has moved into the aquifer adjacent to the infiltration projects and beneath lands of other owners. Avoiding movement of renovated water into the aquifer adjacent to the rapid infiltration system also protects indigenous groundwater resources against encroachment of renovated sewage effluent, whose trace organics and other constituents would otherwise be a threat to high quality potable groundwater resources.

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LAND TREATMENT OF WASTEWATER IN BRAUNSCHWEIG AND IN WOLFSBURG, GERMANY

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INTRODUCTION

Land treatment of municipal wastewater has been a common practice at Braunschweig for many decades. In 1954, a group of some hundreds of farmers and city representatives assisted by the State, formed the Braunschweig Sewage Utilization Association (SUA) in order to solve the twin problems of final wastewater disposal and fertilizing agricultural land by extended sprinkler irrigation upon 3,000 ha of cropland.

There was already considerable local experience in wastewater utilization for crop production available. Before the turn of the century, a combined sewer system 100 km long was built in Braunschweig. It ends in a confusing system of dams and ditches on a sewage farm of 350 ha. Braunschweig was one of those German towns which followed earlier English examples and the recommendation of Justus Liebig to reclaim the plant nutrients in wastewater instead of letting them run into the rivers to the ocean.

In 1895, the sewage farm at Braunschweig made available 350 ha for a population of 115,000, a ratio of about 330 inhabitants per ha of farmland. Already 30 years afterwards, it was recognized that this ratio could not be kept in the proper balance. The population and the quantity of sewage grew continuously, but the sewage farm did not. Much of the collected water did not reach the fields, it was discharged into the river. A project was considered to enlarge the sewage farm many times, to collect and treat the wastewater from a large area, including many communities and three towns, all together about twice the population of Braunschweig. This project was not implemented because of World War II. Afterwards, the new beginning was characterized by a lack of food and fertilizer, a great surplus of sewage, and 50 years of experience in the utilization of sewage as fertilizer.

The great demand for food; the technical progress in equipment development for sprinkler irrigation; and, very important, the encouraging results of bacteriological investigations by Popp et al. (1955, 1956) in sprinkler irrigation of wastewater on farm land near the city of Wolfsburg were all favorable arguments for the proposal to establish the Sewage Utilization Association of Braunschweig in 1954. The union combined the City of Braunschweig and about 350 farmers. They made available a total area of 4,200 ha including 3,000 ha of sprinkler irrigated cropland north of Braunschweig and north of the sewage farm, a sandy land strip between the rivers Erse and Oker, about 15 km long and 3 km wide (Figure 1). The ratio of population equivalent (p.e., including industry) to irrigation area, both 350 ha sewage farm with surface flooding and 3,000 ha of the association, shifted to 350,000 p.e. and 3,350 ha or 104 p.e. per ha, in 1977 (Table 1).

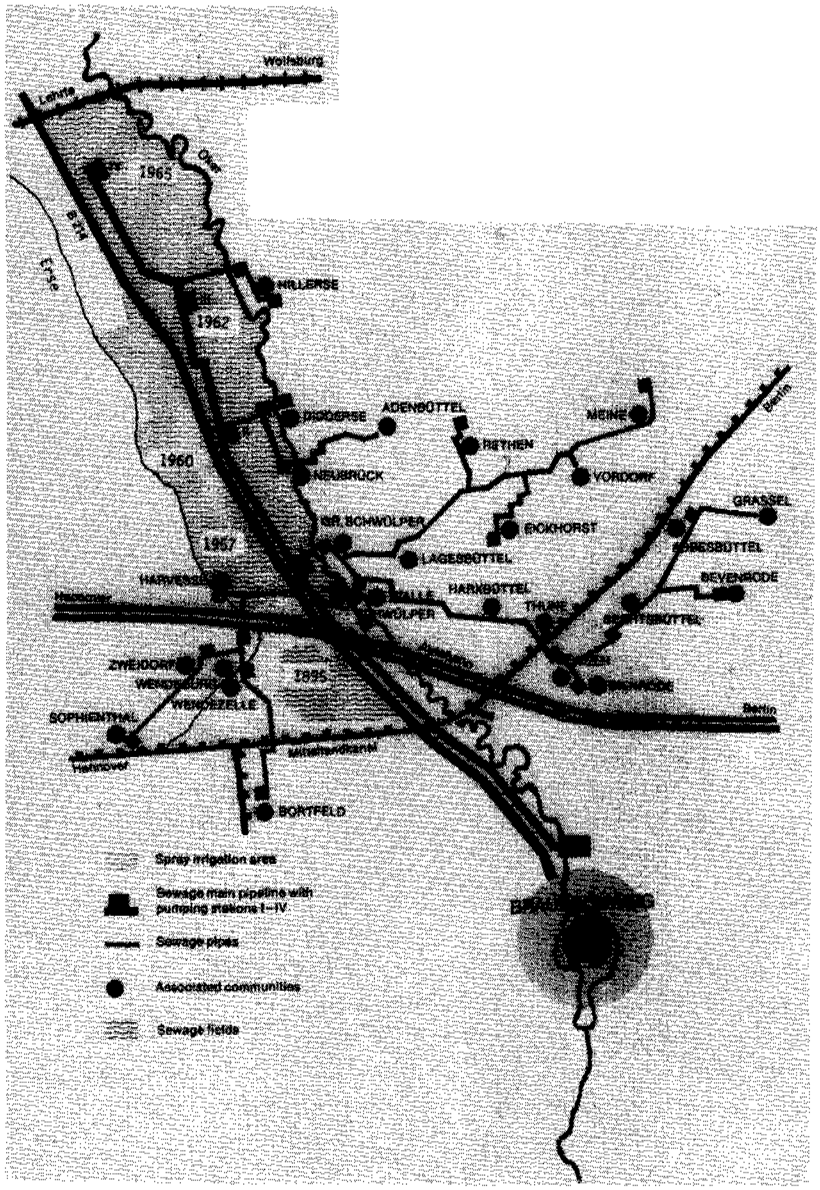

Figure 1. Sewage Utilization Association, Braunschweig

THE SEWAGE UTILIZATION ASSOCIATION OF WOLFSBURG

Wolfsburg was a village before the Volkswagen factory was established there. In 1939, the City of Wolfsburg, some smaller communities and about 100 farmers founded the Sewage Utilization Association to serve 15,000 inhabitants. Today a sewage flow of 19,000 m^3 per day is generated by a population of about 125,000. The facilities are being extended to treat the wastewater from 170,000 inhabitants (Figure 2).

The farmers make available a total area of 2,100 ha including 1,200 ha of sprinkler irrigated cropland and 310 ha of woodland. The percentage of area under cultivation for different crops is shifting in favor of spring grain and sugar beet (Table 2). These crops make a better use of the wastewater and have a higher market value.

The wastewater is collected at the pumping station of Wolfsburg. After primary treatment including three hours of sedimentation it is pumped to the irrigation area where every year 250 to 350 mm are spread on the fields by sprinkler irrigation in five to seven applications of 50 mm per day. The available woodland is considered as reserve area to spread a surplus of wastewater by surface flooding or sprinklers (Sommer, 1976).

Table 2

Wolfsburg SUA, irrigated crops, percentage of area

	1965	1976
Spring grain	16	32
Winter grain	27	31
Potato	22	7
Sugar beet	13	29
Grassland	22	1

Table 1

Wastewater at Braunschweig in 1977 and prognosis for 2000

	1977	2000
Quantity per year	18 million m^3	25 million m^3
Population equivalent	350,000	480,000
Treatment per day		
Summer		
Sewage farm	13,000 m^3	25,000 m^3
Sewage association	40,000 m^3	47,000 m^3
Winter		
Sewage farm	20,000 m^3	30,000 m^3
Sewage association	29,000 m^3	36,000 m^3

Because nuisance odors are not completely excluded, experimental installations have been established for treatment with oxygen in the pressure line and for intensive aeration in storage basins.

Wolfsburg and Braunschweig have the same climate and the same farming conditions; the distance between the two cities is only 25 km. Knowledge and experience of sewage utilization for crop production in a well organized irrigation operation could be transferred without difficulty from one site to the other. The favorable results of hygienic investigations and official supervision were an important basis for the foundation of the Braunschweig association (Bringmann and Kuehn, 1957; Popp, 1957).

THE SEWAGE UTILIZATION ASSOCIATION OF BRAUNSCHWEIG

The Organization

The Association today is made up of the City of Braunschweig, 26 nearby communities and 310 farmers. The Association is headed by a committee of three farmers and two representatives of the city. The landowners, communities and city administration are represented in a commission of 24 members. The water resources board and the chamber of agriculture are the controlling and advisory bodies. State control is exercized by the president of the administration district.

The manager of the Association is an agronomist. At his disposal are

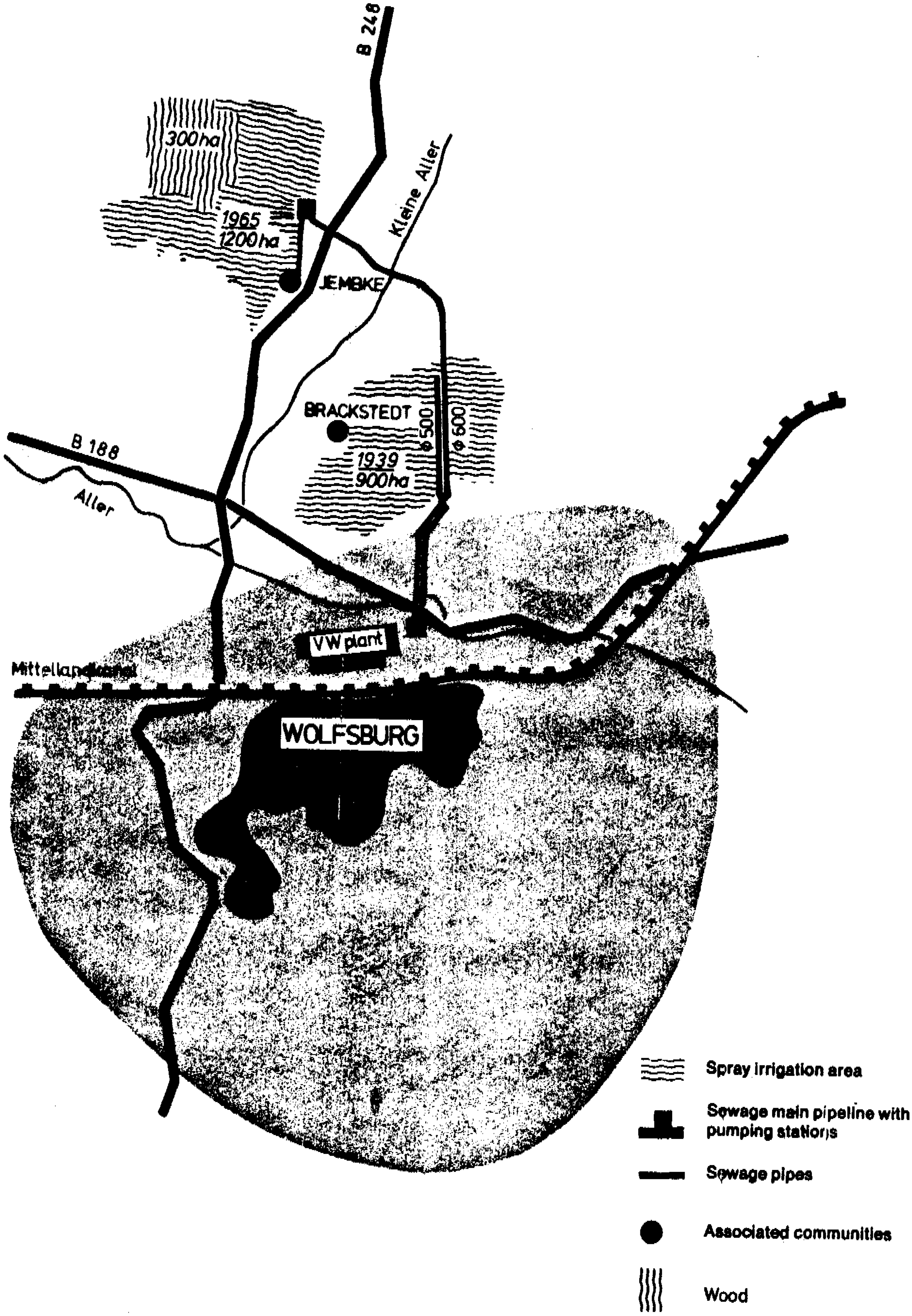

Figure 2. Sewage Utilization Association, Wolfsburg

an office, a workshop, five irrigation specialists with assistants, and four pump operators. They have the task of providing for a controlled use of the wastewater. This involves operating the four pumping stations, shifting the sprinkler equipment, maintaining and renovating Association installations including 90 ha of shelter hedges, a drainage system with subsurface pipes and open ditches, and also carefully advising agricultural members.

Annual fees from the members cover operating costs. According to the budget plan for 1978, the city is required to pay 2 million DM and farmers owe 480,000 DM. The present economic situation of Agriculture permits no considerable increase in farmers' contributions in the foreseeable future. In accordance with the Association's charter, the share of the city is limited by the costs which Braunschweig would otherwise have to bear if operating its own sewage treatment plant. (Schaerff, 1977)

The System

Braunschweig's wastewater is collected in a pumping station downtown. Here it is pumped through a concrete pipe gravity line to the transfer station. The line is 6 km long and has a capacity of 2,700 m^3 an hour.

The Association takes the flow from the transfer station through its own gravity line to the four pumping stations. This conduit of 15.4 km consists of concrete pipe at the beginning and asbestos cement pipe for the remainder. The pumping stations have the task of generating the required pressure to the sprinklers. Each station has four spray pumps with capacities of 400 m^3 an hour; they pump the water from the storage and settling basins with a total capacity of 15,500 m^3 into the underground pressure line. Holding time for settling the suspended solids varies from one-half to several hours, depending on flow rates from the generating sources. The sludge produced daily at the settling basins is pumped into earthen sludge basins of 5.6 ha with a depth of 1 m. After several months of digestion and dewatering, the sludge is carried off and spread on cropland.

The water is distributed through an underground network of asbestos cement pipe more than 100 km in length. Underground hydrants at intervals of 90 m are used to transfer the water from this underground network into movable pipelines. In 1972, the Association began to phase out the original portable hand-moved irrigation pipes and to introduce spraying machines with flexible polyethylene plastic pipes about 300 m in length stored on a drum. The machines are pulled into place by a tow truck. A spray head attached to the end of the plastic pipe sprinkles that water over an area of 50 to 60 m in diameter with a pressure of 6.5 bars at the 22-mm nozzle. The plastic pipe is retracted by a turbine driven by the water of the pressure pipe line.

One hundred of these irrigation machines are necessary to irrigate the 3,000 ha. Instead of the 20 spray attendants employed in the original system, only 7 are now required.

During dry summer seasons, the daily flow of wastewater is not sufficient to match the water requirements of all crops. Wells have been installed at the pumping stations for this purpose to make water available from underground.

Excess wastewater on Sundays, holidays or during freezing periods, about 65 days a year, is collected in percolation basins which cover 14.5 ha, or it is sprinkled on arable land with dormant crops.

For the most efficient use of labor and equipment, land consolidation is helpful and necessary in order to adjust highways and roads, the size and shape of the fields, the subsurface pressure line with hydrants and junctions, subsurface pipe and ditch drainage, shelter hedges and woodlands, and even land use and the cropping system. Fifty km of roads have been constructed; hedges established to protect access roads, paths and settlements from both spray and wind erosion cover 90 ha in strips of 10 m depth; the drainage system of open ditches which is necessary to collect the seepage of percolated and natural groundwater and to carry it to the rivers has a length of 22 km. Much has been done already, but up to now only part of the large project has been finished and efforts continue to improve the process, the operation, the results in pollution abatement and in crop production.

Nuisance odors are not a widespread problem at the installations of the Braunschweig Sewage Utilization Association. The final solution to this seasonal and limited difficulty will consist of aerating the water before sprinkling. After extensive experimental work, R. Kayser, Technical University Braunschweig, submitted the proposal of a sewage pretreatment plant (Figure 3). The wastewater will flow through a screening chamber, 20 mm aperture, into an aerated grit chamber, 425 m^3, and will be treated by 14 dual mammoth rotors, each with a 75 kW motor, in two activated sludge tanks, total capacity 17,000 m^3; secondary settling follows in two basins which hold 10,200 m^3. The problem of sludge disposal will be eluded by re-injecting the sludge into the irrigation pipe line for land treatment and utilization as fertilizer. The installations for the Sewage Pretreatment Plant are now under construction; the outline is projected for a daily wastewater flow of 72,000 m^3 with 25 t BOD, equivalent to 550,000 inhabitants. The operation will be started in 1979.

The Site: Soil and Climate

The soil and climate are favorable for a supplementary water supply. Except for a more clayey strip on both sides of the River Oker, the whole area is characterized by an extremely uniform layer, 20 to 40 m thick, of fine to coarse to gravelly sand without stones. This sand is of very low fertility with less than one percent of organic matter in the humus layer. The groundwater level is between 1.75 and 3.50 m. Damage by wind erosion occurs every spring and fall. There are almost no elevations in the level surface, but the altitude decreases from south to north by 9 m. This smooth grade of about 0.05 percent facilitates the transport of the sewage in a gravity line and also controls the groundwater flow.

The average mean value of the temperature is 8.5°C. In the growing season from April to September, it is 13.9°. The corresponding values for precipitation are 655 mm a year and 393 mm during the growing season.

The balance of precipitation and evaporation is a positive value of 125 mm per year with a negative value of 44 mm in the growth period. This explains the success of the supplementary water supply if it is applied to meet the needs of the growing crop. This also explains the farmers' active interest in this water source; they have experienced years with a still greater shortage of rainfall, and they know its catastrophic effect on plant growth and yields.

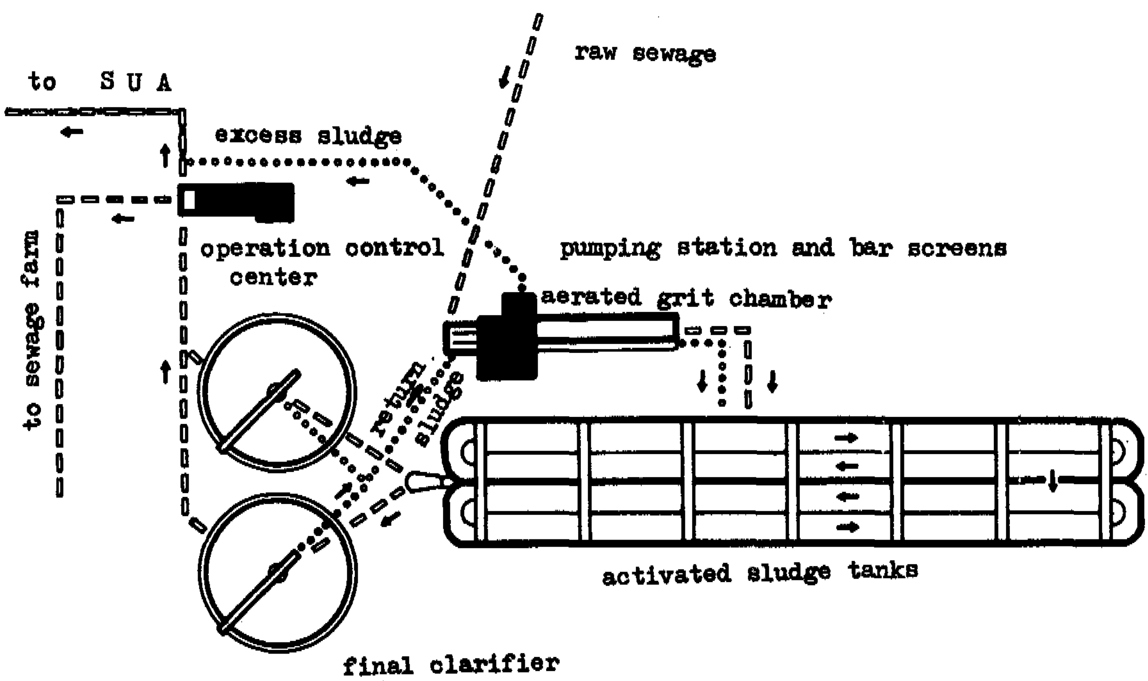

Figure 3. Sewage Pretreatment Plant, Braunschweig

According to the climatic water balance sheet and the available 300 sprinkling days over the entire year, a total of 300 mm of water was estimated to be the optimal supplement needed.

Agriculture

The wastewater is spread on the fields by sprinkler irrigation in six applications of 50 mm, three in summer and three in winter. That is the average; more exact amounts applied to various crops are as follows:

potatoes	2 applications of 30 mm
Winter grain, spring barley	3 applications of 50 mm
Oats	4 applications of 50 mm
Spring wheat, sugar beet	5 applications of 50 mm

Very soon after the beginning of the wastewater utilization at Braunschweig, an experimental field was established with all crops grown in the area of the Association. According to these experimental data, sugar beets and potatoes show a remarkable increase in yield from wastewater application particularly in dry years when the natural water supply is insufficient. For small grains, however, the wastewater is beneficial every year. This indicates that not only the water but also the nutrient constituents are useful if applied with rather small amounts of water and leaching is restricted. With the annual irrigation rate of 300 mm, the quantity of nutrients applied per ha is: 150 kg nitrogen, 40 kg phosphorus, 100 kg potassium, and small amounts of micronutrients as well as heavy metals (Table 3).

Winter and summer cereals, potatoes, sugar beets and asparagus are the principal crops. It is noteworthy that the cultivation of sugar beets, wheat and barley is made possible with an economical success on the area's sandy soils because of the wastewater irrigation. The farmer not only achieves greater harvests with the help of spray irrigation; above all, harvests can now be reliably planned year after year.

According to experience and experimental data, spring grain makes the best use of the wastewater. This influences the ratio of crops grown in the area. The portion of spring grain increased considerably whereas winter grain decreased (Table 4). The cultivation of sugar beets and potatoes is extremely important for a controlled irrigation operation, for they are irrigated at different times from those of cereals. The tendency to grow more cereals and few potatoes because cereals are selling for just as much and cost less to produce is not unlimitedly desirable because it results in peak periods of demand for irrigation.

To counter this trend, a dehydration plant was built to dry potatoes and produce potato pulp for animal feeding. The facility is also used for drying grain, grass, draff, sugar beet chips. Quantities treated up to 1977 were: potatoes 118,000 t, draff 64,500 t, grass 6,000 t, sugar beet chips 4,000 t, grain 2,800 t.

Except for asparagus which is sprayed only during the non-harvest months of June to December, vegetables and low growing fruit are not irrigated. All crops are processed before human or animal consumption so there is no danger that any pathogenic organism remaining in the sprayed wastewater and soil can be transferred. Cattle cannot graze on irrigated pastures until three weeks after spraying.

Minimum distances are maintained between irrigated and inhabited areas and public roads. In addition, in the vicinity of these areas irrigation machines with less aerosol forming are used and spraying is stopped during periods of high winds. Two independent irrigation inspectors are continuously controlling this regulation.

Studies on heavy metals in the wastewater, soil, soil-water and groundwater have revealed that the concentrations are far below the standards established by the Federal Government (Bramm, 1976; Bramm et al., 1977; Sommer, 1976). Toxic accumulations in the soil matrix are not to be expected provided that the rate of irrigation and the concentration in the wastewater are not increased. This also prevents the pollution of the groundwater as can be concluded from investigations of the heavy loaded soil in the area of the old sewage farm and also from migration experiments with radioactive elements in undisturbed soil columns (El-Bassam, 1977; El-Bassam et al., 1977).

Table 3

Braunschweig SUA, water quality data 1973/74

	Sewage	Oker River	Erse River	Groundwater Inside of the irrigation area	Groundwater Outside of the irrigation area
	n = 16 mg/l	n = 12 mg/l	n = 12 mg/l	n = 242 mg/l	n = 58 mg/l
NH_4-N	49	7.0	14.2	2.8	2.9
NO_3-N	0.2	8.4	7.0	30	8.7
P	13	0.9	0.7	0.5	0.4
K	32	11	55	33	85
Na	77	69	98	57	50
Cl	128	153	454	133	158
SO_4	134	202	286	203	237
Fe	2.0	1.2	0.8	12	8.3
Zn	0.9	0.6	0.5	0.4	0.7
Cu	0.15	0.03	0.04	0.06	0.05
Mn	0.3	0.4	0.9	1.7	2.1
Co	0.2	0.12	0.27	0.14	0.19
Cd	0.02	0.01	0.02	0.01	0.02
Pb	0.04	0.02	0.03	0.07	0.04
BOD	192	8	17	1.6	1.7
E C mmho/cm	1.11	0.98	1.91	1.04	1.35
pH	7.1	7.2	7.1	6.7	7.0

Sewage: Pumping station no. 2
Oker and Erse Rivers: Samples taken before the rivers reach the area
Groundwater: 21 observation wells in district no. 1; 5 wells beyond the rivers; groundwater table 0.5 - 3.5 m
n = number of samples

RESULTS AND CONCLUSIONS

Final and complete sewage disposal is a difficult task for sanitary engineers who must protect the environment. The Braunschweig Sewage Utilization Association has shown that its land treatment system is very effective in eliminating pollutants from water courses that flow through large and densely inhabited areas. The Oker River's water downstream of the city and contributing communities is cleaner than the water on the upriver side.

High efficiency is also obtained in land conservation and erosion abatement. Spring sandstorms which regularly appear and used to blow away the topsoil with the first sowing alarm farmers no longer. Crop production on the sandy soil with its very low natural fertility level has improved definitely; farmers of the Sewage Utilization Association can compete in farming success with their colleagues who cultivate better soils.

To return waste into the natural cycle of transformations by land treatment does result in final waste disposal. However, land disposal and land treatment must be considered as separate alternatives. The goal of maximum disposal may be mutually exclusive with

Table 4

Braunschweig SUA, crops growing

Percentage of area under cultivation, district 1

Crop	1957	1960	1965	1970	1975
Winter grain	36	23	20	20	26
Spring grain	10	19	28	31	29
Potato	24	19	14	10	6
Sugar beet	12	19	19	18	21
Asparagus	8	10	18	10	9
Grassland	10	10	11	10	9

the goal of maximum utilization. To produce crops, land treatment of waste must accommodate the natural conditions of the site and the properties of the waste. In fitting cities as well as agricultural areas into the biosphere, designs are needed to make these cycles beneficial. Along with proper engineering design and management, site selection must be recognized as an important ingredient for lasting success (Hartman, 1974). Biologically based recycling systems should be designed so as to optimize the production of economically and socially useful products. Standards for waste utilization in crop production should provide reasonably high levels of public health protection as well as high levels of crop growth-promoting and soil-improving constituents in the waste. Maximum crop growth is a biological proof of an ecologically balanced recycling system.

Remarkable efficiency is obtainable by land treatment of wastewater in a cooperative program of sanitary engineering, water resources policy and agriculture if the soil and climatic conditions are favorable and the farmers' economic situation can be improved. This has been demonstrated by the Braunschweig and the Wolfsburg SUAs since 1957 and 1939, respectively.

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EVALUATION OF EXISTING SYSTEMS

WATER POLLUTION CONTROL THROUGH LAND DISPOSAL OF SECONDARY-TREATED WASTEWATER EFFLUENTS

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ABSTRACT

To prevent the pollution caused by direct discharge of sewage effluents into surface waters, a wastewater treatment scheme is proposed where the effluent is applied to agricultural lands in place of ordinary irrigation water. The environmental effects caused by this scheme are evaluated by systematic measurements of soil, wastewater, and agricultural products. The results indicate that there are no irreparable short-term or long-term hazards associated with the scheme. Measurements regarding budgets of the main nutrients i.e., nitrogen, phosphorus, and potassium, are reported and indicate negligible probability of underground water pollution, even if the technique is utilized for an extended period. Moreover, an increase of nearly 25% in growth rate and final yield density is achieved when wastewater-irrigated plots are compared to their corresponding reference ones where fresh water, with or without commercial fertilizers, is used for irrigation. Based on experimental results obtained to date, it is proposed that the wastewater treatment renovation scheme results in little or no environmental pollution hazards, recharges the much needed groundwater supply under certain conditions, and produces valuable food in the process. As such, this scheme is extremely useful for semi-arid countries such as Iran, where agricultural activities are generally limited by available water resources.

INTRODUCTION

During recent time the world has become increasingly aware of irreparable damages that uncontrolled disposal of treated or untreated wastewater effluents may inflict upon the receiving water bodies. To help alleviate this problem, field experiments have been conducted where secondary-treated sewage collected from the City of Isfahan has been sprayed onto various crops, including alfalfa, corn, and potatoes. The scheme is based on the "living filter" concept (1-3), where the soil and plants treat the nutrient-laden wastewater.

The number of independent variables present in the ecosystem under consideration are so numerous that meaningful results may be obtained only if comparisons are made between wastewater-irrigated plots and some suitable reference plots. Therefore, for each wastewater-irrigated plot, two reference plots with the same crop are irrigated with an equal amount of fresh water for the purpose of comparison. One of the reference plots receives no commercial fertilizer at all, while the other receives a typical amount of a suitable mixture of chemical fertilizer. The fertilizer applications of nitrogen, phosphorus, and potassium are adjusted so that they are similar to those received by the main plots as a result of irrigation with wastewater. In this way, comparative analyses may be made on the basis of the results obtained from the wastewater-irrigated and reference plots.

EXPERIMENTAL RESULTS AND DISCUSSION

Systematic measurements of parameters believed to be most significant in the system were made during the cultivation period. These measurements included chemical analyses of soil, plant tissues and organs, fertilizers, and atmospheric precipitation as well as routine tests that were conducted on the water and wastewater utilized for irrigation. The main emphasis of chemical analysis has been on determination of the three main nutritive elements present in various forms while physical tests were aimed at evaluation of parameters representing integrated effects, such as the product yield.

Product Yield

Annual yields for the three plant varieties tested during recent years are shown in Table (1). It is noted that the yields obtained in the case of wastewater-irrigated plots are invariably higher than those irrigated with fresh water, even when equivalent amounts of fertilizer were applied. This probably reflects the advantage of gradual and uniform nutrient application during the irrigation period as compared to the commonly practiced method where all the fertilizers are applied either at once or in a few applications during the cultivation period.

Chemical Analyses

Standard methods were used to sample and analyze the amount of nitrogen, phosphorus, and potassium in soil, plant tissues, irrigating waters, and the various fertilizers used. Table (2) shows a summary of results of some of the important tests that are used in calculation of nutrient balances.

The analyses reported for wastewater indicate a substantial amount of nutrients which may be readily absorbed by growing vegetation. On the average, application of 1 cm of wastewater is calculated to provide commercial fertilizer constituents equivalent to 1.8 kg of nitrogen (N), 2.1 kg of phosphate (P_2O_5), and 2.2 kg of potash (K_2O), for each hectare of land irrigated with wastewater. This would be nearly equal to applying 6.5 kg of a 7-8-10 commercial fertilizer per hectare. (The application rate in the experiments reported averaged about 5 cm/week during the growing season.)

Nutrient Balances

In principle, a balance may be obtained for each nutrient once it is introduced into the system through application of wastewater, commercial fertilizers, and atmospheric precipitation and is transported out of the system as a result of product harvest, cut weeds or sublimation, or is dissolved and carried away in the percolate. The task here has been made simpler by several experimental observations made in regards to water percolation following each irrigation (4). Humidity measurements indicate that the soil humidity below 125 cm depth does not change as a result of irrigation. This implies that the nutrient movements are restricted to the top 125 cm of soil, and layers below are scarcely affected by the leachate under the present operating conditions. Therefore, by comparing the nutrient content of the top soil layers before planting and after harvest, one may obtain the amount accumulated in soil during the cultivation period.

Figures 1a to 1c illustrate graphically the balances obtained for nitrogen, phosphorus and potassium in the system. Calculations are based strictly on experimental observations which include both physical and chemical measurements. The method for determining the balance is straightforward: the amount of each nutrient included in wastewater, fertilizers, atmospheric precipitation, and that absorbed from air (only in the case of leguminous family plants) are regarded as an "income" to the system (left columns in each graph of Fig. 1) while those included in the harvested biomass and sublimation (only in the case of nitrogen) are considered to be an "outcome" to the system.

The effect on soil (i.e. the net deficit or accumulation) is assumed to be proportional to the area between the nutrient-abundancy distribution curves obtained as a function of soil depth at the beginning and end of the cultivation period. The curves are based on chemical analyses of composite soil samples extracted from layers 25 cm apart down to a maximum depth of 175 cm. If positive, the area values calculated represent a net accumulation in soil, to be regarded as an "outcome" from the system and, if negative, they represent a net deficit of the corresponding nutrients in soil, which is regarded as an "income" to the system.

Table (1) Annual yields in tons/hectare of alfalfa, corn, and potato in various plots during 1976 and 1977.

	1976			1977		
	SECONDARY TREATED WASTEWATER	FRESHWATER WITH FERTILIZER	FRESHWATER WITHOUT FERTILIZER	SECONDARY TREATED WASTEWATER	FRESHWATER WITH FERTILIZER	FRESHWATER WITHOUT FERTILIZER
ALFALFA FIELD	128.6	109.7	105.5	105.0	80.5	76.0
POTATO FIELD						
Product	8.6	7.2	5.1	12.8	12.4	8.1
Leaves	-	-	-	29.0	28.5	27.0
Weeds	46.4	38.2	36.5	59.0	29.0	29.0
CORN FIELD						
Grains	4.8	4.2	4.0	6.2	4.4	4.1
Leaves	-	-	-	3.8	3.4	3.1
Hub	-	-	-	1.2	0.8	0.7
Branches	-	-	-	9.5	9.3	8.5
Weeds	-	-	-	36.8	26.1	25.8

Table (2) A summary of chemical analyses of various parameters. The mean values signify averages obtained from wastewater-irrigated and reference plots.

	NITROGEN			PHOSPHOROUS			POTASSIUM		
	MIN	MAX	MEAN	MIN	MAX	MEAN	MIN	MAX	MEAN
WASTEWATER (ppm)	16.8	24.2	21.0	6.2	7.2	7.0	15.0	22,0	18.5
FRESHWATER (ppm)	1.8	2.8	2.5	0.2	0.35	0.3	2.8	3.6	3
ALFALFA (gr/100gr)	2.14	2.45	2.30	0.26	0.28	0.27	2.1	2.4	2.3
CORN (gr/100gr)									
Grains	2.0	2.3	2.1	0.20	0.24	0.22	0.54	0.68	0.61
Leaves	1.43	1.61	1.52	0.25	0.29	0.27	1.03	1.23	1.10
Hub	0.60	0.76	0.71	0.13	0.18	0.16	0.68	0.86	0.78
Branches	0.7	0.94	0.82	0.23	0.26	0.24	0.82	0.94	0.88
Weeds	2.4	2.6	2.5	0.36	0.37	0.35	3.10	3.24	3.14
POTATO (gr/100gr)									
Product	1.8	2.2	2.0	0.12	0.15	0.13	3.5	4.7	3.19
Leaves	1.7	2.4	2.1	0.22	0.31	0.26	4.4	5.4	3.6
Weeds	1.21	1.23	1.2	0.19	0.21	0.20	5.0	5.2	5.1

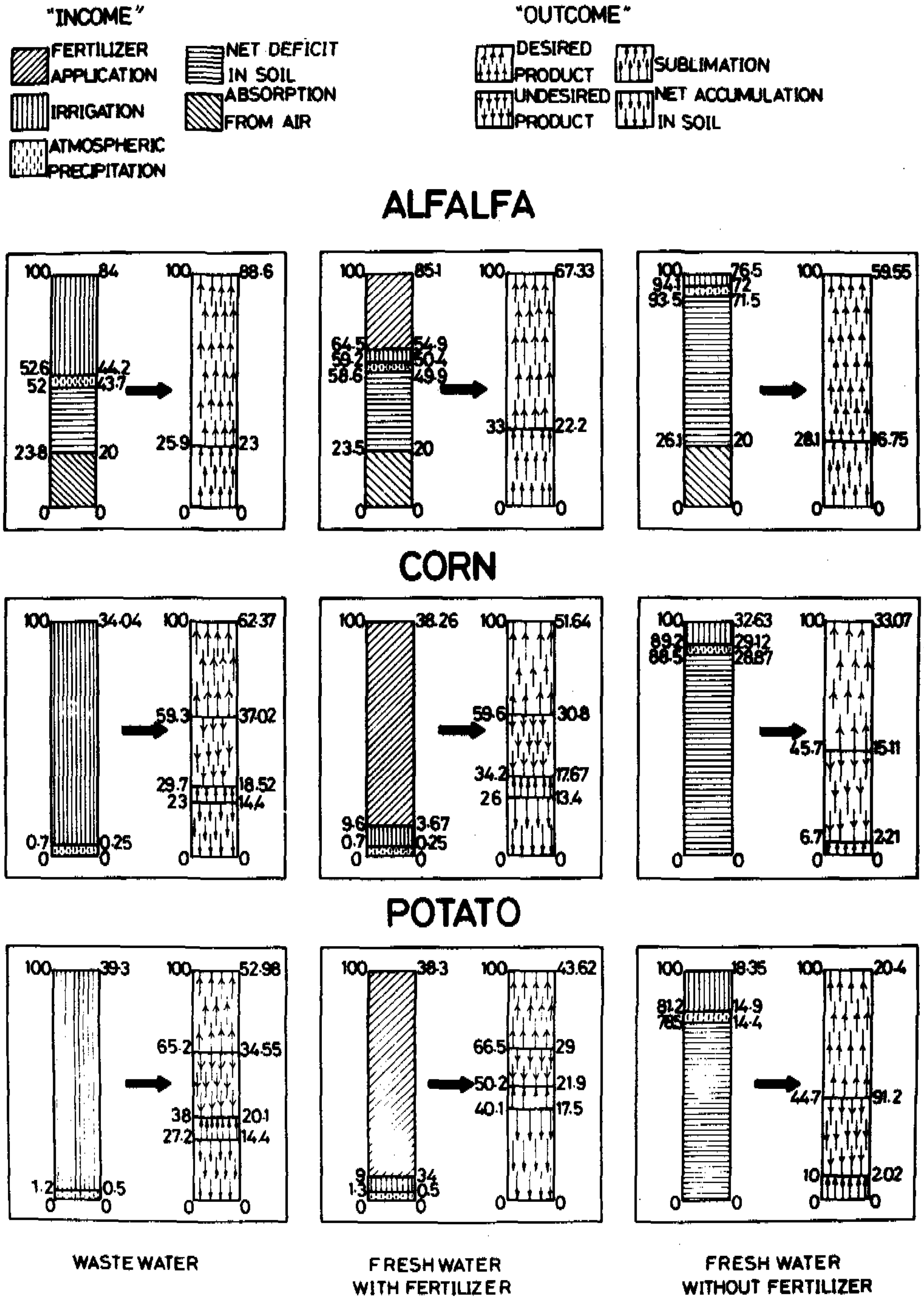

Fig. 1a. Balance of nitrogen.

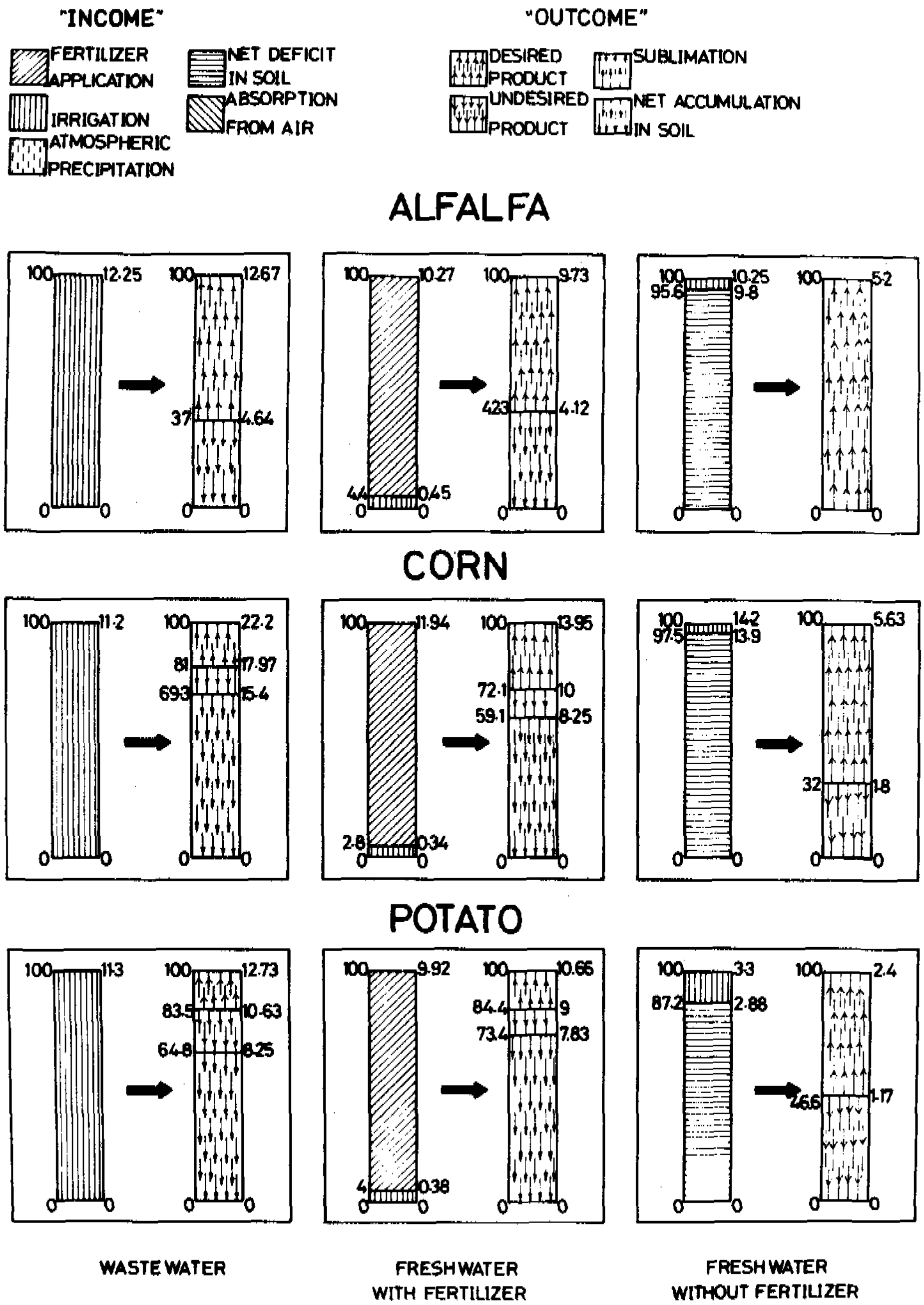

Fig. 1b. Balance of phosphorous.

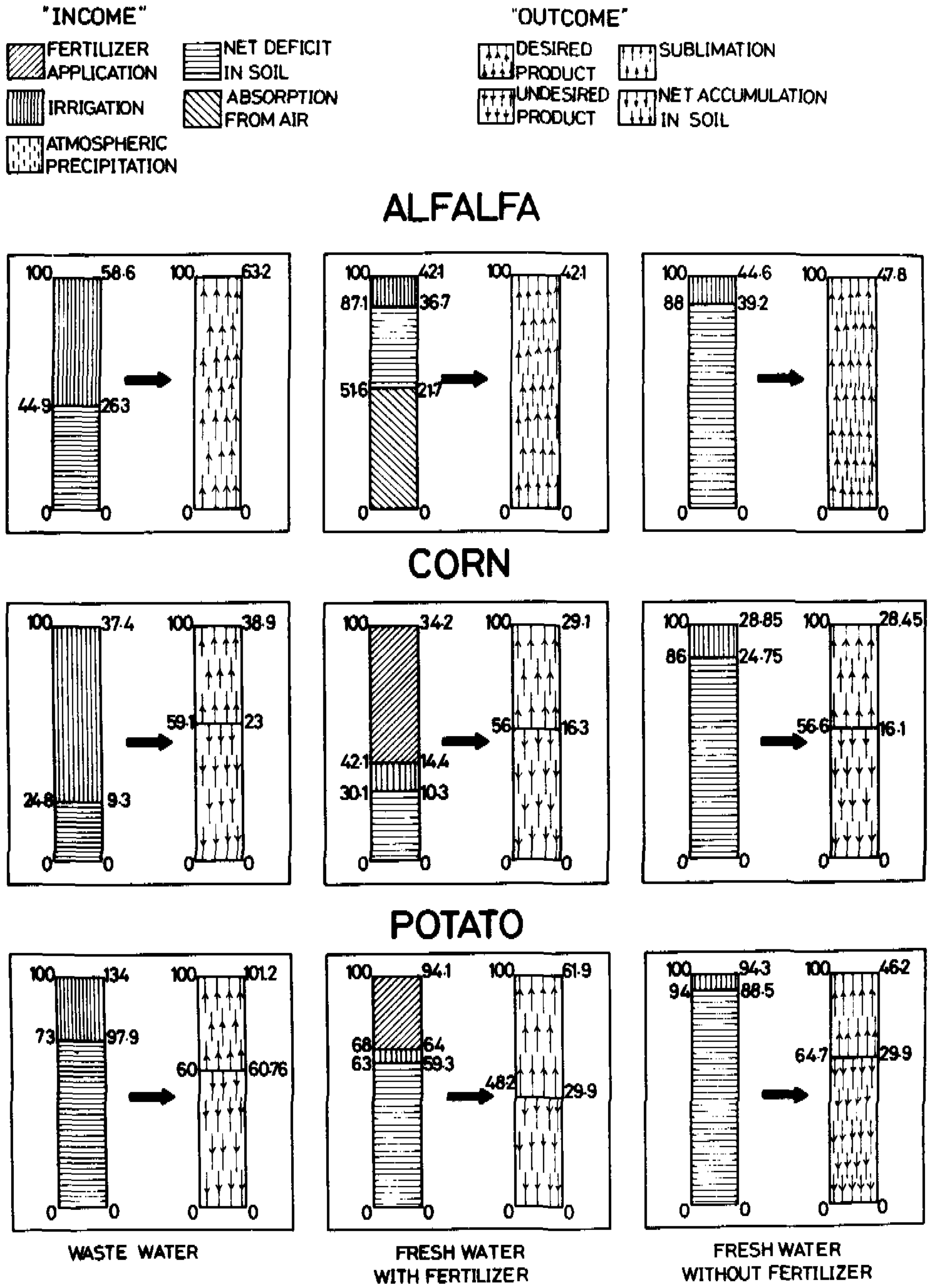

Fig. 1c. Balance of potassium.

As noted from Fig. 1, all the reference plots receiving no commercial fertilizers, as well as many of the other plots, indicate a negative accumulation or a deficit. In such cases, the amount of nutrients added as a result of irrigation or fertilizer application has not been sufficient to meet the amount required by the growing plant. Therefore, to meet their growth requirements, the plants have had to draw upon the nutritive elements residing in soil "reservoir", thus causing a net deficit at harvest time.

The numbers appearing at the right of each of the indicator columns in Fig. 1 represent the actual values calculated on the basis of experimental measurements. Their corresponding normalized values, or percentages, are shown at the left; theoretically, the totals marked at the top of every "income" and "outcome" column in each case must be the same. However, due to many unavoidable experimental approximations, some of the totals obtained are not exactly equal. The discrepancies involved are noted to be mostly below 20%, indicating a reasonable accuracy in the current calculations.

Despite the approximations mentioned above, the material flow analyses just described illustrate the possibility of a build-up in nutrients when continuous irrigation with wastewater takes place. However, by exercising careful management with regards to frequency and amount of wastewater applied and, also, by making a proper choice of vegetation, such shortcomings may be resolved such that a hazard-free system with long term treatment capability evolves for a given set of conditions.

Removal Efficiency

Regarding the whole scheme as a system for removing the eutrophicant agents contained in the wastewater, a removal efficiency may be determined and compared to that reported by a similar project at Pennsylvania State University (6-7). On the basis of chemical analyses made after each harvest, the product yields from the wastewater-irrigated plots may be expressed in terms of nutrient values removed by each product. Table (3) shows the removal efficiencies, defined as the ratio of the amount of nutrients removed out of the system by harvest to the total amount entering by the irrigating wastewater.

In the case of alfalfa, a 200 kg/hectare nitrogen utilization from air is assumed. The values obtained reflect the high sewage renovation capability of the present scheme. Removal efficiencies of above 100% obtained in some cases indicate that some nutrients initially present in the soil were also absorbed by the plants in addition to those introduced through wastewater application. It therefore appears that a greater rate of sewage application is possible without additional difficulties.

CONCLUSION

Results from research experiments on wastewater treatment through agricultural land irrigation illustrate that the proposed scheme can function as a pollution-free long-term sewage treatment/renovation technique, provided that correct management is exercised during the operation. While producing valuable food, the biosystem of soil and plant performs as a "living filter" and is capable of renovating the wastewater for groundwater recharge.

Although no risk of underground water pollution exists under the present operating conditions, the possibility of long-term soil pollution as a result of contamination with excessive amounts of nutrients may occur. This problem may be resolved by either using the plots alternately, or, for continuous operation, it may be controlled by matching the amount of wastewater applied to the type of vegetation cultivated such that the majority of the nutrients contained in the sewage are removed out of the system in the harvested biomass.

On the whole, the proposed scheme may be recommended as an effective method for wastewater re-use and disposal and appears to be specially suitable for semi-arid countries such as Iran.

Table (3) Removal efficiency of potential pollutants by various products.

		NITROGEN	PHOSPHORUS	POTASSIUM
ALFALFA	Present Experiments	82%	65.5%	195%
	Penn. State Project	-	30.3%	124%
CORN	Present Experiments	130%	61%	138%
	Penn. State Project	120%	123%	97%
POTATO	Present Experiments	124%	140%	180%
	Penn. State Project	-	-	-

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EVALUATION OF EXISTING SYSTEMS

GROUNDWATER RECHARGE WITH RECLAIMED WATERS FROM THE POMONA, SAN JOSE CREEK, AND WHITTIER NARROWS PLANTS

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ABSTRACT

The high demand for water in semi-arid and densely populated Southern California has seriously overdrawn groundwater resources. An active groundwater recharge program to replenish the groundwater system has been conducted by the Los Angeles County Flood Control District in the Montebello Forebay area of Los Angeles County. Prior to 1962 the water used for recharge had been exclusively storm water run-off and Colorado River water. Since that time, three wastewater *reclamation* facilities operated by the Sanitation Districts of Los Angeles County have provided reclaimed water for the groundwater recharge effort. The objective of this paper is to present a technical review of the various aspects of the Montebello Forebay groundwater recharge program, as well as a brief discussion of an ongoing health effects study which will determine whether this groundwater recharge program has caused any significant health risk within the population in the vicinity of the recharge operation.

INTRODUCTION

The historic overdraft of groundwater basins in Southern California has encouraged groundwater basin recharge in the South Coastal Hydrologic Region, as indicated in Figure 1, to maintain the groundwater table and to prevent the intrusion of seawater into the freshwater aquifers along the coast. For many years, the Los Angeles County Flood Control District (LACFCD) has been recharging the groundwater basins along the Rio Hondo and San Gabriel Rivers with storm water run-off and surplus Colorado River water. Since the early 1950's the Sanitation Districts of Los Angeles County (LACSD) have proposed the use of water reclaimed from sewage as a supplemental source of recharge water. The diminishing availability of Colorado River water and the continuously increasing overdraft of the basins combined to obtain general public support for the effort.

Beginning in 1962, reclaimed water from the Whittier Narrows Water Reclamation Plant (WNWRP) became available for spreading. The water was purchased by the Central and West Basin Water Replenishment District for the same price paid for Colorado River water under the terms of a contract which provided for the construction of the Whittier Narrows Plant. Based on the success of the Whittier Narrows Plant, the Sanitation Districts of Los Angeles County developed a master plan for construction of more water reclamation plants to meet the need for expanded water reuse. At this time, the Sanitation Districts operate three Water Reclamation Plants (Pomona, San Jose Creek, Whittier Narrows) which produce water for spreading in percolation basins operated by the Los Angeles County Flood Control District. The treated wastewater is disinfected by chlorination before discharge to flood control channels which convey the water to the spreading grounds. The present practice of recharge includes dilution of the reclaimed water with imported supplies on a 2:1 ratio prior to percolation. The dilution criterion is intended to produce groundwater with less than 10 mg/l nitrate nitrogen. The locations of the spreading grounds, the water reclamation plants and the flood control channels are indicated in Figure 2. The total quantity of reclaimed water recharged since 1962 is approximately 370 million cubic meters (300,000 acre-feet).

Groundwater recharge with reclaimed water via percolation is considered a cost-effective and esthetically attractive means of water reuse, because the unsaturated soil provides tertiary treatment for the reclaimed water and protection against the transmission of disease. Additionally, the underground aquifer provides both a storage and distribution system for the water and a method of losing the identity of the reclaimed water by blending with natural waters.

The Montebello groundwater recharge operation constitutes one of the oldest reuse projects in the nation. However, expansion of the recharge activity has been opposed by the California State Health Department. This is primarily due to the concern over the hypothesized relationship between low concentrations of stable organics reaching groundwater supplies and their effects upon human health. In an attempt to alleviate such concern, the Sanitation Districts have initiated a two year study on the health effects of groundwater recharge as part of the Orange and Los Angeles Counties Water Reuse Study.

SANITATION DISTRICTS WATER RECLAMATION PLANTS

The construction of the Whittier Narrows WRP was completed in July, 1962, and by August 20, 1962 the first reclaimed water was discharged into the Rio Hondo behind the Whittier Narrows Dam for delivery to the Rio Hondo spreading grounds. The plant was designed as a 38,000 cu m/day (10 MGD) activated-sludge treatment plant but has been operated successfully at a rate of 57,000 cu m/day (15 MGD). A dual media filtration system that will enhance disinfection will be in operation by the fall of 1978. The average water quality of the plant effluent is shown in Table 1.

The San Jose Creek WRP was constructed in 1971 and the first reclaimed water available for groundwater recharge was released to the San Gabriel River spreading grounds in 1972. Only portions of the 95,000 cu m/day (25 MGD) output by the San Jose Creek activated-sludge treatment plant were used for groundwater recharge in the past because of limits set by historical spreading proportions and also at times when total dissolved solids (TDS) levels exceeded 700 mg/l. At present, the plant contributes approximately 57,000 cu m/day (15 MGD) to the groundwater recharge effort. The average water quality of the plant effluent is shown in Table 1. Installation of an inert dual media filtration system at the San Jose Creek WRP is presently nearing completion and is expected to be on-line by the fall of 1978.

Approximately 15 percent of the 38,000 cu m/day (10 MGD) reclaimed water from the Pomona WRP is presently reused in the Pomona area. The remaining flow is discharged to the lined section of San Jose Creek in Pomona and conveyed to unlined portions of San Jose Creek and the San Gabriel River where it infiltrates near the Whittier Narrows Dam. The average quality of the effluent is shown in Table 1. Since January, 1977 the Pomona WRP has employed an activated granular carbon adsorption-filtration system to further improve its activated-sludge secondary effluent before discharge into San Jose Creek. A substantial portion of this carbon treated effluent is intended for reuse by various local industries.

There are no sludge-processing facilities at the three water reclamation plants described above. All sludges from the plants are returned to trunk sewers, and conveyed to the Sanitation Districts' Joint Water Pollution Control Plant (JWPCP) in Carson, California, where they are removed, anaerobically digested, dewatered, and composted. As indicated by the typical domestic sewage treatment plant effluent qualities in Table 1, each of the three inland treatment plants receives negligible quantities of industrial wastewater. Thus, metal concentrations in the final effluents are very low and are not included in Table 1.

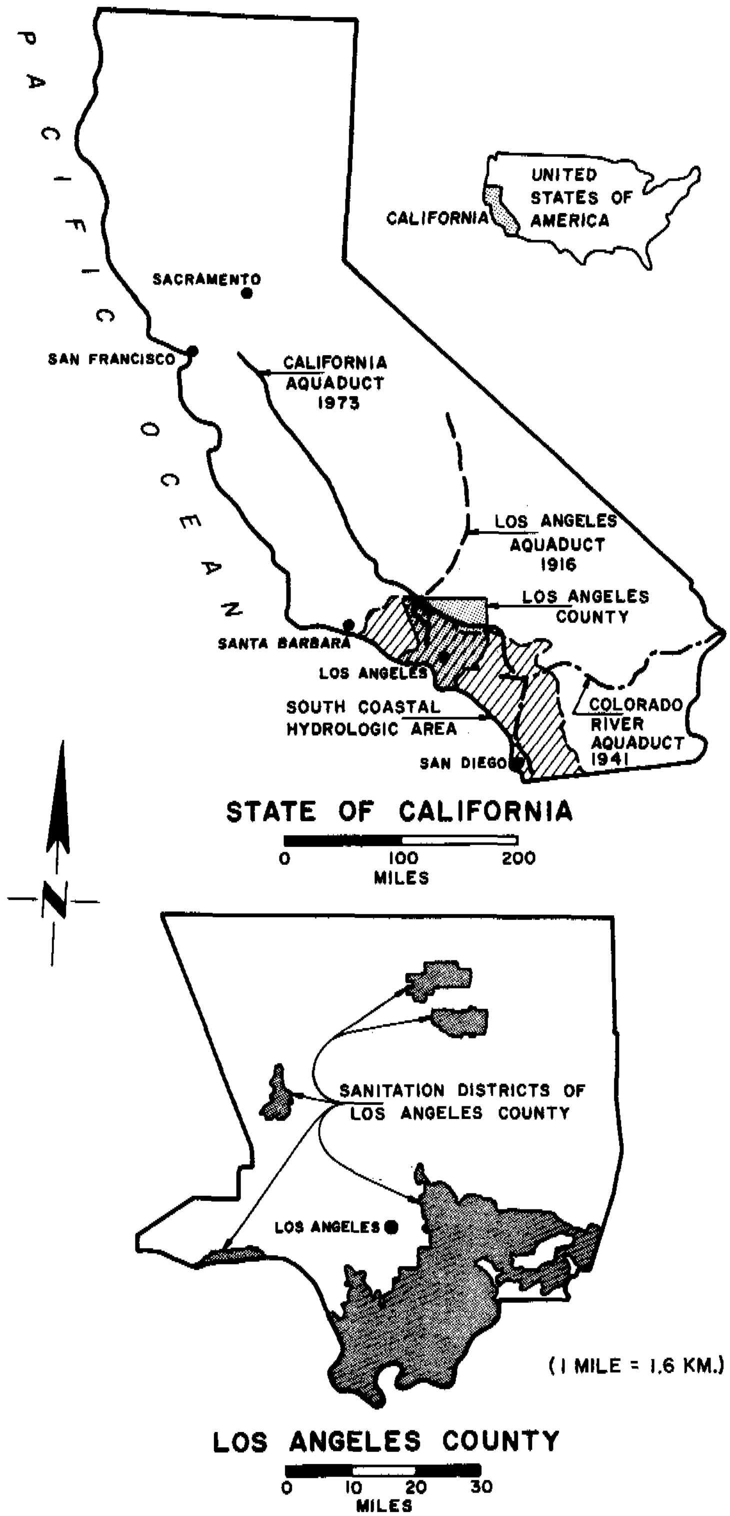

FIGURE 1. LOCATION MAP

Table 1. Summary of Secondary Effluent Quality (1976 Annual Average

Constituents	Whittier Narrows	San Jose Creek	Pomona
Suspended Solids, mg/l	7	5	6
Total COD, mg/l	37	31	43
Dissolved COD, mg/l	28	25	35
Total Nitrogen, mg/l N	19.1	14.0	18.5
Total Phosphate, mg/l PO_4	18.9	18.8	22.8
TDS, mg/l	511	676	559
Total Alkalinity, mg/l $CaCO_3$	219	208	215
Total Hardness, mg/l $CaCO_3$	179	231	212

GROUNDWATER RECHARGE OPERATIONS

Spreading Grounds

The Rio Hondo - San Gabriel River system drains the entire San Gabriel Valley through a 3.2 kilometer (2 mile) break in the topography called the Whittier Narrows. The Whittier Narrows separates the La Merced Hills from the Puente Hills. To the west, the La Merced Hills rise to over 182 meters (600 feet) above sea level, while on the east, the Puente Hills rise more sharply to about 424 meters (1400 feet) above sea level. The ground surface elevation at the Whittier Narrows is about 60 meters (200 feet) above sea level.

A broad floodplain has been formed to the south where the Rio Hondo and San Gabriel Rivers emerge from the Whittier Narrows. This is known as the Montebello Forebay. It forms the connection between the groundwater in the San Gabriel Valley and the two large groundwater basins to the south; namely, the Central Basin and the West Coast Basin. Located in the Montebello Forebay are the Rio Hondo and San Gabriel Spreading Grounds which are operated by the Los Angeles County Flood Control District.

The Rio Hondo and San Gabriel River Spreading Grounds, indicated in Figure 2, have total available percolation areas of 182 hectares (455 acres) and 40.4 hectares (101 acres), respectively. Approximately 53.2 hectares (133 acres) are also available for spreading in the unlined portion of the San Gabriel River. The spreading grounds are subdivided into numerous individual basins ranging in size from 1.6 to 8 hectares (4 to 20 acres).

According to California Department of Water Resources Bulletin No. 104,[1] there is continuity between the ground surface and the uppermost aquifers along the Rio Hondo and the San Gabriel River and between these two streams from the Whittier Narrows downstream to the vicinity of the Santa Ana Freeway. In general, there is a significant degree of hydraulic continuity over much of the Montebello Forebay area between the uppermost Gaspur Aquifer and the successive underlying Exposition-Artesia Aquifer, Gardena-Gage Aquifer, Hollydale Aquifer, Jefferson Aquifer, and Lynwood Aquifer. However, the areas of hydraulic continuity between the foregoing aquifers and the deeper Silverado and Sunnyside Aquifers is, to a large degree, limited to relatively small areas near the Whittier Narrows (and areas above the Whittier Narrows).

Inflows to the Montebello Forebay

Until reclaimed waters were available, groundwater recharge by spreading or natural infiltration, utilized Colorado River water, subsurface inflow, rising water, storm inflow, and local precipitation. In 1974, additional imported water from the California Aqueduct became available for groundwater recharge. Table 2 shows the total inflows of the various sources of recharge waters for each water year from 1960-61 to 1976-77. During this entire period, reclaimed water has represented 13.5 percent of the total inflows to the Montebello Forebay while imported and local waters have represented approximately 41.1 and 45.4 percent, respectively.

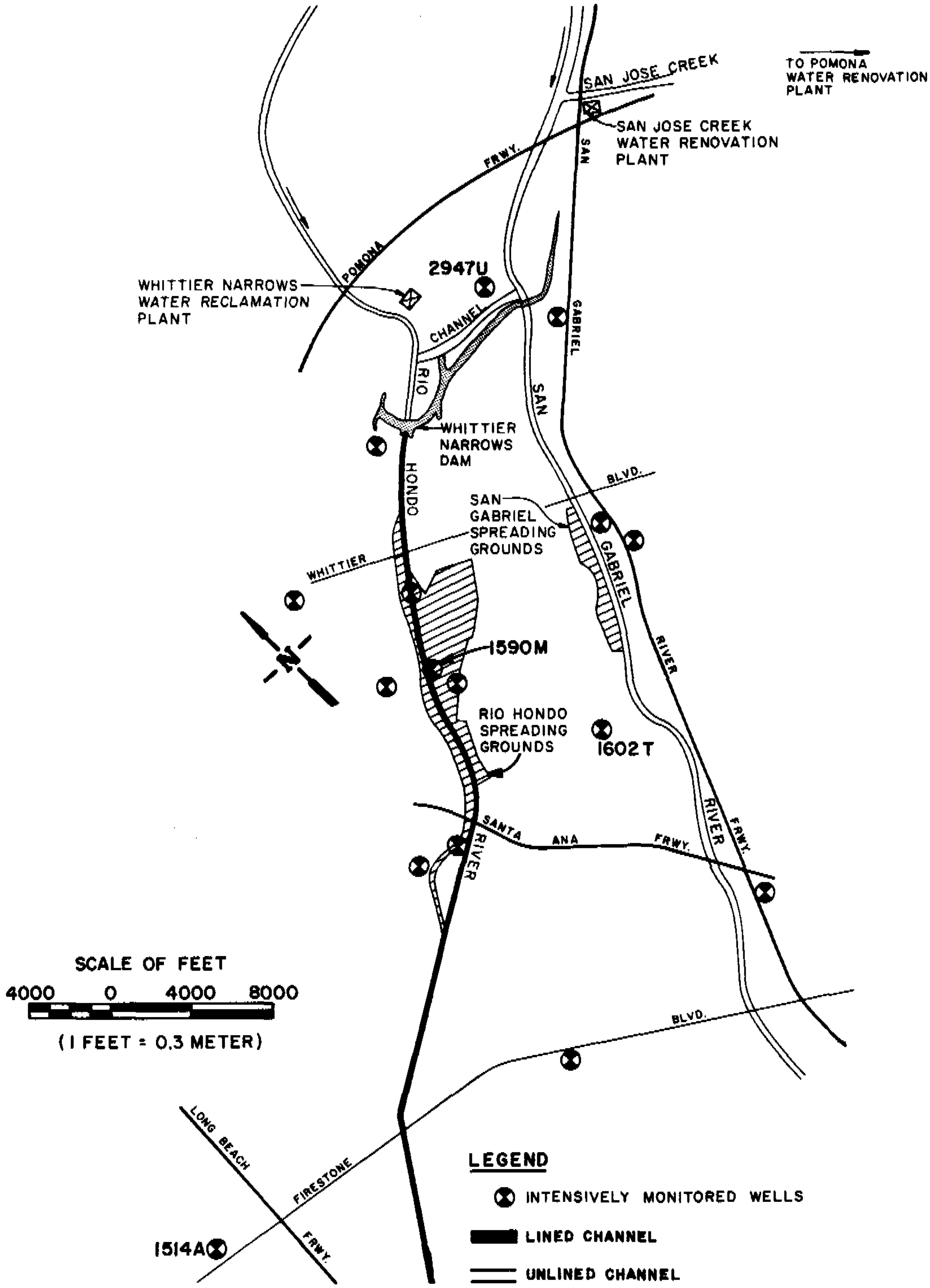

FIGURE 2. LOCATIONS OF SPREADING GROUNDS, CHANNELS AND TREATMENT PLANTS

Operational Procedures

Many of the operational procedures used in groundwater recharge are based on experience with artificial filters, i.e., slow sand filters for water treatment and intermittent sand filters for sewage treatment. Normally, artificial filters have some type of effluent collection system, whereas a surface spreading operation normally permits the percolated water to reach the main body of groundwater.

The Rio Hondo and San Gabriel River spreading basins are filled cyclicly to maintain aerobic conditions in the upper soil strata and to control vector insects. Groups of basins are rotated through a 21 day cycle consisting of filling to a depth of 1.2 meters (4 feet) for 7 days, draining for 7 days, and drying for 7 days. On this rotating basis the capacity of the basins for recharge is approximately 757,000 cu m/day (200 MGD). On a short term basis with all basins in operation simultaneously the capacity for groundwater recharge is approximately 2,271,000 cu m/day (600 MGD). This latter mode of operation is used in storm periods to maximize conservation of storm water.

GROUNDWATER MONITORING PROGRAM

There are approximately 200 wells in the Central and West Coast Basins owned and operated by numerous water companies and monitored by both the California State Department of Water Resources and the Los Angeles County Flood Control District under their Basinwide Monitoring Program. Some of the wells produced water quality data as early as 1931; however, the majority of the wells were not monitored for water quality until the 1950's. Most of the wells are sampled annually and analyzed for calcium, magnesium, sodium, potassium, bicarbonate, carbonate, sulfate, chloride, nitrate, TDS, iron, manganese, and trace metals.

Beginning in 1972, sixteen wells within the Montebello Forebay area were specifically selected for an intensive water quality monitoring program. The general locations of these intensively monitored wells are indicated in Figure 2. Samples from these wells are collected by the Los Angeles County Flood Control District and analyzed by Sanitation Districts' Laboratory and other commercial laboratories at Central and West Basin Water Replenishment District's cost. Intensively monitored wells are sampled quarterly for major minerals, nitrogenous compounds, COD, BOD, TDS, electrical conductivity, pH, and odor, and annually for complete trace metals and chlorinated hydrocarbons analyses.

Data from the above monitoring programs have been used actively to evaluate the impact of groundwater recharge on the well water qualities. These data, as well as the new data to be collected by the Health Effects Study, will aid in the determination of possible expansion or curtailment of groundwater recharge programs.

CHANGES IN RECLAIMED WATER QUALITY DURING PERCOLATION

The water quality changes that occur in reclaimed water as it percolates through soil have been studied by several investigators at the Whittier Narrows Test Basin. The test basin was constructed by the Los Angeles County Flood Control District in the reservoir area north of the Whittier Narrows Dam. Prior to the installation of the test basin, the land was used for farming. Consequently, there was an abundance of organic material in the uppermost portion of soil. The soil profile at the Whittier Narrows site presented in Table 3 shows the soil to be non-homogenous with thin, discontinuous layers of silt and micaceous material. At the time of construction, the water table was approximately 2.7 meters (9 feet) below the ground surface.

During the various studies, undiluted reclaimed water from the Whittier Narrows WRP was spread at rates varied to maximize the hydraulic loading while maintaining aerobic conditions in the soil. Grab samples were collected from sampling pans buried at depths of 0.6, 1.2, 1.8, and 2.4 meters (2, 4, 6, and 8 feet, respectively) below the ground surface. A brief discussion of each individual series of experiments follows.

McKee-McMichaels Study[2]

During the McKee-McMichaels study, the spreading schedule initially consisted of continuous flooding which was subsequently changed to flooding on Mondays, Wednesdays, and Fridays. The

Table 2. Annual Inflows to the Montebello Forebay
(Thousands of Acre-Feet)

	Reclaimed Water			Imported Water		Local Water			
Water Year	Whittier Narrows Plant	San Jose Plant	Pomona Plant	Colorado River Water	Calif. Aqua-duct	Subsur-face Inflow	Rising Water	Storm Inflow	Local Precipi-tation
1960-61				121		26	6	5	11
1961-62	1			180		23	11	34	23
1962-63	12			68		21	7	9	12
1963-64	13			86		21	3	11	11
1964-65	15			136		19	2	15	13
1965-66	15			102		17	6	41	20
1966-67	16			67		17	14	47	22
1967-68	18		6	79		21	17	16	12
1968-69	14		6	17		24	26	88	26
1969-70	17		9	61		24	23	21	11
1970-71	19		8	63		23	14	22	12
1971-72	18		8	30		24	6	18	11
1972-73	14	8	8	80		26	8	30	22
1973-74	13	8	6	80		27	7	26	14
1974-75	15	7	5	11	56	26	4	18	13
1975-76	12	9	5	1	34	28	1	18	5
1976-77	10	13	1	12	14	32	0	19	11

NOTE: 1000 acre-feet = 1.23 million cubic meters.

final schedule consisted of flooding from Monday through Friday.

The results of the study indicated that the sum of organic and ammonia nitrogen was diminished by more than 54 percent during percolation through the upper 60 centimeters (2 feet) of soil. Past this depth, the change of the organic and ammonia nitrogen concentration was minimized. Continuous increase of nitrate nitrogen was observed through the 2.4 meter (8 feet) sampling depth. The nitrate nitrogen concentration at the 2.4 meter sampling point was about five times higher than the surface concentration. This increase was slightly less for samples collected during periods of continuous flooding. Total nitrogen decreased slightly (about 10 percent) over the first 1.8 meters (6 feet) of percolation, but increased (about 18 percent) between the 1.8 and 2.4 meter (6 and 8 feet) depths.

Approximately 53 percent of the applied COD was removed by the first 1.2 meters (4 feet) of soil. From 1.2 to 1.8 meters (4 to 6 feet), COD concentrations were observed to increase to approximately 8 percent over the surface concentration. Below this level, the COD concentrations decreased to slightly below the surface concentration. The increase in COD observed in some of the 1.8 meter (6 foot) percolates probably resulted from contamination in the 1.8 meter (6 foot) pan.

The study results indicated an increasing trend for both TDS and hardness with depth. TDS increased by 18 percent while hardness increased by 53 percent. Both increases were attributed to dissolution of minerals by weak nitrous acid formed during the intermediate steps of ammonia to nitrate conversion in the soil.

Contrary to the findings of previous investigators that vertical travel through soil removed coliform organisms, this study demonstrated that well-ripened filtration beds generate coliforms as well as myriads of other bacteria. Further tests revealed that coliforms in the percolates were of a non-fecal origin, i.e., they were normal

Table 3. Geologic Soil Profile at the Whittier Narrows Test Basin(2)

Depth Below Surface (cm)	General Description
0-	Dark brown very fine to medium silty sand and soil
60-	Light brown to tan fine to medium sand with lenses of gray fine sand. Moist, oxidized, orange fine sand streaks are common in tan portion.
120-	Wood fragments up to 7.6 cm long in dark brown to black medium to fine sand. Sand is highly micaceous.
180-	Tan fine to medium soft, micaceous sand, with gray fine sand lenses. Tan portions commonly show orange streaks of oxidized fine sand.
240-	Dark brown to black micaceous fine sandy silt stringer.
300-	Gray medium to coarse sand and "pea gravel" with occasional gravels to 1 cm.

soil bacteria that thrived in the environment created by the application of reclaimed water.

Virological analyses performed on the reclaimed water indicated that on the average, enteric virus concentrations were very low (less than 100 PFU per liter). Samples taken during and shortly after the Sabin Oral Sunday inoculations of February, 1963 showed about 250 PFU of enteric viruses per liter in the Whittier Narrows effluent but no measurable concentration in the percolate from the 60-cm (2-foot) sampling pan. A subsequent addition of Sabin Type - III vaccine to the effluent applied to the testing basin failed to produce measurable concentrations of viruses in the percolates.

1970 LACFCD Study(3)

The Los Angeles County Flood Control District initiated a study in June, 1970 to evaluate the effects of long-term spreading and to further investigate changes in reclaimed water quality during percolation. During the study, the test basin was flooded for seven consecutive days and then allowed to dry at least 14 consecutive days. During this investigation, measurements of COD, coliforms and viruses were not made.

The results indicated that the ammonia concentration at the 1.2 meter (4-foot) level was essentially zero, while the nitrate concentration had increased by a factor of eight from the surface concentration. One unexplained phenomenon occurred at the 1.8 meter (6-foot) level where percolate samples showed a sharp increase in ammonia and a decrease in nitrate. Following 2.4 meters (8 feet) of percolation, the ammonia concentration was essentially zero and the nitrate concentration returned to its initial concentration.

As in the McKee-McMichaels study, TDS and hardness both increased with depth. In this study, TDS increased by approximately 11 percent and hardness by 31 percent following 2.4 meters (8 feet) of percolation.

LACFCD - LACSD Study(3)

The Los Angeles County Flood Control District and the Los Angeles County Sanitation Districts began a joint investigation in March, 1973 of the effects of spreading nitrified effluent. Spreading and sampling methods were similar to those used in the 1970 LACFCD study. The main thrust of this study was to evaluate the interrelationship of nitrification and TDS increase. Each agency collected data for a period of approximately one year.

LACSD data indicated 51 percent of applied COD was removed during 2.4 meters (8 feet) of percolation. This removal closely approximates the removal found by the LACFCD of 57 percent. In both cases, the greatest change occurred within the first 1.2 meters (4 feet) of percolation. No further nitrification was observed to occur as a result of percolation since nitrate and ammonia concentrations remained constant with depth.

The LACSD data revealed that the TDS increased about 19 percent over surface concentrations. This increase was approximately 30 percent for LACFCD data. These results combined with data from the two previous studies which used non-nitrified effluent demonstrate that some differences did exist for nitrified and non-nitrified percolated effluents in terms of TDS increase; however, these differences were not too significant. Hardness also increased with depth. The average increase, according to LACSD data, was 52 percent which is about the increase found by the LACFCD of 63 percent.

Summary of Test Basin Studies

Combining the data from all Whittier Narrows Test Basin studies, the following conclusions can be made regarding the changes in reclaimed water quality which occurred during 2.4 meters (8 feet) of percolation:

1. Approximately 50 percent COD removal was achieved. This average did not include data from the McKee-McMichaels Study because that data did not fit the pattern of more recent studies.
2. TDS and hardness were increased approximately 20 and 50 percent, respectively.
3. Nitrate and ammonia nitrogen behavior varied from study to study and also varied between effluent type. However, both nitrification and denitrification seemed to occur during percolation.
4. On the basis of one study, both viruses and fecal coliforms were shown to be effectively removed.

Although the studies have contributed to the understanding of changes in reclaimed water quality during percolation, the fate of many constituents, such as heavy metals and trace organics, is still unknown. The capacity of the spreading grounds to continue to remove biological and chemical contaminants should also be investigated. Additional test basin work has been planned as part of the Health Effects Study to address some of these questions.

EFFECTS OF RECLAIMED WATER ON GROUNDWATER QUALITY

The water quality data from monitoring wells indicate that the various well waters are generally of good quality. Outside of the fairly high mineralization of some of the well water supplies, no significant water quality problems have been revealed by the groundwater monitoring program.

Well No. 2947U is located slightly upstream from the Montebello Forebay spreading grounds, as indicated in Figure 2, and contains 300 mg/l or less TDS which is a measure of the total mineralization of the water. Well No. 1590M which is located within the Rio Hondo spreading ground is shown to contain higher TDS in the range of 340 to 400 mg/l. The difference in TDS concentrations of the above two wells can be attributed to the groundwater recharge operation. Although the reclaimed water may contribute some TDS to the groundwater as revealed by the results of the Whittier Narrows test basin studies, yet the TDS increases in the wells influenced by the spreading operations are predominantly due to the spreading of Colorado River water which contains an average TDS of 750 mg/l and has comprised over 40 percent of the total amount of water spread. A similar increase pattern is also observed in the results of well water hardness analyses. The hardness increases in the wells are also primarily due to the spreading of the Colorado River water which has an average hardness of 330 mg/l as $CaCO_3$.

The average nitrate and COD concentrations of the sixteen intensively monitored wells are shown in Table 4 under a special grouping. The grouping of the wells is based on the distance from spreading grounds and the depth to the first well perforation. This categorization is crude because complexity of the aquifers has not been considered. Nevertheless, the table seems to indicate a trend between nitrate concentration in the well and the distance of the well from the spreading grounds. In spite of this concentration and distance relationship, the slight increase in nitrate concentration can probably be attributed to sources other than reclaimed water, such as fertilizer from previous farming operation in the Montebello Forebay.

The average nitrate concentration in the affected wells is still well within the 10 mg/l nitrate nitrogen limit. Any concern for nitrate increases can be minimized by maintaining a proper ratio between the reclaimed water and the imported surface water in the spreading operation. The same practice can be applied to control the increase of TDS and hardness by using more California Aqueduct water, which has average TDS and hardness concentrations of 190 mg/l and 90 mg/l (as $CaCO_3$), respectively, instead of Colorado River water in the blending operation for groundwater recharge.

As indicated in Table 4, the COD concentrations in the wells seem to be independent of the distances between the wells and the spreading grounds. The average COD concentrations in the wells are further shown to be generally lower than those of local surface water supplies. Therefore, no serious impact of reclaimed water on the groundwater quality can be recognized. However, the effects of reclaimed water on groundwater quality cannot be fully evaluated without more complete data on the fate of trace metals, trace organics, and microorganisms. Therefore, an accurate evaluation cannot be made until the completion of the Health Effects Study.

HEALTH EFFECTS STUDY

A study has been initiated by the Sanitation Districts to determine the health effects of the groundwater recharge program in the Montebello Forebay. The study is carefully devised to produce the needed data to allow regulatory agencies to make decisions on whether to continue and possibly expand the Montebello Forebay recharge effort, and, in addition to provide information upon which other recharge programs could be established. Although closely interrelated, the study has been divided into the following four tasks:

Assessment of Reclaimed Water Movement

This task is to quantify as accurately as possible the percentage of the well water that consists of water that was once reclaimed water. This information is needed to determine the exposure of population in the vicinity of the recharge operation to water of reclaimed water origin. The most promising approach to this problem involves the use of a previously developed water movement model for the Montebello Forebay based on sulfate ion concentrations in Colorado River water acting as a tracer.

Additionally, a new test basin study will be conducted under this task to determine the changes in reclaimed water quality during percolation. Factors such as soil properties, depth to groundwater, and spreading schedule, which are believed to influence the quality changes, will be thoroughly investigated.

Characterization of Water Quality

This task will produce detailed water quality data for well waters, reclaimed waters, imported waters, and other sources of waters involved in the recharge effort. Water quality analyses will include existing analytical parameters under the intensive monitoring program; however, more emphasis will be placed on the characterization of trace organics, trace metals, and viruses. Initially, a list of specifically selected organic compounds of suspected health significance will be used as a set of target compounds. Samples will be analyzed using gas chromatograph/mass spectrometry (GC/MS). All GC/MS computer output will be recorded on magnetic tape so that additional compounds not on the list can be identified without the burden of additional sampling.

Table 4. Summary of NO_3^- and COD in Intensively Monitored Wells

Group*	No. of Wells	Distance from Spreading Grounds (kilometer)	Depth to First Perforation (meter)	Avg. COD (mg/1)	Avg. NO_3^- (mg/1 N)
I	7	0-1.3	12- 90	3.1	3.2
II	5	1.3-6.1	90-150	3.5	1.7
III	4	1.6-3.2	44-159	3.1	0.2

* Group I = Wells close to the spreading grounds and with shallow perforations.
Group II = Wells moderately close to the spreading grounds with deeper perforations.
Group III = Wells upstream from spreading grounds.

Toxicologic Experiments

The Ames test and a mammalian cell transformation test will be used to isolate the discrete organic fractions of water concentrates which exhibit possible health effects, i.e., mutagenic and presumptive carcinogenic activity. Specific compounds or fractions with positive test results will be further identified by GC/MS/COMP methods.

Epidemiological Study

The objective of this task is to determine whether the groundwater recharge program in the Montebello Forebay causes any increase in health risk compared to other areas where potable well water has not been influenced by planned water reuse. The success of this task is heavily dependent on the fulfillment of the goals of the above three tasks.

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EVALUATION OF EXISTING SYSTEMS

THE SEWAGE FARMS OF PARIS

Robert B Dean LunDean Environmental Co

Abstract

The sewage treatment farms of Paris, France, have been in operation since 1880. Some early fields close to the City were abandoned in 1955 and that land is now used by industry. The present farms some 30 km NW of Paris were started in 1895 and now occupy essentially the same areas they did in 1920. Sewage in excess of the capacity of the fields is treated by activated sludge. The farms now treat only 15% of the total flow and 30% of the solids. They will be retained as a safety feature for so long as the land is not expropriated by local communities. Raw sewage, screened and degritted, is applied at rates up to 4 meters per year. The major crop is maise used as dry cattle feed. Grass, trees and vegetables are also grown, but with one minor exception, no vegetables that may be eaten raw. The Ministry of Health inspects vegetables sold in the Paris markets and has found no evidence of infectious disease germs in produce from the sewage farms. Irrigation of vegetables is stopped if the World Health Organization announces a cholera alert for Europe and vegetables are then quarantined for 10 days after the last irrigation. There is no evidence of damage to the soil by heavy metals. The authorities do not consider metal uptake in foods to be a problem and analyses of foods for heavy metal content have not been made. Underdrain effluent is usually better than the standards for secondary treatment but contains nitrates in excess of WHO standards for drinking water, at least in the warm months.

The utilization and treatment of sewage on land by the City of Paris, France, started in 1880 following years of discussion by, among others, Victor Hugo who as early as 1862 in "Les Misérables" looked upon night soil and sewage as "flowering fields .. green grass .. game and cattle". Land in the first meander of the Seine river north of the City in the region of Grennevilliers was used for sewage purification until the mid 50's. This may be the area referred to by Rhode as showing signs of exhaustion (1962). He attributed the exhaustion to excess zinc in the soil and found zinc values ranging from 500 to 2200 ppm for unhealthy soils and 305 to 455 ppm for healthy soils. The lowest pH was 6.7. There have been several unsuccessful attempts since then to find the exhausted soils of the Paris sewage farms (Bauer 1973). The present management of the system has no knowledge of any problems with fertility. In any case the old sewage farm at Grennevilliers was closed in 1955 and converted to industrial use. The soil analyzed by Rhode is probably covered by the General Motors now.

The first sewage farm at Grennevilliers was already too small by 1895 and additional land was purchased in meanders further down stream from the city. By 1920 large trunk sewers had been built to Achères, and sewage was irrigated on the land from permanent concrete head boxes spaced 20 to 60 meters (yards) apart along 30 cm (1 foot) mains. Figure 1. Sewage was and still is distributed in conventional irrigation furrows. Much of the land drains naturally through porous subsoil to the river. There are some tile drains and open drain-

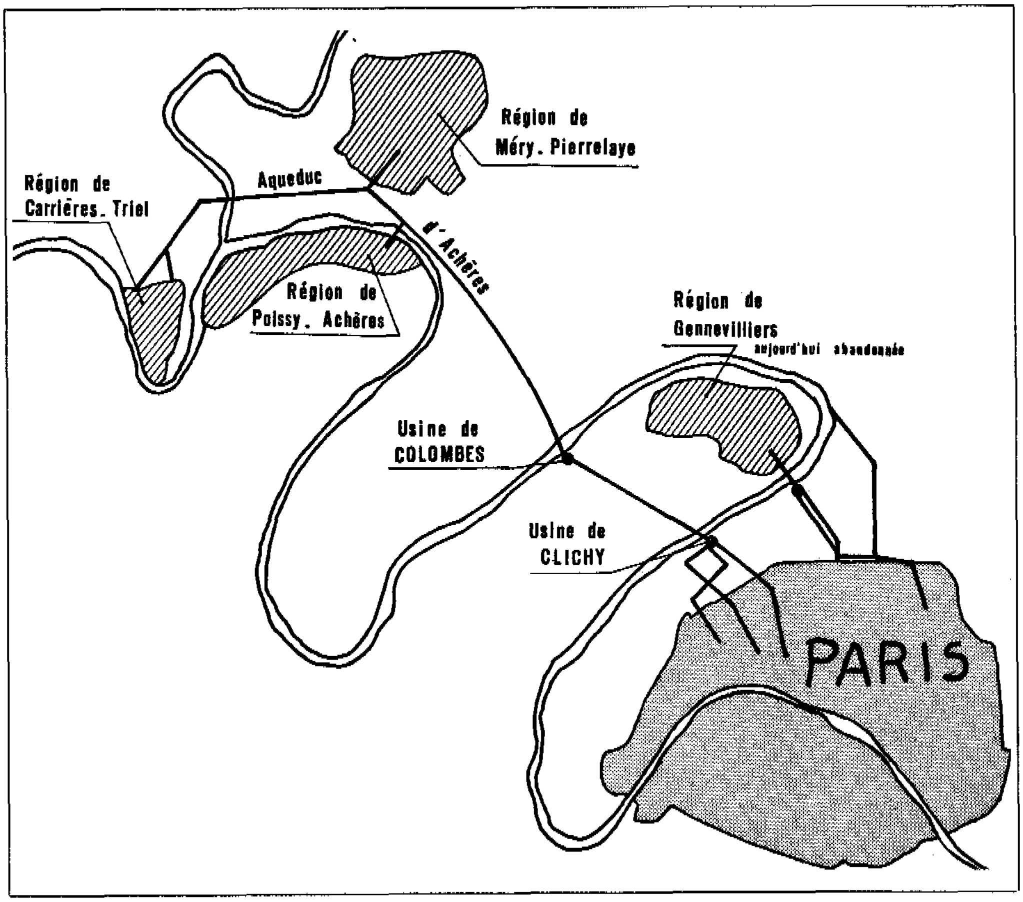

Figure 1 Map of the Sewage Farms of Paris and Trunk sewers.

age ditches 2 to 3 meters (7-10 feet) deep to convey surplus water to the river. A small recreational lake is also filled by seepage from the sewage fields.

Almost all of the land in the Archere district that was included in the sewage farms in 1920 is still receiving sewage but some of the land has been taken over for sewage treatment plants to handle the increased volume due to the population growth since 1929. At the present time the sewage farms take 70 to 80 million cubic meters per year (about 20 billion US gallons per year) out of a total annual flow of 520 million cubic meters (150 billion gallons) or less than 15% of the flow. The rest of the sewage is treated by primary sedimentation followed by activated sludge. Waste activated sludge and all other excess liquids are returned to the primary tanks. The combined sludge is digested in two-stage high-rate digesters operated at 35°C (95°F). About 15% of the digested sludge is pumped back into the sewer main serving the irrigated farms more than doubling the solids content of the raw sewage going on the land. Sewage going to the farms receives only degritting before being pumped from the main collection station in Clichy on the edge of Paris proper. The management of the farms considers that primary sedimentation or any other form of sewage treatment would be a waste of good organic matter that might otherwise benefit the soil.

Typical compositions of sewage as it was applied to the land in 1977 is shown in Table 1.

Table 1.

	Solids	Organic	Total N	Total P
Normal Sewage	13,000 mg/l	60% of Solids	300 mg/l	100 mg/l
Heavy Sewage	42,000	40	1680	

It is obvious that this fortified sewage could equally well be described as a dilute sludge. The metal content of digested sludges taken from drying beds in the same year was below the median concentration of US sludges reported by Salotto et al (1974) for all elements with the exception of zinc and cadmium.

The soil of the farm is a mixture of sand, clay and gravel that was laid down by the river as it meandered back and forth across the valley. The underlying rock is limestone and the soil is full of flints and is naturally well limed. Sewage is applied at rates which by regulation must not exceed a total of 4 meters per year (3 inches per week). The highest average rate was 3.6 meters in the drought of 1976. Some fields appear to have received more than their share of sewage. Turbidity in the effluent in one drainage ditch was attributed to heavy loading of a few fields to get rid of some sludge. There are two types of farms in the sewage disposal system. About half of the area is leased to independent farmers who operate their land subject to regulations by the City and the Department of Health. The average farm covers 150-200 hectare (375-500 acres) and must have one sixth of the area in pasture land that can take sewage at any time of the year. Otherwise the leased farms take sewage only when they want it, usually during the dry summer months. The city operated farms take sewage all year and are happy to have the warm sewage when frost or snow covers the rest of the land. The city is taking over the leases of the tenant farmers as their contracts expire so that it can use the land more efficiently for sewage treatment. About 400 of the 1300 hectares operated by the city is used to grow poplar and other trees for the city parks. Otherwise the principal crop is maise or indian corn which is used as grain for cattle. Hay is not cut from the meadows and only one farm is raising beef cattle. The only vegetables permitted are those that will be cooked such as beans, onions and celery root (celeriac). Celery root however is also eaten raw in France, usually grated as a garnish on salads and meats. One farm at Achères ships truckloads of hand peeled celery roots to the Paris Markets. Spraying of sewage is not permitted in France and no sewage may be applied to the leaves of growing crops.

Vegetables from Archères are carefully checked for Salmonella by the Hygiène Laboratory of the City of Paris. None have been found and no disease has been traced to the sewage farms. However recommendations from the Health Dept have stopped the raising of dairy cattle on the sewage farms. In the event that a cholera outbreak is officially announced by the World Health Organization all vegetables would be quarantined for 10 days after the last application of raw sewage. Cholera vibrio have not been identified on vegetables from farms during a cholera outbreak. Ascaris (round worm) eggs have sometimes been found but it is not known whether they came from humans or dogs. Odors are noticed from time to time but they are not significant outside the boundry of the farms.

All sludge separated from sewage is digested. Some is dried on open air beds and shipped to vineyards. Some is given heat treatment similar to Porteous treatment. The heat treated sludge is filter pressed to 50% solids and shipped 50 km (30 miles) to a farmers association that is buying the sludge on a 5 year contract. Some vacuum filtered sludge is dumped in a landfill. The rest is added to the raw sewage going to the land.

Although France has regulations regarding the quality of sludge used in agriculture no city seems able to meet them and they all get specific variance. The maximum solids loading is currently set at 25 tons/ha over a 5 year period. Sludge in the sewage shown in table 1 would be at least 130 tons per ha for each meter of sewage applied. The Ministry of Culture and the Environment have recently started a research program on "Soil and Solid Refuse". The first symposium, which was held in March 1977 at Orléans, dealt with the early stages of research into the properties of sludges and their behavior on the land (Colin 1978). The early studies failed to find Salmonella on sludges used at Archères although this negative finding does not mean that the bacteria were absent. It is known that it is very difficult to culture Salmonella in the presence of activated sludge.

The entire operation at Archères follows the general requirements given by Abel Wolman at the 50th Annual WPCF Conference last year with some minor exceptions. It is carefully managed on an appropriate site, potential hygienic risks are detected and controlled and the process is cost effective. There is a prohibition against crops eaten raw, but there is one notable exception in the case of celery root. Monitoring of ground water has shown only nitrates in excess of drinking water standards and all ground water flows to a large polluted river. The operation does not however stop during rain or freezing weather and there is no provision for storage. Furthermore solids are not removed from the raw sewage so that in effect raw sludge is being applied to the land along with additional digested sludge. Metal hazards to human health are indirectly controlled by the prohibition on leaf vegetables which might be eaten raw. The major crop consists of grains which take up much less metal than leaf vegetables. The grain is furthermore fed primarily to short lived meat animals where the risk from cadmium accumulation in the kidney is minimal. There is little doubt that sewage treatment on the land is successful in Paris. One million people more or less (15% of 6.4 million) have been using the land for fifty years and, as has been said before, "50 million Frenchmen can't be wrong".

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EVALUATION OF EXISTING SYSTEMS

LAND TREATMENT SYSTEM IN POLAND

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ABSTRACT

Agricultural utilization of sewage has a long tradition in many European countries, extending over several dozens of years. Also in Poland there are several effectively operating systems using municipal sewage in agriculture. It is required that the already operating sewage land treatment plants and those which are now under construction provide full biological treatment of sewage through its most economical utilization in production of crops. It is also required that sewage is utilized under proper sanitary and hygienic conditions and with protection of environment.

This article presents results of observations and experiments concerning the operation of Wroclaw Sewage Farm, which was operated continuously for 90 years. The positive effects of land treatment of sewage, observed at this farm, are characteristic for many such systems operating in Poland.

INTRODUCTION

The present situation in agriculture, which suffers from a deficit of fertilizers and the increased activity for conservation of aquatic environment are bringing about a world-wide interest in agricultural use of sewage, particularly of municipal sewage, because of its large fertilizing potential (4).

The fertilizing value of sewage is beyond question. Apart from its irrigating value, sewage contains dissolved organic compounds, suspensions and biogenic substances of particular value in agriculture. The fertilizing value of sewage is not limited to those nutrient macrocomponents. Sewage, in addition to organic matter and macrocomponents, contains also a full blend of microelements as well as several microorganisms and enzymes.

The second essential argument, fully justifying the purposefulness of sewage use for cropland irrigation, is provided by the volume of sewage. For example, in Poland during dry years sewage represents over one half of all surface water flow all over the country. During such dry seasons sewage constitutes the most reliable source of irrigation water, particularly because there is also a scarcity of water reservoirs. Sewage is available the year round, regardless of hydrological conditions and weather (3,4).

Irrigation of croplands with sewage is economically important because unlike irrigation with pure water, it decreases the demand for mineral fertilizers such as NPK.

The above considerations indicate that one of the main purposes of installation of irrigated fields is maximum utilization of sewage in production of crops. Many data, collected by experiments conducted over many years, demonstrate that sewage is utilized most economically by grasslands and fodder root crops. For example, the long-term experimental cycles at the Institute of Melioration and Grasslands demonstrated the indisputable advantages of fodder plants. Sewage utilization by cereal plants is much less efficient.

The yields, usefulness and quality of crops grown on irrigated fields depend on local soil and water conditions, technology and technique of irrigation, methods of agricultural utilization as well as on the type and quality of sewage. All these factors are closely interconnected.

In addition to positive effects, the land systems of sewage treatment or utilization have also some negative aspects. Municipal sewage contains large quantities of chemical substances and various microorganisms, even pathogenic ones, being potentially hazardous to the health of animals and, indirectly, also to human health. Both these groups of components may produce a durable or temporary contamination of plants, ground waters or even air when sewage is incorrectly managed.

As a result of the awareness of these hazards, there were organized, in several scientific institutions in Wroclaw broad, multi-sided studies. Because of the nature of the problem the studies were collective and involved such areas as medicine, veterinary medicine, hygiene, sanitary engineering, water management and agriculture. The results of these demonstrated that it is possible to use sewage effectively in plant production and, at the same time, to achieve safe and effective treatment of sewage by its purification in irrigated soils (4). Attention was brought to the fact that there exists an effective possibility of elimination of biogenic sewage components through their assimilation by irrigated plants.

This article presents some results of observations and experiments carried out at Wroclaw Sewage Farm. This farm is among the largest in Poland, where the requirements of agriculture were successfully combined with correct technology of municipal sewage treatment.

GENERAL CHARACTERISTIC OF WROCLAW IRRIGATION FARM

Wroclaw in its central part has a combined sewage system while in other parts there is a separate system. Municipal sewage is treated mainly at the irrigation farm which accepts about 93% (about 170,000m^3/day in 1975) of total sewage. Irrigated fields were designed and built about 1890. The fields were extended as the city grew larger and now their area is about 1500 ha (Fig. 1). Extension of fields was accompanied by their modernization, concerning preliminary treatment. There also was built a new system of pipelines bringing sewage to the farm. In 1973 there was put into operation the experimental farm in Szewce, serving to diminish the load of irrigated fields (Fig. 1). The irrigated soils consist of light and compact medium.

Subsoil consists of small-grain sands, sands with gravel or gravel with pebbles. Impenetrable layer is at the depth of 10 - 20 meters. Filtration coefficient of aquiferous layer is 4×10^{-4}m/s, as determined by experimental pumping. On this area there are irrigated mainly the grasslands (1081 ha), 45 ha of filtration fields serve the purpose of around the year treatment of sewage. The fields are divided into plots irrigated using the basin slope systems. The area of individual plots is 0.35-4.0 ha. Other technical data are shown in Table 1.

Sewage to be used for irrigation is treated mechanically in 12 two-chamber settling tanks (capacity of each chamber - 10,000 - 18,000 m^3). The tanks are situated at irrigation farm, along the main pipeline bringing sewage. Average period of chamber filling with sludge is 4 to 5 years. Sludge drying, performed during summer, takes 4-5 months. During such long time there occurs almost complete mineralization of preliminary sludge. Dewatered and dried sludge is removed from chambers and sold. Empty chambers are prepared for the next cycle.

PHYSICO-CHEMICAL CHARACTERISTICS OF MUNICIPAL SEWAGE

Crude municipal sewage applied to irrigated fields is characterized by increased contents of most contaminants (Table 2). This sewage is more concentrated than typical municipal sewage. In principle, its composition is characteristic for sewage from large urban areas. Statistical analysis of the content of some sewage components indicates large variability of organic contaminants, represented mainly by chemical oxygen demand and dry residue.

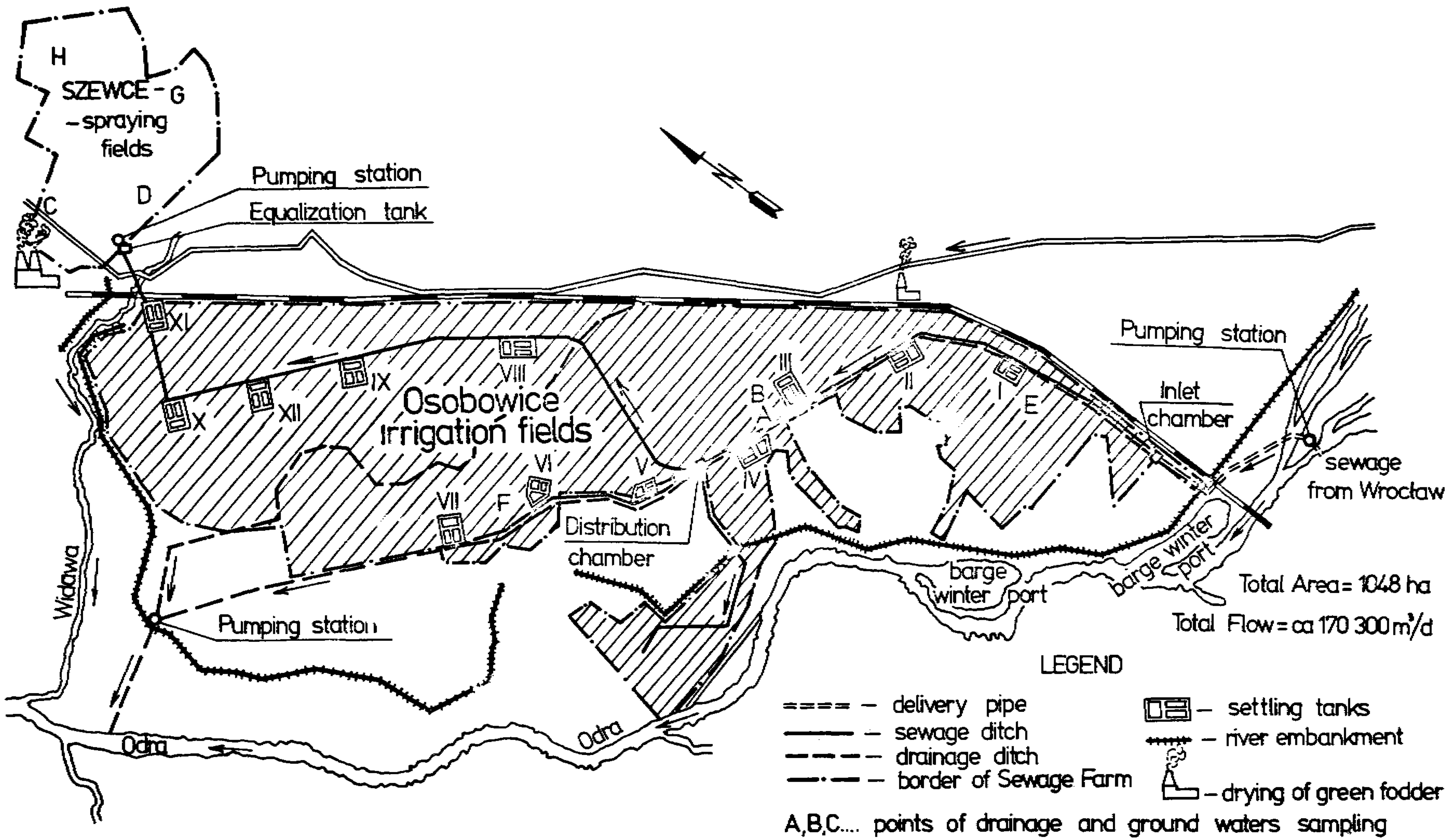

Figure 1 Wroclaw sewage farm

Table 1

Principal technical and economical data of Szewce and Osobowice farms

No.	Specification	Farm	
		Osobowice	Szewce
1.	Irrigation system	basin slope and basin	semi-permanent sprinkling installation
2.	Area in hectares	1048	445
3.	Average load*	5720	990
4.	Annual operation costs	4300	6000
5.	Unit costs of treatment (zl/m^3) of sewage	0.08	0.61
6.	Employment (persons/100ha)		
	- irrigation	1.0	2.2
	- conservation	1.1	1.3

*including filtration and dumping fields

In the analysis of sewage attention was also paid to the presence of trace elements. However, their contents (Table 2) are not excessively high, except for zinc. The concentration of this element is above values permissible for sewage used in agriculture. Crude sewage from Wroclaw does not represent yet a serious source of heavy metals but, at the same time, it is not possible to dump this sewage to receiving waters without treatment.

Analysis of the effluent from settling tanks, i.e. of sewage directly applied to fields, indicates a degree of treatment which is higher than in classical mechanical treatment (Table 2).

EFFECTS OF SEWAGE TREATMENT

The effects of sewage treatment on irrigation fields were assessed by analysis of drainage waters coming from underground systems and by analysis of ground waters present under the drainage system. Essential analytical results are shown in Table 3. These results demonstrate that the levels of many contaminants are very effectively reduced.

Observation of ground water composition, conducted for two years at the irrigated fields by means of an array of piezometers and drilled wells, served to evaluate the effect of irrigation with sewage on ground waters. The conclusions are as follows:

1. Ground waters in areas irrigated with municipal sewage can be considered as an important source of water for many uses.

2. Ground waters, being continuously re-supplied by irrigation, can be more reliable as water source for many users than surface waters.

3. At suitable geological conditions (large thickness of aquiferous layer), the ground waters at irrigated fields can be regarded as an underground reservoir serving to assure a constant supply of water during different periods. Such conditions prevail at the Osobowice farm, presented in this article.

4. Chemical and bacteriological analysis of ground waters from piezometers situated on irrigated fields demonstrated quite good sanitary indices, often better than those for water from farm wells, situated beyond the limits of irrigated fields.

5. Still better, from the sanitary point of view, are waters taken from deeper layers by means of deep wells. Test pumping demonstrated

Table 2. Contamination indices of municipal sewage used for irrigation

No	Index	Units	Crude sewage	Mechanically treated sewage
1.	pH	pH	6,5 - 7,7	7,3 - 7,5
2.	Acidity	mval/l	0,0 - 1,0	0,4 - 0,6
3.	Alkalinity	mval/l	3,4 - 10	3,3 - 3,4
4.	BOD_5	mg O_2/l	160 - 320	110 - 116
5.	Permanganate value	mg O_2/l	180 - 240	57 - 74
6.	Chemical oxygen demand	mg O_2/l	967 - 1237	191 - 294
7.	Chlorides	mg Cl/l	142 - 200	156 - 168
8.	Conductivity	us	1560 - 2000	1650 - 1890
9.	Total solids	mg/l	1061 - 1721	914 - 1001
10.	Total dissolved solids	mg/l	489 - 870	726 - 785
11.	Total suspensions	mg/l	372 - 851	186 - 228
12.	Sulphates	mg SO_4/l	147 - 192	172 - 215
13.	Total nitrogen	mg Nog/l	45,3- 50,9	41,9 - 45,3
14.	Organic nitrogen	mg Norg/l	15,8 - 19,1	13,9 - 14,0
15.	Total phosphorus	mg P/l	8,1 - 193	12 - 15,1
16.	Sodium	mg Na/l	83 - 115	-
17.	Potassium	mg K/l	25 - 43,5	-
18.	Calcium	mg Ca/l	35 - 185	-
19.	Arsenic	mg As/l	0,001 - 0,076	-
20.	Mercury	mg Hg/l	0,4 - 2,0	-
21.	Boron	mg B/l	0,13 - 0,24	-
22.	Silver	mg Ag/l	0,001 - 0,005	-
23.	Copper	mg Cu/l	0,11 - 0,57	-
24.	Nickel	mg Ni/l	0,06 - 0,15	-
25.	Zinc	mg Zn/l	1,07 - 2,80	-
26.	Lead	mg Pb/l	0,02 - 0,1	-
27.	Cadmium	mg Cd/l	0,012 - 0,100	-
28.	Manganese	mg Ma/l	0,10 - 0,35	-

Table 3. Physico-chemical composition of drainage and ground waters from irrigated fields.

No	Index	Units	Drainage water A,B	Ground water from field E	Ground water from field F	Drainage ditches
1.	pH	pH	6,9 -7,1	7,0	7,2	6,0 - 7,2
2.	Conductivity	us	765 - 915	900	1215	1025 - 1380
3.	Chemical oxygen demand	mgO_2/l	13.8 - 16.0	2,4	4.3	3.9 - 16.0
4.	BOD_5	mgO_2/l	12.4 - 14.8	2.2	1.0	1.6 - 17.6
5.	COD_5	mgO_2/l	16.5 - 40.6	40.6	10.2	9.9 - 60.9
6.	Ammonia	$mgNH_4/l$	6.5 - 8.2	2.8	2.8	2.8 - 9.1
7.	Organic nitrogen	NgNorg/l	1.8 - 2.6	28.0	2.8	1.4 - 7.0
8.	Total nitrogen	mgNog/l	9.1 - 10.0	30.8	5.6	7.8 - 10.5
9.	Sulphates	$mgSO_4/l$	160 - 254	342	272	125 - 238
10.	Sodium	mgNa/l	131	144	90	68 - 138
11.	Potassium	mg K/l	32 - 36	18.0	38	7.0 - 265
12.	Calcium	mgCa/l	210 - 235	225	130	200 - 285
13.	Arsenic	mgAs/l	0,04	0,05	0,009	0,019- 0,032
14.	Mercury	mgHg/l	0.2 - 0.2	0.7	0.8	0.2 - 0.9
15.	Boron	mgB/l	0.240	0.185	0.11	0.137 -0.183
16.	Silver	mgAg/l	0.002 -0.003	0.006	0.002	0.003 -0.004
17.	Copper	mgCu/l	0.040 -0.060	0.210	0.03	0.020 -0.110
18.	Nickel	mgNi/l	0.075 -0.120	0.060	0.095	0.061 -0.120
19.	Zinc	mgZn/l	0.68 - 1.46	1.27	1.18	0.20 - 2.83
20.	Lead	mgPb/l	0.034 -0.052	0.024	0.043	0.023 -0.043
21.	Cadmium	mgCa/l	0.002-0.011	0.005	0.003	0.003 -0.012
22.	Manganese	mgMn/l	0.090-0.64	0.086	0.430	0.170 -0.750

that water quality from such wells is within standards for drinking water.

Analysis of soils from irrigated fields gave results shown in Fig. 2. The overall picture resulting from these analyses is not too negative. Higher concentrations of trace elements are of course found in upper soil layer, in comparison with deeper layers, thus demonstrating the sorptive action of soil, but concentration values are almost the same as those given in the report by Page (2) for soils which are not irrigated with sewage.

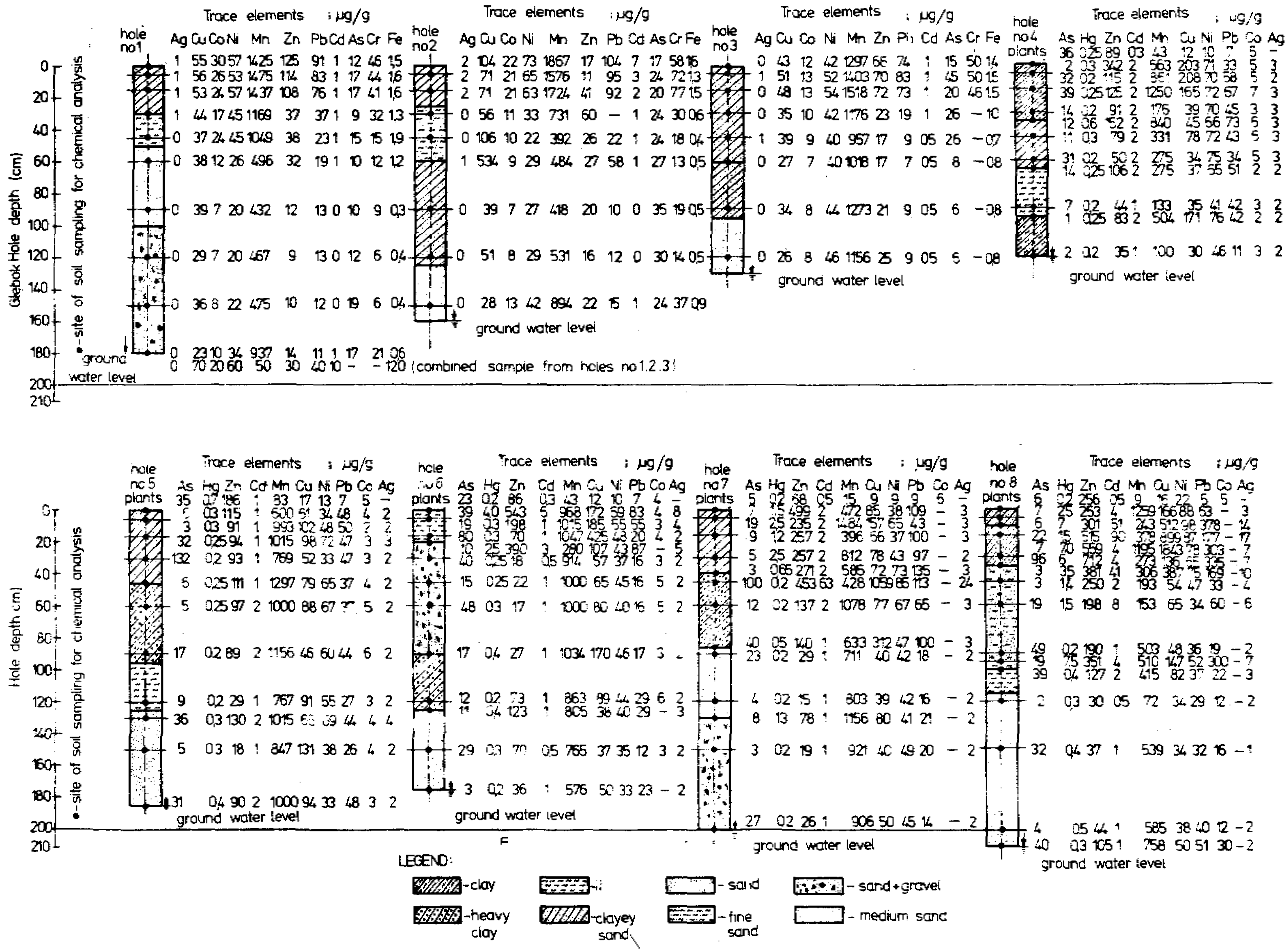

Figure 2 Occurrence of selected trace elements in the soil.

AGRICULTURAL EFFECTS OF SEWAGE UTILIZATION

Dense drainage, large soil permeability and temperature of sewage have the effect that vegetation on grass and pasture land starts sooner than on non-irrigated fields or fields irrigated with pure water. Cattle can start grazing towards the end of April and continue to the end of November. Harvesting of grasses begins in the first ten days of May (in exceptional years even 20-30 April) and ends 20-30 November. Due to large irrigating doses and frequent harvesting only few species of grasses grow on these fields, mainly spear-grass, orchard grass, canary grass and meadow grass. Spontaneous monocultures of these grasses were formed on some areas. There are encountered also small numbers of tussock-grass, fescue and herbs, improving the taste of grasses. In 1973-1975 the average yield of green grass was 400 q/ha and even 700 q/ha from some plots. Grasses are harvested with 9 efficient self-propelled mowing machines using one-phase or two-phase harvesting. Grasses are transported to thermal drying plants. Three such plants (SB 1500 type, Dutch license) are working for the sewage farm and are situated in Osobowice and Szewce. Combined output of these drying plants approaches 4500 kg/hour of hay meal. One-phase harvest of grasses ensures continuous harvesting also in rainy weather while, during good weather, the two-phase system permits partial drying on field. This saves energy but brings about losses of protein. Thermal drying plants are a necessity in agricultural utilization of sewage because traditional drying is not applicable due to high protein content in grasses and necessity of harvesting regardless of weather.

Average hay meal contains: protein 20%, fiber 22%, silica 3%, moisture 10%, as calculated for dry weight. These values correspond to the Extra type of hay meal, the majority of which is exported. Some laboratory analyses indicate exceptionally high protein content, approaching 28%. The quality of hay meal is determined mainly by protein and fiber content, depending on the time of harvest, and by the content of silica, resulting from contamination with sand. Silica content decreased when mowing machines were equipped with hydraulic adjusting of the height of mowing.

CONCLUSIONS

1. Agricultural utilization represents one of the most effective methods of sewage treatment.

2. Properly designed and operated systems of agricultural sewage utilization do not pose any hazards to natural environment.

3. Agricultural systems of sewage treatment have a very advantageous effect on hydrogeological conditions on irrigated fields.

4. Using of sewage for irrigation provides an opportunity of increasing the yields of cultivated plants, particularly of fodder plants which, after suitable treatment, are valuable as high-protein feed.

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EVALUATION OF EXISTING SYSTEMS

LAND TREATMENT OF MUNICIPAL WASTEWATER ON STEEP FOREST SLOPES IN THE HUMID SOUTHEASTERN UNITED STATES

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There is little information available for many climatic and physiographic regions to aid in the design and interpretation of land treatment systems. One such region is the upland, humid region of the southeastern United States. The region is characterized by moderate to high rainfall, deeply weathered soils, moderate to steep slopes and forest vegetation. A research program was initiated in 1973 to develop the design information base for land treatment systems. The region prototype research project was established on a 30 percent forest slope receiving a weekly municipal wastewater loading of 76 mm, for a combined annual loading of wastewater and precipitation of 5600 mm. Renovation of the applied wastewater constituents has been high. Site hydrology plays an important role in the nitrogen renovation through enhanced denitrification. Soil and vegetation responses to the applied wastewater indicate a general increase in soil and vegetation nutrient storage and forest tree growth. The results of the prototype study were applied in the design of a municipal wastewater land treatment system with a daily flow of 7.4×10^4 m^3. The final design requires a weekly wastewater loading of 63 mm on 957 ha of a 1475 ha contiguous forested site. The domestic water supply intake for the municipality is 11 km downstream from the land treatment site and the return flow will augment the water supply.

INTRODUCTION

The land treatment of municipal wastewaters has been practiced worldwide for many years, yet little information is available for many specific climatic and physiographic regions to aid in the design and operation of land treatment systems. Specific knowledge of the hydrologic and chemical interactions between soil, geology and vegetation is necessary to properly design land treatment systems and predict with some degree of certainty the long term impacts of the system operation. One such region where this type of response information is needed is the upland southeastern United States.

The upland portions of the southeastern United States include the Piedmont Plateau and the southern Appalachian Mountain physiographic provinces. The climate of the region is classified as humid, temperate with mean annual precipitation ranging from 1200 mm to over 2000 mm and mean annual temperature ranging from 12 C to 17 C. Soils are generally developed from residual parent material and have deep, well-developed profiles. The soils are acid with low to moderate cation exchange capacities and have textures ranging from sandy loam to clay. The major rock formations from which the residuum and soils developed are granite, hornblende, and biotite gneiss, schist and quartzite. The residuum, or saprolite, often ranges in depth from 1 m to 20 m depending on the rock formation. Slopes range from gentle to steep and the drainage pattern is usually deeply incised and dendritic.

Forests occupy 70 percent of the region, however, the slopes greater than 12 percent are nearly 100 percent forested. Principal forest species are oak (Quercus spp.), hickory (Carya spp.), loblolly pine (Pinus taeda L.) and shortleaf pine (Pinus echinata, Mill.).

Although agricultural sites within the region are suitable for land treatment, forest sites have gained favor for several reasons. They are: forest land is generally readily available at a more favorable cost, forest land is more easily managed than agricultural land under the constraints of high precipitation and wastewater loading and forests are removed from the human food chain. If forests are to be used extensively for land treatment in the region, the moderate to steep slopes will have to be utilized as part of the land treatment system. Sepp (1973) reports general success in the use of forested mountain slopes for land treatment in California. The infiltration capacities of the forest floor and underlying mineral soil are generally high and the applied wastewater moves through the soil as rapid subsurface flow down the slopes receiving a high degree of renovation. Nutter (1973 and 1975) and Hewlett (1961) have shown that soil water movement is laterally down the slope which improves opportunities for renovation of the wastewater by increasing the pathlength of flow through the more reactive soil and root zones. Nitrogen renovation by denitrification, in particular, could possibly be enhanced by the periodic anerobic conditions developed at the base of the slopes.

Reported in this paper are the results of a four year continuing study on the renovation of wastewater applied to steep forest slopes and the extrapolation of the results to the design of a 0.9 m^3/s (7.8×10^4 m^3/day) land treatment system on 1475 ha of forest land near a large metropolitan area.

THE RESEARCH STUDY

In 1973 a land treatment facility utilizing steep forested slopes was designed for the Unicoi State Park and Conference Center in the Blue Ridge mountains near Helen, Georgia. The park operates throughout the year and produces a wastewater similar to that expected from a small, non-industrial community. The 1.5 ha site selected for application of wastewater is located in an area typical of much of the land below the 800 m elevation in the Blue Ridge mountains and ranges in elevation from 490 m to 520 m with slopes of 15 to 30 percent and lengths of 90 m to 110 m.

The predominant soil is a Hayesville sandy loam, a member of the clayey, oxidic, mesic family of Typic Hapludults. The texture grades from a sandy loam at the surface to clay in the B2 horizon (25-70 cm), to clay loam in the B3 horizon (70-120 cm) and loam in the C horizon (120-155 cm). The saprolite extends in many places to a depth of 8 m or more. The soil pH averages 5.4 and cation exchange capacities range from 5 to 14 meq/100 g (Gibbs and Perkins, 1966).

Vegetation on the site consists of a mixed hardwood-pine forest with predominant overstory species of oak (Quercus spp.), white pine (Pinus strobus L.), and Virginia pine (Pinus virginiana Mill.). The lower slopes contain a dense understory of mountain laurel (Kalmia latifolia L.) while the upper slopes contain a mixed herbaceous-woody plant understory with blueberry (Vaccinium spp.) and huckleberry (Gaylussacia spp.) the predominant species. The stand is uneven-aged with an average age of approximately 45 years.

The climate is characterized by warm summers, cool winters, and the high rainfall, averaging 1800 mm per year, is somewhat uniformly distributed throughout the year with a maximum in spring and a minimum in early fall. Only a few days of above 32 C or below 0 C temperatures are recorded each year and the mean annual temperature is 15 C.

Wastewater is applied to a depth of 76 mm one day each week at a rate of 8 mm per hour. The irrigation system consists of above-ground portable aluminum pipe with impact-type sprinklers on 1.5 m risers at 12 by 18 m spacing. The operating pressure is 3.2 kg/cm^2.

Wastewater Treatment

The wastewater, averaging 1.03×10^{-3} m^3/s, is collected and delivered to a 0.4 ha oxidation lagoon of which 0.1 ha receives mechanical aeration. The wastewater is chlorinated as it is pumped from the lagoon. The range and mean quality of the wastewater applied to the land treatment site during the 42 month period January 1974 through June 1977 is presented in Table 1. Also shown in Table 1 is the amount of each constituent applied annually in kg/ha.

Table 1. Quality of the Wastewater Applied to the Land Treatment Site during the Period January 1974 through June 1977.

Constituent	Concentration Minimum	Maximum	Mean	Amount Applied[1/]
	----------mg/1----------			kg/ha/yr
Total Kjeldahl N	4.8	37.1	18.0	684
Ammonia - N	2.3	14.6	7.1	270
Nitrate - N	0.02	5.8	0.49	19
Total phosphorus	1.6	22.1	12.1	460
Calcium	0.11	30.8	9.5	361
Magnesium	0.44	3.9	1.4	53
Sodium	22.0	52.1	34.8	1323
Potassium	0.49	33.7	10.5	399
Chloride	16.5	54.1	29.0	1103
Total organic carbon as carbon	42	405	220	8366
Biological oxygen demand - 5 day	22	78	60	2282
pH	5.0	6.6	5.7	

[1/] Fifty irrigation periods per year with 76 mm wastewater per period.

Quality of the Renovated Wastewater

The portion of the study area selected for monitoring is uniform and wastewater is irrigated on an area beginning at the ridge and extending 67 m down a 30 percent slope. The base of the slope is approximately 95 m from the ridge. Soil water percolate is sampled on the irrigated site and an adjacent non-irrigated site and at the base of the irrigated slope where the lateral subsurface drainage collects. Vacuum porous cup samplers are installed at four depths and four locations in both the irrigated and non-irrigated areas and at three depths and three locations along the base of the irrigated slope. A nearby first-order stream is also monitored to provide regional background water quality. The mean quality of the soil percolate at each sample depth and the nearby stream for the period January 1974 through June 1977 is presented in Table 2.

In general, the results presented in Table 2 indicate that with the exception of Na, Cl and NO_3-N, the concentrations within the soil percolate at the base of the slope are little different from the concentrations found in the soil percolate in the non-irrigated area or the regional natural water quality as evidenced by the stream. The reduction or renovation of phosphorus in the water collected from the surface 275 cm of the irrigated site averages 97 percent. Comparison with the non-irrigated site indicates little difference in concentration, although the total amount leached from the irrigated area would be greater due to greater volumes of water moving through the soil. The renovation of calcium, potassium and magnesium within the irrigated area is also high and renovation is improved as the wastewater percolates to the base of the slope. Although chloride is subject to some anionic adsorption, the low reduction in concentration as the wastewater moves from the irrigated area to the base of the slope does serve to illustrate that the applied wastewater is moving laterally down the slope. Sodium, on the other hand, is subject to cationic adsorption but being weakly bonded is replaced by other more strongly adsorbed cations allowing a high proportion of the sodium to be flushed to the base of the slope.

As evidenced by the results presented in Table 2 there is little change in the soil percolate quality with depth in the irrigated area and at the base of the slope or in the non-irrigated area.

Nitrate-nitrogen is a critical element in the design and evaluation of land treatment systems because it is not only a preferred plant nutrient but is highly mobile in the soil percolate and concentrations greater than 10 mg/1 in drinking water can be hazardous to human health. The nitrogen cycle for a land treatment site is complex and the degree

Table 2. Mean Quality of the Soil Percolate for the Period January 1974 Through June 1977 for the Irrigated, Non-Irrigated and Base of Slope Areas and a Nearby Stream.

Site and Depth	P	Ca	Mg	K	Na	Cl	TKN	NH_4-N	NO_3-N
------cm------					mg/l				
Irrigated									
30	0.45	0.47	0.65	2.7	26.1	25.1	1.5	0.02	7.7
60	0.27	0.37	0.76	1.9	25.3	24.4	2.0	0.02	8.1
120	0.26	0.20	0.47	1.5	23.2	23.7	1.6	0.01	7.2
275	0.49	0.63	0.69	2.0	26.1	26.2	1.5	0.03	8.7
Base of Slope									
60	0.25	0.12	0.57	0.98	15.6	16.5	1.3	0.01	3.2
120	0.20	0.10	0.47	0.78	15.5	19.0	0.67	0.01	3.5
200	0.24	0.12	0.48	1.1	17.5	19.3	0.97	0.02	4.5
Non-Irrigated									
30	0.22	0.40	0.50	0.56	0.81	1.9	0.07	0.02	0.006
60	0.22	0.22	0.65	0.47	0.63	1.5	--	0.01	0.008
120	0.27	0.19	0.34	0.45	0.65	1.6	0.17	0.01	0.005
275	0.23	0.10	0.32	0.49	0.65	1.4	--	0.02	0.008
Stream	0.29	0.64	0.22	0.58	1.8	1.1	3.1	0.07	0.03

of renovation is a function of the form of nitrogen in the wastewater, net plant storage, soil temperature, pH, moisture regime and other factors (Broadbent, et al., 1977 and Iskander, et al., 1976). The organic and ammonia forms of nitrogen are readily retained in the soil throughout the year and transformation to the easily leached nitrate form occurs primarily during the summer months. Net tree storage of nitrogen is low due to the annual recycling of leaves. Therefore, the primary manner in which nitrogen is lost or renovated in a forest land treatment system must be nitrate in the percolate or nitrogen gas diffusion to the atmosphere following denitrification.

Since the NO_3-N concentration in the wastewater is low (Table 1) nitrification must take place before there can be a substantial loss of nitrogen. As shown in Table 2, a substantial portion of the nitrogen leached from the site is in the NO_3-N form. There is a better than 50 percent reduction in NO_3-N concentration as water moves from the irrigated area through the area at the base of the slope. These results and several other observations help to clarify the interactions and relationships between nitrification and denitrification.

In July 1974 the oxidation lagoon aerators were turned off resulting in an increase in the total nitrogen loading to a rate equivalent to over 900 kg/ha/yr. The NO_3-N concentration in the irrigated area soil percolate increased markedly to greater than 20 mg/l and the concentration at the base of the slope reached a peak of 10 mg/l, the drinking water standard limit. When lagoon aeration was resumed one year later the nitrogen loading decreased and there was an almost immediate drop in nitrate levels in the soil percolate. The levels in the soil percolate continued to decline as the levels of organic and ammonia nitrogen stored on the site during the increased loading also declined. Thus, the denitrification capacity of the site under the given moisture regime and availability of carbon was exceeded by an overload of nitrogen.

The temperature dependence of nitrification and denitrification is also apparent from observations of NO_3-N in the soil percolate. As soil temperatures increase in the spring NO_3-N concentrations also increase. Ammonification of the organic matter applied to the site during the winter increases at the same time and the released ammonia plus that applied in the wastewater and stored during the winter is readily available for nitrification. Within several weeks a decrease in NO_3-N concentrations is noted which is likely due to active uptake of nitrate and ammonia by plants, and increased rates of denitrification.

A nitrogen budget based on an

Table 3. Occurrence of Nitrifying and Denitrifying Bacteria and the Evolution of Nitrogen Gas (after Rowe, et al., 1976)

Site	Bacteria: Nitrifiers	Bacteria: Denitrifiers	N_2 Gas Evolution
	Number/g dry soil		$\mu m/m^2/day$
Irrigated	280000[a]	56000[b]	12.7[a]
Base of Slope	3100	50300[c]	7.2[a]
Non-Irrigated	4100	2400	0.7

Mean significantly different from non-irrigated site mean at a, 95 percent; b, 90 percent; and c, 50 percent.

assumed hydrologic budget for the irrigated and base of the slope areas indicates that the difference between nitrogen input and output as measured in the wastewater and soil percolate is approximately 470 kg/ha/yr. To determine if denitrification may account for part of the unaccounted nitrogen a study was carried out from May through November 1975 to measure relative rates of N_2 gas evolution from the soil and the populations of nitrifying and denitrifying bacteria (Rowe, et al., 1976). Numbers of nitrifying bacteria on the irrigated area (Table 3) are significantly higher than in the non-irrigated or the base of slope area. In contrast, the number of denitrifiers on the irrigated and the base of slope areas are significantly higher than on the non-irrigated area. More important than the presence of the denitrifying bacteria is the significant increase in N_2 gas evolved from the irrigated and the base of slope area as compared to the non-irrigated area.

Recent studies by Hook and Kardos (1978) and Barr, et al. (1978) point out the importance of maintaining periodically saturated soil conditions as well as maintaining a readily available source of carbon if denitrification rates are to be enhanced. Barr, et al. (1978) report that carbon from plant residues is not adequate, except for short periods of time during the year, to maintain rates of denitrification above natural levels. They suggest that available carbon in the wastewater itself be increased through a decreased level of treatment.

As shown in Table 1, the total organic carbon in the wastewater is high and could be expected to supplement the plant residue carbon sources. Apparently the degree of saturation and source of carbon are sufficient at the base of the slope to maintain denitrification at a rate perhaps as high as 60 percent of that on the irrigated site. Groundwater levels at the base of the irrigated slope range from 30 cm to greater than 200 cm below the surface. Saturation at the base of the slope in the 60 to 120 cm depth occurs frequently within 12 to 48 hours following irrigation, lasting from 24 to 72 hours depending on the season of the year and precipitation. Although we cannot estimate at this time how much of the unaccounted nitrogen in the budget can be attributed to denitrification, it is apparent that it is a major source of nitrogen loss from the system.

Soil Nutrient Responses

Chemical changes in the land treatment site soils have been monitored at the end of each growing season for the past four years. Four depths between the surface and 120 cm are sampled at six locations each year in both the irrigated and non-irrigated areas. Extractable cations Mg, Na, Ca, and K are determined following extraction with 0.025N H_2SO_4 and 0.05N HCl, while the total P and total Kjeldahl nitrogen is determined following digestion with a solution of H_2SO_4 and H_2SeO_3. In general, the irrigated area showed higher mean concentrations of P, K, Ca and Na over the four year period at each sampled depth while Mg concentrations in the irrigated soil were higher in the top 60 cm of soil but lower in the 60 to 120 cm depth. The total Kjeldahl nitrogen in the irrigated soil was only slightly greater than in the non-irrigated soil at all depths except the 15 to 30 cm depth where it was slightly

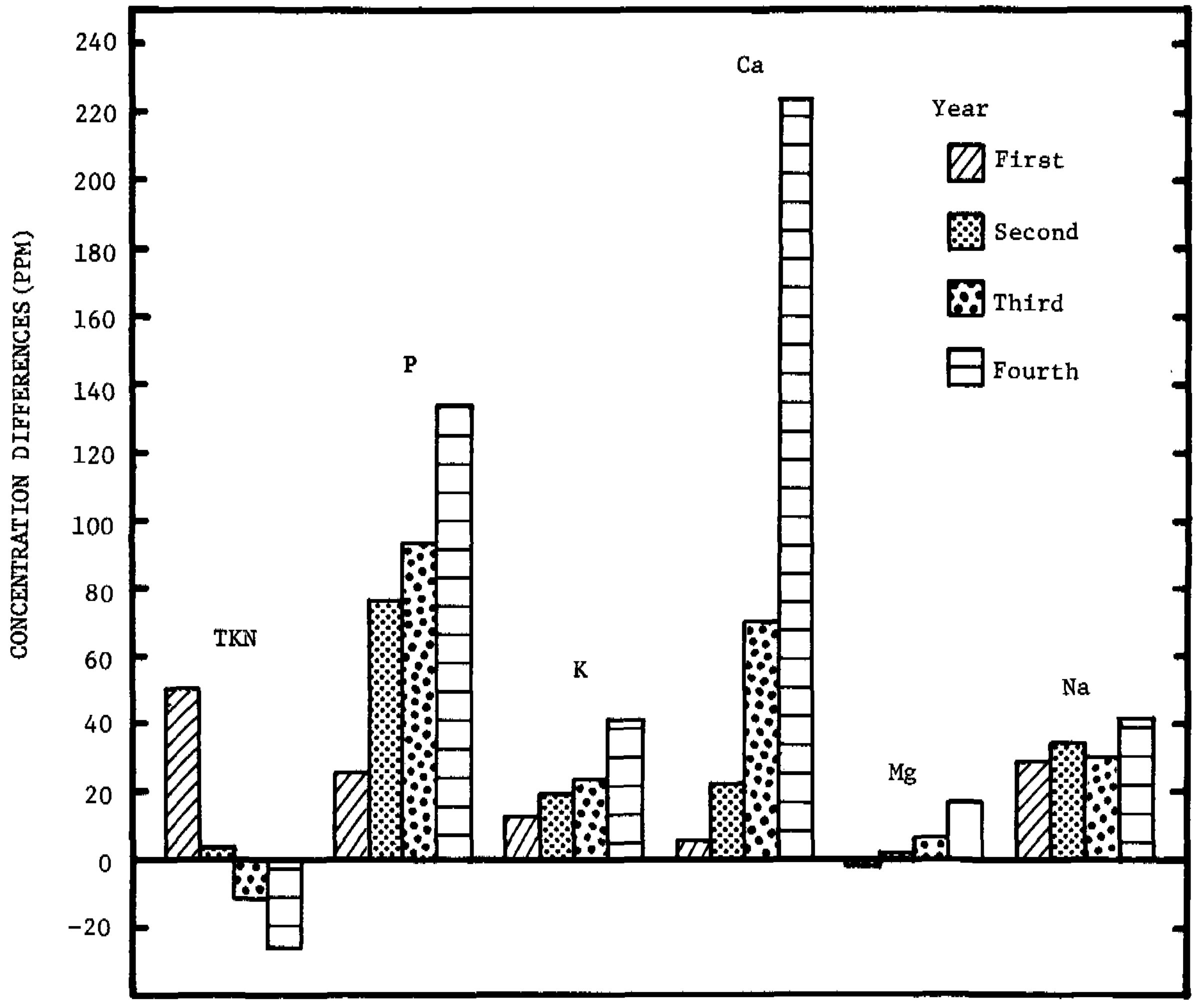

Figure 1. Soil Nutrient Concentration Differences Between Irrigated and Non-Irrigated Soils During the Four Years of Irrigation (Depths Pooled).

less.

The trends suggested by these four year mean data can be seen more clearly by comparing the concentration differences between the irrigated and non-irrigated soils for each year as presented in Figure 1. Differences between the P, K, Ca and Mg concentrations of the irrigated and non-irrigated soils have increased steadily over the four years of treatment. The differences have been greatest for Ca which is generally considered to be held more tightly by the soil colloid than Mg, K or Na. The greatest differences in calcium concentration have occurred at the shallow depths with differences decreasing with increasing depth. Less K has been adsorbed by the soil as evidenced by the smaller differences between the irrigated and non-irrigated soils and the greater proportion of K leached from the site (Table 2). The differences in K concentration between the irrigated and non-irrigated soils were greatest at the shallow depths during the first two years of irrigation. However, by the fourth year there were no obvious differences between the depths of the irrigated soils. Only small quantities of Mg have been added in the wastewater. However, differences in Mg concentration between the irrigated and non-irrigated soils have also been increasing steadily since irrigation began. The greatest differences in the irrigated soils have taken place in the upper 30 cm of the profile. Sodium concentrations have been consistently higher in the irrigated soil since it is the least tightly held of the cations and therefore the most mobile. This is strongly suggested by the fact that differences in depth in the irrigated soil over the treatment period have remained almost constant since the first year of irrigation.

Phosphorus has responded as expected since the soils have a high phosphorus fixation capacity. The added phosphorus is quickly immobilized and the greatest concentration is found in the surface layers of the soil. The differences in P concentration between the irrigated and non-irrigated soils have been increasing at nearly a linear rate since irrigation began.

Total nitrogen in the soils has shown an interesting response. Over the four year irrigation period the differences in concentration between the irrigated and non-irrigated soils have gone from a gain to an increasing loss. This would suggest that the irrigated soil is losing nitrogen in spite of large additions from the wastewater. The losses are occurring in the upper 15 cm of soil while gains are occurring below 15 cm. This would suggest that the added organic matter and the decomposing organic matter of the forest floor are moving into lower depths of the soil profile. The depth of the forest floor on the irrigated area appears to be decreasing, although the total forest floor biomass has remained unchanged, while the structure of the forest floor is changing from a mor to a mull. The changes observed seem reasonable to expect in light of the higher calcium contents (increased pH) and the added moisture. In general, the responses in soil chemistry that have been observed at Unicoi are similar to those found by Richenderfer, et al. (1975) for northern forested soils.

Vegetation Responses

To identify the response of the vegetation to wastewater irrigation both growth and nutrient data were collected from the site during the first three years. Comparisons between the irrigated and non-irrigated vegetation have indicated that both growth and nutrient contents have been altered as a result of irrigation.

Both diameter at breast height and height of the dominant trees have been significantly increased as a result of irrigation. Understory species have not shown any significant stimulation from the irrigation. The overstory on the site is quite dense allowing little light penetration to support a vigorous understory. A significant decrease in both the laurel understory and the lower ground vegetation has occurred as a result of irrigation. The laurel, which was the densest understory shrub species on the lower slopes, has disappeared from the irrigated area. High moisture and ice damage from year round irrigation may be the reason for its demise. Likewise, the blueberry (*Vaccinium* spp.) and the huckleberry (*Gaylussacia* spp.) which were the most prevalent shrub species on the upper slopes have also been greatly reduced in numbers since irrigation began and red maple (*Acer rubrum* L.) and poleweed (*Phytolacca americana* L.) appear to be taking their place. Both of these latter species are usually found on moist sites. The high moisture conditions on the irrigated area have made the dominant, shallow-rooted, white pine more susceptible to windthrow. High winds have blown over several of these trees on the irrigated area but none on the non-irrigated area.

The total nutrient content of the trees on the irrigated area showed significant increases over similar trees on the non-irrigated area (Table 4). Total Kjeldahl nitrogen, P, K and Mg showed increases in a majority of the tissues of the trees. However, the significant differences were usually a result of large differences in only one or two of the seven major species sampled. No one species had consistently high concentrations in all tissues. Although differences between the irrigated and non-irrigated trees exist, the large inherent variability and the small number of trees available for sampling ruled out any clear identification of species differences. Similar problems occurred in sampling the nutrient concentrations of the lower vegetation.

The understory, or lower vegetation, and forest floor nutrient contents were also determined and the results are presented in Table 5. Although there were increases in concentrations of TKN, P, K and Mg in the irrigated lower vegetation, none of the differences are significant.

The nutrient data for the plants is somewhat inconclusive in this experiment. Although steep slopes require a forest cover to be considered for irrigation, the plant community does not seem to play a major role in the renovation process by taking up and storing large quantities of the nutrients. Natural stands of second growth mixed pine-hardwood forests are not highly efficient renovators of applied wastewater effluents. No doubt some improvement in the vegetation's role can be expected through species selection and intensive forest management.

Table 4. Mean Nutrient Concentrations for all Trees Measured During the Study Period.

Constituent and Area	Bark	Branches	Twigs 1st yr	Twigs 2nd yr	Deciduous leaves	Conifer leaves 1st yr	Conifer leaves 2nd yr
	Percent						
TKN							
Irrigated	1.04[a]	.80	1.71[a]	1.24	2.05[a]	1.64[a]	1.64
Non-Irrigated	0.73	0.84	1.24	1.05	1.59	1.13	1.56
	ppm						
P							
Irrigated	354[a]	668[a]	1309[a]	918[a]	1951[a]	1427[a]	1158[a]
Non-Irrigated	215	419	904	600	1063	1040	816
K							
Irrigated	2156[a]	2098[a]	5439	3555[a]	5331	5921[a]	4786
Non-Irrigated	1180	1562	4961	2657	5155	4560	4366
Ca							
Irrigated	9075[a]	4519	5831	4620	8580[b]	1945	3090
Non-Irrigated	6399	4530	5678	5028	10390	1707	3284
Mg							
Irrigated	435[a]	596[a]	1029[a]	547[a]	2777	1267	829
Non-Irrigated	231	488	689	485	2904	1011	821

Concentrations in irrigated trees are significantly different from means in the non-irrigated trees at a, 99% and b, 95%.

APPLICATION TO SYSTEM DESIGN

The prototype land treatment system at Unicoi has served in several ways to improve regional land treatment design concepts. Regulatory agency officials and designers have been able to observe a system within the region in operation over an extended period of time. The study results have been directly applied in establishing regulatory agency guidelines. The regional prototype system has been of greatest benefit in providing the basis for design of systems within the region.

Although several large land treatment systems in excess of 500 ha are currently in operation, few have been designed under the constraints of the Federal Water Pollution Control Act Amendments of 1972 (P.L. 92-500) requiring wastewater treatment facility planning in which the most cost-effective plan must be implemented. One such study in accordance with the Act was a planning study undertaken for an area in Clayton County, part of the metropolitan Atlanta, Georgia area, located in the Piedmont Plateau province. A number of treatment systems, including land treatment and advanced wastewater treatment, were evaluated for the project's 7.4×10^4 m^3/day flow.

As part of the land treatment feasibility portion of the study, 4800 ha were evaluated using criteria for soils, geology, topography, climate, hydrology, vegetation and other factors established at the prototype study. Land costs near the large metropolitan area were high and system design was further constrained to use of the smallest land area possible. A contiguous 1475 ha site containing better than 80 percent forest cover with the optimum combination of factors that would enable high wastewater loading and an acceptable degree of renovation was selected.

Although the average annual rainfall in the area is 1320 mm, domestic water supply is in short supply due to large populations located in headwater basins. Clayton County's domestic water supply intake is located 11 km downstream from the selected land treatment area and the irrigation return water would augment the water supply through recycling. The recycling posed additional constraints to the land treatment system design, particularly with regard to levels of nitrate-nitrogen leached to groundwater and streams.

Table 5. Mean Nutrient Concentrations for Lower Vegetation and Forest Floor.

Constituent and Area	Lower Vegetation	Forest Floor L & F	Forest Floor Humus
	Component		
	Percent		
TKN			
Irrigated	1.59	1.56	0.76
Non-Irrigated	1.16	1.18	0.71
P			
Irrigated	3026	1172[a]	782[a]
Non-Irrigated	1051	572	520
K			
Irrigated	9679	1550[a]	2264
Non-Irrigated	4946	1012	2056
Ca			
Irrigated	5353	12307[a]	5207
Non-Irrigated	6245	4887	1267
Mg			
Irrigated	1647	1045[a]	605[a]
Non-Irrigated	1181	476	299

Concentrations in irrigated samples are significantly different from means in the non-irrigated plants at a, 99 percent.

Removing bottomlands along streams and land required for buffer from consideration, over 50 percent of the site has slopes greater than 12 percent. This topographic feature plus the good hydrologic condition of the soil made possible by the presence of forest cover indicate that a high hydraulic loading is possible. The principal forest vegetation to be managed on the site will be loblolly pine, although many of the steeper slopes will remain in natural mixed hardwood-pine stands.

Because the net storage of nitrogen in trees has been shown to be low, denitrification must be enhanced to assure adequate nitrate renovation and protection of the drinking water supply. A wastewater loading of 63 mm/week to be applied in a 12 hour period was established as optimum for conditions of the site and the enhancement of denitrification. The steep slopes will direct subsurface flow to create anerobic conditions. Further opportunities for denitrification will occur in the bottomland areas of the site where the water table will fluctuate over a 0.5 m to 1.5 m range. Nitrate-nitrogen concentrations in the applied wastewater will be maintained at low levels such that winter season flush-through of applied nitrates will not occur. The total nitrogen concentration in the applied wastewater is 18 mg/l. The anticipated level of nitrate-nitrogen in the streams during operation of the system will not exceed 5 mg/l. The total wetted area needed for the design wastewater loading is 960 ha.

The system is currently under construction and will be in operation in 1980. An intensive forest management program is currently underway to assure maintenance of the sites' hydrologic condition and to achieve the most vigorous growth conditions possible.

CONCLUSIONS

The prototype land treatment research study in the upland southeastern United States has demonstrated that with proper design and operation steep forest slopes provide more than adequate treatment of applied wastewater. Nitrogen, although recycled rapidly through the forest with little net storage gains, is adequately renovated through denitrifification. The combination of a reasonably high periodic wastewater loading and lateral subsurface flow create moisture conditions on the irrigated area and at the base of the slope that enhance high rates of denitrification. Although forest growth is enhanced to some degree, the uptake and net storage of nutrients within an early-mature forest stand is low and plays little role in the direct renovation of the wastewater.

The regional prototype land treatment system has demonstrated the value of such systems in providing base data and experience for use in aiding design of systems within the physiographic region. This value was realized in the design of a large forest land treatment system where recycling of the land treated wastewater as drinking water will be achieved.

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EVALUATION OF EXISTING SYSTEMS

IRRIGATION WITH UNTREATED SEWAGE WATER AND ITS EFFECT ON THE CONTENT OF HEAVY METALS IN SOILS AND CROPS

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ABSTRACT

Treatment of municipal and industrial sewage waters is practically non existent in Chile. Untreated sewage waters from the Zanjon de la Aguada canal (ZA) which collects 2/3 of the domestic and industrial sewage of the city of Santiago with 3.5 million inhabitants, are used to irrigate approximately 4,500 ha of vegetable producing farmlands. The effects on soils and crops of more than 40 years irrigation with these waters were investigated. Special attention was paid to the content of the heavy metals, Cd,Cr,Cu,Ni, and Zn in the waters, soil profile, and crops. The quantities of Cu in the water-soil-plant system were relatively high but no phytotoxic effects were observable. Cadmium levels in vegetable leaves were below 3 ppm. Leaf Cr,Ni, and Zn values were in the normal range. Cd,Ni and Zn were evenly distributed in the soil profile and averaged 1.5, 18, and 160 ppm respectively down to the water table at the 2 m depth. Copper increased with depth of the soil and reached over 1,000 ppm. Chromium accumulated in the top soil where 50 ppm were found, and decreased with depth to 15 ppm. Fertility of the soil investigated is high and no salt accumulation has occured. Commercial fertilizers are not used and N, P and other nutrients are supplied exclusively by ZA waters. Two crops of lettuce and celery and 4 cuts of swiss chard are harvested each year.

INTRODUCTION

Municipal and industrial waste waters are frequently used to irrigate agricultural land. In many countries these waste waters are treated to eliminate suspended solids, microbial pathogens, and some soluble mainly toxic components before they are returned to natural drainage channels or are used for other purposes. In other countries, such as Chile, few or no treatment facilities exist and raw sewage waters are allowed to flow into streams and rivers or are used directly to irrigate significant agricultural areas. The consequences of these practices with respect to soil and water properties and plant nutrition have not received much atention, except for epidemiological aspects related to the presence of microbial pathogens in crops and waters consumed by the population.

Since water requirements for domestic and agricultural uses are increasing, waste waters constitute an additional source to satisfy increasing needs (Day, 1974). The effect of treated or reclaimed waters and of the sludges removed from them on soils is presently receiving wide attention (Elliot and Stevenson, 1977; Lund et al., 1976; Pratt et al., 1978). Special attention has been focused on the accumulation of heavy metals in soils irrigated with treated waste waters or sewage sludges, nitrate movement in treated soils, and possible contamination of ground and or well waters (King and Morris, 1974; Ludwick et.al., 1976; Pratt et al., 1972). Few studies have been published on the effect of the use of untreated domestic and industrial sewage waters on soils and on crops irrigated with them.

In Chile, treatment of municipal waste waters is practically non existent. However, some efforts have been made in a number of small towns to introduce some form of processing domestic waste waters, by use of sedimentation ponds. One experimental program has been underway in the town of Melipilla (around 60,000 inhabitants) since 1972. Progress made during the last five years has been such that the Ministry of Public Works is now recommending this method for small towns all over the country. For larger cities, however, this approach is not practical and resources for more sophisticated treatments are not available. Therefore in big cities like Santiago with 3.5 million inhabitants, the disposal of sewage waters will remain a problem for many years. At present the raw domestic and industrial sewage is collected by a canal known as Zanjon de la Aguada (ZA) and these waters are used to irrigate vegetable producing agricultural areas. Excess sewage waters are emptied into the Mapocho river, a tributary of the Maipo river, which eventually flows into the Pacific Ocean.

The purpose of this presentation is to briefly summarize some of the results of a study made to determine the influence of the continuous use of untreated sewage waste waters on highly productive agricultural lands, with special emphasis on heavy metal accumulation in soils.

GENERAL DESCRIPTION OF AREA STUDIED

Geographically Chile is located between paralells 18° and 56° L.S. and 71° to 74° longitude west. It is 4,200 km long and on the average, 120 km wide. All major climatic zones except tropical, are present in this country. The northern part down to lattitude 33° S is completely arid except for a few oasis and transversal valleys. Further south and down to 36° L.S. lies a semi-arid region, followed from there to the south by a semi-humid to humid region where rainfall increases from 1,000 to 2,500 mm per year and no dry seasons exists.

The city of Santiago lies in a central valley at approximately 70°30' longitude west and 33° lattitude south, in a semi-arid zone were the mean annual rainfall is around 400 mm with a dry season of around 8 months. The rainy season comprises late May to early September and each rainfall does not exceed 10 to 15 mm. Air temperatures during the dry season reaches highs of 33°C during daytime, dropping to around 10° C at night. Relative humidity during this season rarely exceeds 40%.

During the dry season an area of around 4,500 ha is irrigated exclusively with untreated domestic and industrial sewage waters. On this intensily farmed agricultural land a 15 ha plot was selected for the present study. The area is located on the western and lower, side of the city of Santiago, next to the town of Maipu and is delineated on the north and south by two main highways and on the west by the Mapocho river and the Lo Aguirre mountains. Irrigation water is drawn exclusively from the ZA canal which collects more than 2/3 of the domestic sewage of Santiago and the industrial wastes from medium to small sized industrial operations such as: foundries and tanneries, and metal processing, copper manufacturing, electro-battery, etc., plants. The average flow of the ZA canal at the point where secondary outlets start, varies between 8 and 20 m^3 per second during daytime, with two peaks, one at midmorning and one in the late afternoon. Only a small part of the total water flow is diverted for irrigation, while the major portion empties into the Mapocho river (Figure 1).

The depth of the water table in the irrigated area varies between 1 m and 15 m with an east-west flow. Below the water table lies a compact (up to 80 m deep) non-permeable volcanic deposit which confines an acquifer which, because of the impermeable nature of the layer should not be influenced by irrigation of the overlying land (Trepiana, 1976). Drinking water for a population of a few thousand people comes mostly from wells which are subject to pollution from surface run-off and leaching both from the cultivated soils and from the ZA canals. Figure 1 shows the approximate shape of the water table.

The soils in the area are Vertisols and the soil of the 15 ha plot chosen for this study is a Typic Chromudert. The plot is representative of the whole area. Mainly vegetable crops are grown on this farmland, specially lettuce, celery, swiss chard, cucumbers, squash, and onions. Wheat is also grown to a limited extent and a few deciduous fruit orchards exist. The soils are highly

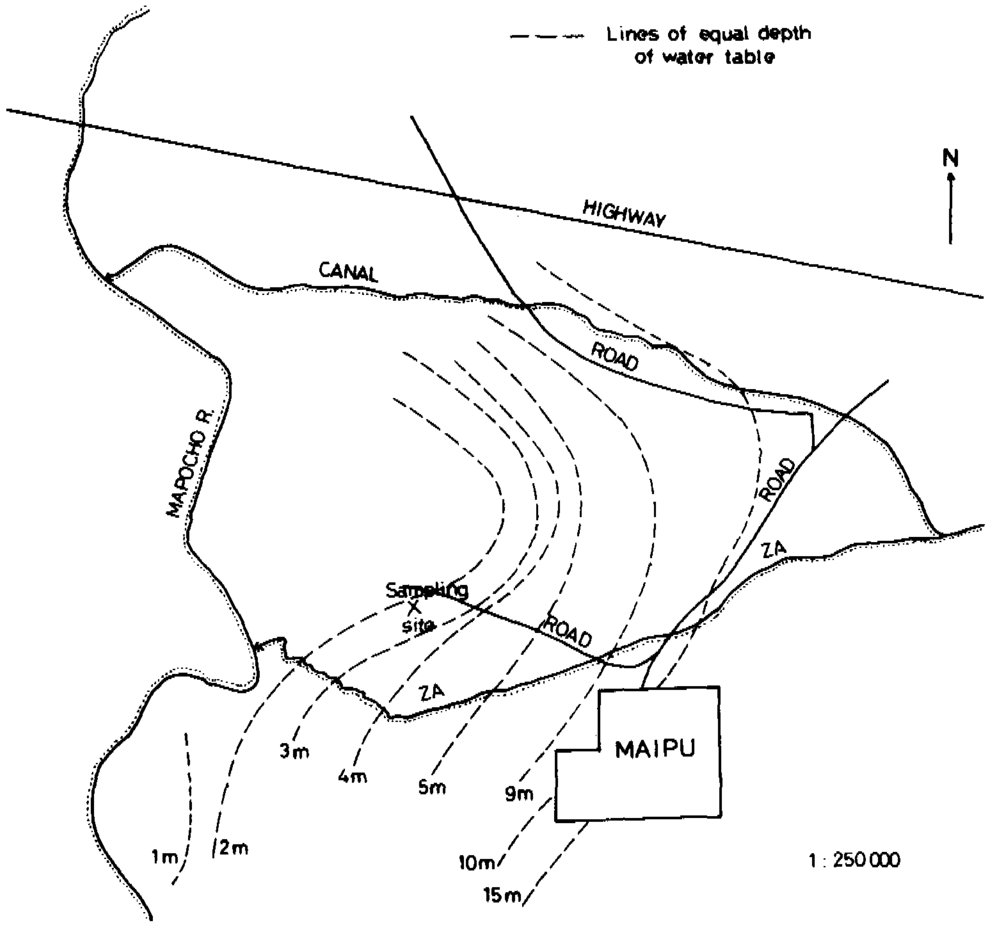

Figure 1. Map of experimental area and depth of water table

productive and from the plot under study three lettuce or two celery crops are harvested each year. Swiss chard yields four cuts per year. Irrigation averages around 2.4 m per hectare per year and is accomplished by the furrow system. No data on evapotranspiration are available.

Vegetable crops from the area supply a major part of the needs of the population of Santiago. Therefore contamination with microbial pathogens and other types of toxic substances that may be taken up by plants irrigated with ZA waters pose a potential pollution problem. This is compounded by the habit of farmers of washing their vegetables to keep them fresh looking before marketing using unfortunately ZA water. In spite of intensive efforts by health authorities to change at least the water used for washing to deep-well water made available at convenient points, the old habit persists. Since March of this year (1978), however, farmers are no longer allowed to grow lettuce in the area because of the epidemiological problems involved.

MATERIALS AND METHODS

Water analysis:

Composite samples collected in plastic containers between 10 AM and 4 PM were filtered and the filtrates and residues analyzed. COD, pH and EC were determined according to standard procedures (Jackson, 1958). Total N was determined by a modified Kjeldahl method and NO_3^--N by the phenol-disulfonic acid colorimetric method. Heavy metals in the filtrate were determined by Atomic Absorption Spectrophotometry (AA) after extraction by the APDC-MIBC method described by Brooks et al., (1967) and Koirtyohann and Wen (1973). Heavy metals in the suspended solids were determined by AA after digesting with H_2O_2 and HNO_3.

Soil analysis:

Soil samples were collected at 0.4 m intervals down to the water table, using a screw-type Auger. Field-water content was estimated by weighing fresh-

TABLE 1. Some characteristics of sewage, river and well waters studied

Source	pH	EC	COD	NITROGEN NO_3^--N	NITROGEN Total N	SOLIDS Suspended	SOLIDS Soluble
		mmhos/cm	mg/l				
ZA canal	7.8	1.71	375	0.2	38.0	310	940
Mapocho	8.4	0.96	96	0.6	3.6	137	680
Well	7.6	1.35	54	13.7	15.1	13	910
Well (control)	8.0	1.50	95	0.9	1.3	3	950

ly collected samples before and after a 24 hr air-drying period (Pratt et al., 1972). The pH of a saturation paste was determined with a glass electrode, total N by a modified Kjeldahl method, and organic C by the Walkley-Black method. Nitrates were extracted with a $CuSO_4$-Ag_2SO_4 solution (Jackson, 1958) and determined colorimetrically by the phenoldisulfonic acid method. Heavy metals were determined by AA after extraction under reflux for 12 hours at 80 C with 4 N HNO_3.

Vegetable analysis:

Aerial parts of lettuce, celery, and swiss chard were sampled just before harvest and washed, dried at 65 C and weighted. The dry samples were ashed, dissolved in HCl, and heavy metals determined by AA.

RESULTS AND DISCUSSION

Some properties of the irrigation and well-waters are given in Table 1. The electrical conductivity of the ZA and shallow well waters is moderately high, with 1.7 and 1.35 mmhos/cm respectively. The pH is slightly alkaline varying between 7.8 and 8.0. Suspended solids in the ZA waters are about 300 mg/l and total N content around 38 mg/l, mostly in the ammonia form.

The soils irrigated with ZA waters are medium to light textured (Table 2), with a pH that increases from 6.9 at the surface to 7.5 at the 150-190 cm depth. Field water content is about 20% throughout the soil profile which indicates thorough and uniform irrigation practices. Organic-C decreases from 2.15% in the surface soil to around 0.05% in the deepest part of the soil profile studied. The C/N ratios are about 13 in the surface soils and decrease to about 5 at the lower depths. Electrical conductivity of the saturation extract averages about 1.0 mmhos/cm which denotes moderate amounts of soluble salts and indicates that the soil is well leached.

An average of 8 ppm NO_3^--N on a dry soil basis is found throughout the soil profile. Inasmuch as the NO_3^--N of the ZA waters is very low (Table 1) a considerable portion of the soil nitrate probably originates through nitrification of the ammonium-N in the waters. Additional NO_3^--N could originate through mineralization and nitrification of organic nitrogenous compounds in the waters and in the surface soil. Sizable amounts of NO_3^--N are therefore available for leaching into ground waters (Schalscha et al., 1978). That this occur is confirmed by the presence of NO_3-N in shallow wells (Table 1) where

TABLE 2. Some properties of soils studied (average of 4 samples)

Depth cm	pH	Field moisture	Organic C	Total N	NO_3-N	EC	Texture*
		%	%	%	ppm	mmhos/cm	
0 - 30	6.9	21.2	2.13	0.16	7.1	0.99	SCL
30 - 70	7.2	18.8	0.62	0.06	8.8	1.02	SCL
70 -110	7.3	18.5	0.25	0.02	6.2	0.95	SL
110 -150	7.2	18.2	0.14	0.02	8.4	0.89	SL
150 -190	7.5	19.3	0.05	0.01	7.4	1.08	SL

* SCL: sandy clay loam; SL: sandy loam

TABLE 3. Heavy metals in ZA irrigation waters

Season of sampling	Cd ss[1]	Cd f[2]	Cr ss	Cr f	Cu ss	Cu f	Ni ss	Ni f	Zn ss	Zn f
	ug/l									
Spring	17	2.5	1115	11	460	7	83	97	270	26
Summer	19	0.9	2080	15	530	23	130	12	720	42
Fall	20	1.8	1020	8	390	20	99	54	250	63
Winter	21	0.7	350	3	148	11	72	30	140	49

[1]ss = in suspended solids
[2]f = in filtrate or soluble form

nearly 14 ppm have been detected which is near the limits set by the USPHS for human consumption.

Heavy Metals: The concentration of the heavy metals, Cd, Cr, Cu, Ni and Zn in the ZA waters are given in Table 3. They are found predominantly in the suspended solids except for Ni where up to 50% is present in the filtrate or soluble form.

While investigating the source of abnormal concentrations of heavy metals in ZA waters, it was found that a few industries store their waste waters containing heavy metals in special tanks thereby complying with existing regulations. However, when the tanks are filled they discharge the contents into the ZA canal selecting a time period when no inspection takes place.

The levels of Cd in the ZA waters are in the medium to high range (around 20 ug/l) but up to date no toxic effects on plant growth have been detected. Copper concentrations range from 159 to 555 ug/l and should not pose any special problem. The same is true for Ni and Zn with concentrations fluctuating respectively, between 102 to 180 ug/l and 189 to 762 ug/l, Zn being surprisingly low. Chromium concentrations vary between 353 and 2095 ug/l and these high amounts can only be explained by the above cited emptying of storage tanks from tanneries into the ZA canal. Incidentally only Cr is subject to special regulations as to the amount dischargeable.

Considering the heavy metal content of the ZA canal waters and their use in irrigation of vegetable producing farmlands, a potential hazard exist. However, yields of vegetable crops irrigated exclusively with these water are high, and no toxicity symptoms have been detected. If criteria for irrigation water quality of the U.S. Federal Water Pollution Control Administration (FWPCA) of 1968 are used, only Cd would be above tolerable levels (5 ug/l) and in some instances Cu (200 ug/l).

The concentrations of heavy metals in the soils irrigated exclusively with ZA waters are shown in Table 4. It is interesting to observe that the amounts of Cd, Cr, Ni, and Zn in the profile below the root zone are more or less evenly distributed down to 190 cm, which is just above the water table. Only Cr accumulates in the top layer, and Cu surprisingly increases with depth. On the other hand Cd concentrations in the soil range from 0.7 to 2.8 ug/g and therefore exceed in some instances the FWPCA limits. The same is true for Cu where amounts up to 1,700 ug/g of soil are found. In the root zone Cu concentrations fluctuate between

TABLE 4. 4 N HNO_3 soluble Cd, Cr, Cu, Ni and Zn in soils collected from raw sewage water irrigated farmland. (averages of 3 samplings of four replicates.

Depth cm	Cd	Cr	Cu*	Ni	Zn
	ug/g				
0 - 30	1.7	51	706(273)	15	161
30 - 70	1.3	35	767(221)	16	156
70 -110	1.5	16	1041(218)	18	155
110 -150	1.0	15	1060(105)	21	201
150 -190	1.2	16	1061(104)	20	180

* Data in parenthesis for Cu from a control soil not irrigated with ZA waters

580 and 850 ug/g, much higher than the critical level for plant growth described by Bingham et al., (1975). The abnormally high concentrations of Cu were first thought to be of geological origin. However, analysis performed on adjacent soils of the same type but not irrigated with ZA waters, showed a concentration range of 104 to 273 ug/g of soil. Therefore the source of copper could be attributed to the ZA irrigation waters. The Cu values appear somewhat inconsistent considering the amounts of the other metals found in these soils, but except for the facts mentioned there seems to be no other source of Cu. The nearby Lo Aguirre mountains contain exploitable copper ores, but they are separated from the area under study by the Mapocho river into which the ZA canal and its natural runoff empties.

Heavy metal contents of vegetable crops grown in this area are shown in Table 5. The data show leaf Cd levels of 0.1 to 2.8 ppm and Cu concentrations of 15.6 to 23.0 ppm which are relatively high (Page, 1974, Dowdy and Larson,1975). However no limitation of plant growth nor visible toxic symptoms were obsered. The Cd content of swiss chard which is considered a Cd accumulator varied from 1.1 to 1.8 ppm which is lower than the values for lettuce, namely 1.6 to 2.8 ppm. Celery contained only 0.1 to 0.4 ppm Cd, which is very low. Copper concentrations in leaves of the three crops were practically the same, around 20 ppm. Chromium levels were about 1.5 to 6.0, 0.3 to 2.5, and 0.1 to 0.3 ppm respectively for swiss chard, lettuce, and celery.

Some typical yields of vegetable crops are also presented in Table 5. Compared to yields in general throughout Chile and other countries these are high notwithstanding the fact that three crops of lettuce, two of celery, or 4 cuts of swiss chard are harvested each year. In addition no commercial fertilizers and no pesticides are applied to this area.

The observations and results summarized indicate that the fertility of the soils under investigation have been maintained by the N and other elements present in the sewage water used for irrigation, and even after 40 years or more toxic amounts of salts and heavy metals ions have not accumulated in the soil profile. It appears that salts have moved through the soil profile with the waters. It is also possible that some of the heavy metals have complexed with small molecular weight soluble organic compounds originating from the sewage waters and have moved down or through the soil in the complexed form. However, concentrations of Cu and Cd in the soil may be reaching critical levels, so it is important that industrial wastes containing these elements not be discharged into the sewage canal.

TABLE 5. Heavy metals in leaves of vegetables irrigated exclusively with ZA waters (dry matter basis).

Crops		Cd	Cr	Cu	Ni	Zn	Yield
		ug/g					Tons/ha/ys
Swiss Chard (Beta cicla, L)	1st cut - Winter	1.1	6.0	18.1	2.8	95.4	5.0
	3rd cut - Summer	1.8	1.5	18.7	1.3	70.2	
Lettuce (Lactuca Sativa)	Winter	1.6	2.5	23.0	2.1	78.1	6.7
	Summer	2.8	0.3	22.8	2.0	99.7	
Celery (Apium graveolens dulce)	Winter	0.1	0.1	15.6	3.1	99.0	13.7
	Summer	0.4	0.3	20.2	2.1	95.9	
Celery, stalks	Winter	0.5	0.7	9.1	2.3	45.3	
	Summer	0.2	0.4	9.5	1.5	50.3	

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PRETREATMENT REQUIREMENTS FOR LAND APPLICATION OF WASTEWATER

PREAPPLICATION STRATEGIES FOR WASTEWATER IRRIGATION SYSTEMS

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ABSTRACT

The preapplication approaches that should be used with wastewater irrigation systems are those that reduce risks to the public, the environment and the equipment, and permit the pollutant removal mechanisms in the soil to renovate the wastewater. The use of the limiting parameter concept in design, specific application and site management, and the following preapplication approaches should result in reliable irrigation systems that have extremely low health and environmental risks. The preapplication approaches for wastewater irrigation systems are: storage, odor control, removal of large solids, and possible pretreatment to reduce abnormal levels of nitrogen, metals, pH and SAR.

INTRODUCTION

Land treatment of wastewaters can be a cost effective treatment process for many communities and industries. Although land treatment can be considered for untreated wastewaters, it generally is utilized as an alternative to secondary or tertiary treatment processes.

There are three basic land treatment processes: a) slow rate, b) rapid infiltration, and c) overland flow. This paper discusses the preapplication strategies for the slow rate process. This is a process in which vegetation is a critical component for managing water and nutrients, and the water application rate is low, up to about 4 inches (10 cm) of wastewater per week. The wastewater is applied by sprinkler or surface methods. Slow rate systems also are referred to as irrigation systems; however, the emphasis is on wastewater treatment rather than on crop irrigation.

Many advantages can be associated with land treatment such as: a) reclamation and recycling of the nutrients and water, b) reducing the discharge of pollutants to surface waters, and c) favorable energy requirements and economics. While land treatment is not a panacea to be used in every situation, it can be a viable treatment alternative that should be adequately considered when the most appropriate treatment process is determined.

The potential of land treatment has been recognized in federal water pollution control legislation. Evaluation of land treatment in waste treatment facilities planning has been mandatory since July 1974. A detailed assessment of one or more land treatment systems is required as part of the Environmental Protection Agency (EPA) construction grants program. EPA also has indicated that any preapplication requirements shall not be unnecessarily stringent. Thus it is appropriate to consider the types of preapplication methods that are commensurate with the proper design and management of a slow rate system.

This paper considers only preapplication strategies for the treatment of wastewaters by the slow rate process. It does not address the requirements that may be necessary for subsequent use of the renovated wastewaters such as for a public water supply.

RISK

There have been philosophical differences regarding the levels of pretreatment that are needed before wastewaters are applied to land. One view is that the equivalent of secondary treatment, including possibly chlorination, is necessary before the treated wastewaters are applied to the land. This view infers that land application of wastewater is a disposal mechanism that does not include wastewater treatment capabilities. Another view is that only a minimum amount of preapplication is necessary, i.e., that which minimizes nuisance conditions, increases system reliability and reduces the risk to the public.

The second view recognizes that the soil has a capability to assimilate waste and renovate wastewater. The first view indicates a preference to have the control of wastewater treatment in the hands of man rather than depend upon the capacity of the soil. Both views represent conscientious attempts to minimize the risks to the public and the environment.

Human activity will always and unavoidably involve risks and there can be no hope of reducing all risks to zero. These statements are valid for all aspects of waste management including land treatment. Because safety is a judgemental decision, a technology such as slow rate land treatment will be judged safe if its risks are judged to be acceptable by the public. Whether something is "safe" can not be measured, because the physical and biological sciences can assess only the probabilities and consequences of events, not their value to people.

The many concerns about acceptable risk can be summarized by four basic considerations (1):

1) definition of conditions of exposure, i.e., who or what will be exposed? to what? in what way? for how long?

2) identification of adverse effects, i.e., what is the threat or adverse effect to the individuals or objects that are exposed?

3) relationship of exposure with effect, i.e., how much adverse effect results from how much exposure?

4) estimation of the overall risks

The need for preapplication treatment is related to reducing the risks to equipment, to the public, and to the environment. Judgements concerning the risks associated with using or not using certain preapplication methods must be included in assessing the preapplication methods that should be used with slow rate systems.

GENERAL GUIDANCE

In identifying preapplication methods that may be needed, there are several criteria that are basic:

- there shall be no public health hazard
- applicable groundwater quality standards shall not be contravened
- applicable water quality standards shall not be contravened in any adjacent surface waters

In addition, there will be different preapplication needs depending upon the degree of public access to the land treatment site, whether the crops are for human consumption, and if so, whether they are to be eaten raw. The need for preapplication methods also is related to the ability of the soil to remove contaminants and renovate the wastewater. This ability, in turn, is a function of the pollutant removal mechanisms in the soil.

POLLUTANT REMOVAL MECHANISMS IN THE SOIL

General

Any wastewater treatment system, including a land treatment system, is designed to convert raw sewage into an acceptable effluent and to dispose of the solids removed in the system. The basic approach is to determine the characteristics of the untreated waste and to utilize the capabilities of various treatment processes to achieve the desired effluent quality. With conventional wastewater treatment systems, few constraints are placed upon the input to intermediate processes except where such constraints are necessary to protect these processes from overloading or from breakdown of mechanical components. The same approach should be taken with design of land treatment systems.

With wastewater irrigation, the soil is the important treatment process and its capability to remove pollutants should be utilized. There should be neither over design of the slow rate system nor overloading of the soil.

The design relationships for irrigation of wastewaters are governed by the soil and subsurface conditions, the climate, the availability of land, the desired quality of the renovated wastewater, the crops to be grown on the land, and the expected management of the system. Included in the design will be the controlling limiting parameter and the preapplication possibilities that permit the site to be managed satisfactorily.

The effectiveness of a wastewater irrigation system is related to the characteristics of the soil and the resultant pollutant removal mechanisms. When wastewater is applied to the soil, some constituents may pass through the soil to the groundwater, others are utilized by growing plants, some are metabolized by the soil microorganisms, and others are retained within the soil. The design of a wastewater irrigation system, and therefore the need for preapplication methods must relate the quantity of pollutant in the wastewater to the pollutant removal mechanisms in the soil. These can be categorized as physical, such as filtration and dilution; chemical, such as adsorption and precipitation; and biological, such as microbial transformations and plant uptake. The need for preapplication can be determined by evaluating the possibility of overloading the removal mechanisms.

Physical

As wastewater moves through the soil pores, suspended solids are removed by filtration. The depth at which removal occurs varies with the size of the particles, soil texture, and rate of water movement. The larger the hydraulic application rate and the coarser the soil, the greater the distance the particles will move. However, at the wastewater application rates used with the slow rate process, large suspended solids are removed in the surface soil and smaller particulates, even as small as bacteria contained in the wastewater, are removed in the upper few feet of all but the very coarse soils.

Constituents in the applied wastewater can be diluted by rain and snow melt. Chemical and biological transformations and removals in the soil also can reduce concentrations of specific constituents. Where evaporation losses are high, such as in arid climates, increases in the concentration of conservative constituents, such as salts, can occur.

Excessive suspended solids can clog the soil pores as well as clog distribution systems. Clogging of the soil will reduce the soil infiltration rate. Natural decomposition of the organic solids during nonapplication or "resting" periods will allow the infiltration rate to recover.

Application of two inches of primary municipal effluent to the soil will result in the application of about 45 pounds of suspended solids per acre (40 kg/ha) per application. With wastewater irrigation systems, the wastewater is applied intermittently, i.e., once or twice a week. Thus a two to four inch application rate will add about 45 to 90 pounds SS/acre (40 to 80 kg/ha). This is less than 0.1 ounce/ft^2 (28 g/m^2), an amount well within the absorption and assimilation capacity of the soil.

The design hydraulic application rate for a wastewater irrigation system will be less than the infiltration rate of the soil. At this application rate, soil clogging due to suspended solids will not be a significant problem. Thus preapplication methods for suspended solids should be limited to methods such as screening or primary sedimentation that will avoid clogging of the irrigation distribution equipment or avoid excessive wear of pumps and piping.

Chemical

Chemical reactions in the soil affect the mobility of dissolved ions or compounds with the result that some constituents are retained within the soil profile for extended periods of time while the movement of others may only be temporarily restricted. Liquid residence times for normal wastewater irrigation rates are on the order of weeks, with organic residence times on the order of years.

Adsorption and chemical precipitation are the most important chemical reactions governing the movement of constituents in the irrigated wastewater with cation exchange being the most important adsorption phenomenon. The cation exchange capacity (CEC) of soils can range from 2 to 60 meq/100 grams of soil (2) with most soils having a CEC value between 10 and 30. The differences occur because soils vary widely in their humus and clay content, the components that have the highest CEC.

Typical soils have considerable capacity to adsorb many of the cations in wastewater, including many of the metals which may adversely affect the health of humans and animals eating the crops grown on the irrigated fields.

Cation exchange of ammonium nitrogen is a possible control mechanism for nitrogen. However, the ammonium ion is biologically oxidized to nitrate in aerobic soils. Nitrate is an anion and will move with the soil water.

Phosphate is the only anion appreciably retained in soil. The primary mechanism is the formation of insoluble or slowly soluble precipitates.

In arid regions, wastewater irrigation rates may not be enough to avoid the accumulation of sodium ions in the soil. Such accumulations can lead to a degradation of soil structure and a reduction in infiltration and percolation rates.

When industrial wastes are included in the wastewater to be irrigated, adjustment of the pH and the sodium adsorption ratio (SAR) may be needed. The wastewater to be applied should have a pH within the range of 6.0 to 9.5 to avoid adverse effects to site vegetation. Wastewaters with high sodium adsorption ratios must be accompanied by special soil management procedures to compensate for the effect of the sodium. The SAR of wastewaters used for irrigation should be no more than 8 to 10.

One example may illustrate the type of preapplication methods that may be necessary with specific wastes. Before the irrigation of cannery wastes, pretreatment was necessary to remove the coarse organic solids and to adjust the pH and SAR (3). Coarse screens were used to remove the large solids, and the SAR and pH were adjusted with gypsum. No further preapplication was used even though the organic content of the applied wastewater was very high.

With normal wastewater irrigation rates and suitable management, the chemical mechanisms in the soil control wastewater constituents of concern such as phosphorus and certain potentially toxic elements and chemicals. The preapplication methods that may be related to the chemical reactions in the soil could include:

a) pretreatment controls, such as industrial source control or chemical precipitation, if the amount of potentially toxic elements and chemicals in the wastewater are likely to exceed the chemical removal mechanisms in the soil.

b) adjustment of wastewater pH and SAR to acceptable levels.

Biological

The biological transformations that occur in the soil include organic matter decomposition and nutrient assimilation by plants. These transformations occur in the biologically active upper few feet of the soil, i.e., the rooting zone. The numbers of bacteria are large ranging from one to three billion per gram of soil. One thousand pounds of live bacterial mass per acre two feet of soil is probably a modest estimate (2). The great diversity of native organisms enhances the capability of a soil to degrade the variety of natural and man-made organic compounds in the applied wastewater.

The presence or absence of oxygen in the soil has a significant effect on the rate and end-products of degradation. The oxygen status of the soil is a function of soil porosity. Soil properties that favor rapid infiltration and transmission of the applied wastewater also favor oxygen movement. Low and intermittent wastewater application rates used with irrigation systems favor aerobic conditions, rapid organic matter decomposition, and oxidized end-products.

As a result of organic matter decomposition, elements such as nitrogen, phosphorus, and sulfur are converted from organic to inorganic forms. Many of these mineralized constituents can be assimilated by plants. Crops are an integral part of the slow rate process.

Organic matter is added continuously to soils as plant residue and is continuously oxidized by the soil organisms. In general, soils have a high capacity for organic carbon oxidation. Data reported for an irrigation system treating vegetable processing wastes indicate that organic oxidation rates could exceed 5000 pounds per acre per day (4110 kg/ha/day) (4). In contrast, a two-inch application of municipal primary effluent would add about 90 pounds of BOD per acre (80 kg/ha) per application which would amount to the addition of 90 to 180 pounds BOD/acre/week (80 to 160 kg/ha/week) assuming a once or twice per week application. Based upon this comparison, it is unlikely that the organic oxidation capacity of the soil will be exceeded in a slow rate process, and therefore no preapplication requirements will be needed for this parameter.

The biological processes in the soil form nitrate from ammonia and organic nitrogen under aerobic conditions. Nitrogen oxides and nitrogen gas result from nitrate reduction when aerobic conditions are followed by anaerobic conditions. It is possible to consider both gaseous nitrogen losses (volatilization and denitrification) and nitrogen removal by plant uptake as control mechanisms for the nitrogen in the applied wastewater.

Crop selection and management are important components of a wastewater irrigation system. Plant uptake of nitrogen is in the range of 100 to 400 pounds per acre per growing season, depending upon specific crop and management techniques.

The nitrogen application rate should be determined from a nitrogen balance on the system (5). The important processes involved in nitrogen removal from wastewater applied to the land are ammonia volatilization, crop uptake and removal, soil adsorption of ammonium, incorporation into the soil organic fraction, and denitrification. With the slow rate process, nitrogen management is principally due to crop uptake with some denitrification. The proper application rate will be that which, when crop uptake and denitrification are considered, maintains the nitrogen concentration in the percolating water below allowable limits, generally less than 10 mg/l nitrate nitrogen.

Denitrification losses can range up to 50% depending upon how the irrigation site is managed. A conservative estimate would be to assume denitrification and volatilization losses to be 20 to 25% of the applied nitrogen. Consideration of plant uptake and these losses as the nitrogen control mechanisms will reduce the risk of excessive nitrogen in the percolate.

Preapplication approaches related to biological mechanisms could include nitrogen removal prior to irrigation where the nitrogen application rate is the limiting factor and the required land area is excessive. Such removal could include nitrification followed by denitrification in preliminary wastewater storage ponds.

LIMITING DESIGN PARAMETER

There are many factors that determine the land area that will be required for a wastewater irrigation system. These factors are related to the characteristics of the soil, climate, wastewater, and crop and should be evaluated using site specific information.

The application rate of the following parameters will significantly affect the required land area: a) water, b) organics, c) nutrients, d) potentially toxic elements, and e) salts. When evaluating the required land area, the land area for each potentially limiting parameter should be determined. That parameter which requires the largest land area to avoid environmental problems becomes the limiting parameter. This "limiting parameter principle" states that the design land area shall be no less than that allowed by the limiting environmental parameter.

Figure 1 illustrates the concept of the limiting design parameter. In the example, nitrogen is the controlling design parameter as is the case in most land treatment systems treating municipal wastewaters.

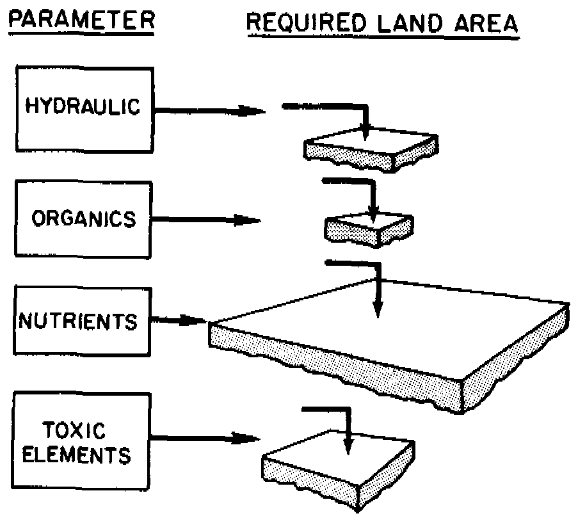

Figure 1. Relationships Between Waste Constituents and Potentially Limiting Design Parameters for Land Application Systems.

When the land area determined for the limiting parameter is used for the design of the irrigation system, there is an added degree of safety in terms of the application rates of the other constituents of potential concern. The application rate of another constituent will be considerably less than the rate that would occur if the constituent would be the limiting parameter. In addition, because of the conservative approach using plant uptake and gaseous nitrogen losses to determine the nitrogen loading rate and hence the maximum land area, there is even less risk associated with the application of the other constituents in wastewater.

The basic concept inherent in the limiting parameter approach is to use the site specific characteristics of the soil and the wastewater to meet the desired "effluent," i.e., groundwater quality, and calculate the loadings accordingly. The concept is to use soil loading criteria and not specific preapplication treatment criteria for the design of a land treatment system.

HEALTH

Protection of the public from pathogens is a frequently cited reason for preapplication requirements. Because human pathogens must be assumed to be in municipal wastewaters, concern must be given to the potential health hazard associated with wastewater irrigation systems. Irrigation systems have the potential to transmit pathogens with the percolate to the groundwater, with surface runoff, with aerosols which may be generated, and with the crops grown on the irrigation site.

Numerous critical reviews have assessed the health risks associated with the land application of wastewaters and sludges (5-9). Pathogenic bacteria and viruses in applied wastewater and sludge are the organisms of greatest concern.

Bacteria are the most fragile of the pathogens and are greatly reduced in wastewater storage systems and by sunlight, drying, or competition in the soil. Contamination of plant surfaces can occur by direct contact and by rain splashing, but survival is only on the order of a week or two (6). Bacteria can live longer if protected from sunlight and drying.

Viruses may persist in soils and on vegetation for several weeks or months. If exposed to sunlight and drying, viruses eventually will be inactivated.

Although the reviews agree that the epidemiological evidence is not abundant, they also indicate that little health risk is associated with well-managed land application systems for sludges and wastewater. The key is a "well-managed system" since reported disease outbreaks have been associated with the use of night soil and raw sludges, when crops grown on soils that have received raw sewage were eaten raw, or when grazing cattle have been contaminated by raw sewage. Percolation of wastewater at a moderate rate through medium to fine texture soils removes bacteria and viruses in a short distance due to physical entrapment, adsorption, and die-off.

Disinfection of wastewater by chlorination prior to irrigation has been suggested but does not appear necessary if sound application and management procedures are followed. Reliance on preapplication disinfection rather than proper application rates and site management procedures can provide a false sense of security. Management procedures to minimize health risks are noted in Table 1.

Table 1. Wastewater Application and Site Management Procedures to Minimize Health Risks

- restrict public access to the site
- wastewater application rates that are consistent with limiting parameter relationships
- prevent runoff from the site
- crop production restricted to those not eaten raw
- establishment of buffer zones
- low trajectory, low pressure spray distribution systems to minimize aerosol drift
- surface distribution irrigation systems to avoid aerosol drift
- use of vegetation that does not have a high accumulation of toxic elements and chemicals

As Wolman (9) notes, the application of wastewater to land is a practicable method of wastewater disposal provided that "it is carefully, efficiently, and continuously managed." Under such conditions, the public health problems associated with wastewater irrigation should be minimal. The contribution of health risks to the general public should be extremely small, especially if the public has limited contact with the irrigation site. To protect the public from waterborne diseases, it is more appropriate to collect and disinfect the renovated groundwater prior to public use rather than chlorinate the wastewater applied to the land.

Based upon the lack of specific evidence that there is a serious health risk associated with well-managed wastewater irrigation systems, and the fact that there are a number of management approaches to minimize potential health risks, it is not possible to justify the ubiquitous use of chlorination as a preapplication requirement.

SYSTEM RELIABILITY AND MANAGEMENT

The above sections discuss preapplication strategies based upon pollutant removal mechanisms in the soil and reduction of health hazards. There are preapplication methods that are necessary or desirable for consistent system reliability or sound system management. These include: storage, odor control, and prevention of distribution system clogging.

Disposal of wastewater on land in cold climates may not be practiced continuously because of cold weather operating problems and because crop uptake for nitrogen control is minimal in the winter. Storage of the wastewater during such periods is desirable. In addition, storage will be necessary when natural precipitation prevents application of the wastewater. Storage lagoons or ponds can provide intermediate treatment of the wastewater through biological action, solids deposition and pathogen reduction. Thus wastewater storage has benefits beyond holding of the wastewater during periods when the land can not be irrigated. The quality of the storage lagoon effluent may approximate that of secondary treatment effluent.

Since wastewater can become odorous when stored, the storage lagoons must be designed to maintain aerobic conditions. Such ponds and lagoons also serve to equalize wastewater flows and loads and provide flexibility to the operation of the irrigation system.

With a slow rate system, the grit, organic solids, and fats and grease in the wastewater may adversely affect distribution system components and performance. This material can be removed in storage ponds or by separation units such as screens prior to application to the land.

Thus there are a number of physical and mechanical preapplication methods that can be utilized to enhance the reliability and management of slow rate wastewater treatment systems.

SUMMARY

The purpose of a land treatment system is to use the soil mantle as a reactor capable of treating contaminants and renovating wastewater. Wastewater irrigation is a treatment process in which the soil is the basic unit process with the physical, chemical, and biological mechanisms, including the vegetation, as the important components in the process. With wastewater irrigation, the applied wastewater does not have to have a quality equivalent to that which would be permitted for stream discharge. There are numerous examples of slow rate systems treating primary treated municipal sewage and producing percolate water that is virtually free of organics, pathogens, and toxic elements and chemicals. The soil has a capacity to treat contaminants in wastewater and the capacity should be utilized.

The sound design of a well-managed wastewater irrigation system will be based on the limiting parameter principle, i.e., that parameter which will require the largest land area to be utilized. Each site will have a specific limiting parameter that will be based upon the waste characteristics, soil characteristics, topography, climate, crops to be grown, and the management to be utilized. Generally the soil is capable of renovating any wastewater containing any type of pollutant. In extreme cases and with no pretreatment, certain pollutants, such as toxic elements and chemicals may have to be applied at such a low application rate that land treatment would not be a cost-effective method.

However, with the constituents in typical municipal wastewater, slow rate systems can be cost-effective, and the limiting parameter approach reduces the risk of adverse effects due to contaminants in the wastewater.

An important part of the planning and design of a slow rate system is knowledge of the constituents in the wastewater to be applied. If the characteristics of the wastewater are not known in adequate detail, a wastewater analysis program should be initiated.

When a wastewater has characteristics that may be detrimental to the equipment, the public health, the soil, or the crops--such as excessive grit or organic solids, a high SAR or pH, potentially high concentrations of toxic compounds, or high numbers of human or animal pathogens--the adverse effect of wastewater irrigation should be evaluated carefully prior to full scale design and operation. Preapplication methods that will reduce the parameters of concern should be used.

Preapplication methods can enhance the capability of a wastewater irrigation system to function reliably and continuously and minimize health or environmental risks that are associated with the system. Based upon consideration of the removal mechanisms that occur in the soil and the management approaches that can be utilized, certain preapplication strategies for wastewater irrigation systems can be identified and are noted in Table 2. Two types of preapplication requirements exist: a) those needed to assure that the soil can function as a treatment process, and b) those needed for other purposes such as odor control and storage.

Table 2. Preapplication Strategies for Wastewater Irrigation Systems

A. Strategies to sustain the capability of the soil as a treatment process

- pretreatment to reduce excessive amounts of nitrogen and potentially toxic elements and chemicals and to adjust abnormal pH and SAR values in the wastewater to be applied

B. Strategies to avoid nuisances and maintain system reliability

- wastewater storage
- odor control
- removal of large solids to avoid distribution system wear and clogging

The fact that the potential health effect of wastewater irrigation has not been completely resolved is cause for using conservative management and design procedures but should not preclude the use of wastewater irrigation systems where they are cost effective. It also should not result in the use of unnecessarily stringent preapplication requirements.

Satisfactory wastewater irrigation systems do not just happen. They are carefully designed and receive sound and continuous management. The use of the limiting parameter concept in design, the use of management procedures such as those noted in Table 1, and the incorporation of the preapplication strategies noted in Table 2 should result in wastewater irrigation systems that can be cost effective, reduce risks to the public, the environment, and the equipment, and can meet accepted surface and groundwater quality.

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PRETREATMENT REQUIREMENTS FOR LAND APPLICATION OF WASTEWATER

PRETREATMENT REQUIREMENTS BEFORE LAND APPLICATION OF MUNICIPAL WASTEWATER

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Abstract

The equivalent of primary treatment usually will be needed as minimum pretreatment to remove suspended solids and heavy metals before land application of wastewater. Pathogen populations are not greatly reduced by primary treatment but removal of suspended solids could reduce pathogen survival time on crops and soils. Suspended solids removal also removes most of the heavy metals. Secondary treatment minimizes odors by stabilizing the wastewater, and removes some pathogens but leaves relatively high pathogen populations in the effluent. Disinfection of secondary effluent removes most of the bacterial pathogens, helminths, and protozoa. However, removal of most of the turbidity by advanced treatment is required before disinfection will reduce the virus populations to low levels. Conventional treatment methods leave most of the nutrients in wastewater while lagoons can be managed for extensive nutrient removal. Tertiary treatments remove nutrients but will be too expensive for pretreatment for most land application systems. In many cases, treatment beyond the primary level will not be required before land application, but the exact degree of pretreatment needed is highly site specific and depends upon the wastewater, soil characteristics, climate, degree of human exposure, and intended use or disposition of the renovated water.

Introduction

Land application of municipal wastewater offers a unique opportunity to simultaneously recycle valuable resources and reduce the pollution of our water resources. However, since wastewaters contain pathogens, nutrients, and heavy metals which can be health hazards or pollutants, land treatment systems must be carefully designed and managed to provide safe effective treatment. Municipal wastewater is usually treated somewhat before land application. Thus land application is practiced as part of a total system that may include other treatment methods before or after land application or both. Usually, most of the other treatment precedes land application.

Pretreatment may be needed to minimize health hazards by removing pathogens, to promote infiltration by removal of suspended solids, to prevent overloading the system by removing nutrients or to make the system more aesthetic by reducing and stabilizing organic loads. The degree of pretreatment needed is determined by the capacity of the land treatment system to remove the various components mentioned above. Therefore, the pretreatment needed is site specific. In this discussion, we will focus on principles that can be used to assess the degree of pretreatment needed for various wastewaters and soils.

Pathogen Removal

The large variety of pathogens present in domestic wastewater includes pathogenic bacteria, viruses, protozoa, and parasitic worms. These pathogens may reach humans by vegetables contaminated by wastewater used for irrigation, by farm workers working in contaminated crops or soils, by farm workers or residents in the area exposed to aerosols containing pathogens or by pathogens moving through the soil into groundwater supplies. Some of these hazards can be alleviated by pretreatment, but management practices and exclusion of certain uses are often more practical than pretreatment.

Foster and Engelbrecht (15) assessed the pathogen removal efficiency of conventional sewage treatment systems (Table 1). In general, primary treatment reduces the populations of helminth ova but has little impact on numbers of bacteria or amoebic cysts. Virus removal is probably significant because most viruses are associated with suspended particles, although some laboratory studies have failed to demonstrate significant virus removal. Thus, primary treatment removes only those pathogens heavy enough to settle out of the water or associated with suspended solids. However, primary treatment may have a significant impact on pathogen survival on crops and soils due to removal of suspended solids which protect pathogens from adverse conditions.

Activated sludge treatment results in about a 99% removal of bacteria and viruses but little additional removal of amoebic cysts or helminth ova. Removal by trickling filters can be more efficient with up to 99.9% removal of bacterial pathogens and amoebic cysts and up to about 75% removal of helminth ova and viruses.

The infective dose (the number of organisms necessary to cause disease in healthy humans or animals) should be considered when evaluating the treatment

Infective doses for most bacterial and protozoan pathogens are relatively high. For example, ingestion of 10^8 enteropathogenic Escherichia coli or Vibrio cholerae, 10^4 to 10^9 Salmonella, and 10 to 10^0 Shigella organisms are necessary to cause infection in man (21). The infective dose of a protozoan, like Entamoeba histolytica, is believed to be as high as 20 cysts (15). The infective dose for viruses ranges from 1 to 10^0 or more (33, 48). Thus because of the low infective dose, viruses must be kept at relatively low concentrations in reclaimed water or groundwater.

Foster and Engelbrecht (15) also estimated the number of pathogens remaining per liter of wastewater after conventional treatment (Table 2). Usually the infective dose is equal to the pathogen population of at least a few liters of secondary effluent and very large volumes of disinfected secondary effluents. Since virus is a notable exception we will discuss the removal of those pathogens in more detail.

Virus Removal by Different Sewage Treatments - Infectious viruses are always present in raw domestic wastewater and appear at some level in the discharges of conventional sewage treatment plants. The concentration of viruses in raw wastewater depends on many factors like the time of year, hygienic conditions in the community, per capital consumption of water, etc. The concentration of enteric viruses has been reported as high as 500,000 PFU per liter (9) in wastewater in some parts of the world. In the United States, the expected average concentration of enteric viruses has been calculated to be about 7,000 PFU per liter in raw sewage (44). Considering the apparent low infectious dose

Table 1. Removal of Pathogens from Sewage Water by Conventional Treatment Systems[a]

Pathogen	Primary	Activated Sludge	Trickling filter
	%	%	%
Salmonella	15	96 to 99	84 to 99.9
Mycobacterium	48 to 57	slight to 87	66 to 99
Amoebic cysts	No removal in 3 hr.	No apparent removal	11 to 99.9
Helminth ova	72 to 98	No apparent removal	62 to 76
Virus	3 to extensive removal	76 to 99	0 to 84

[a]Adapted from Foster and Engelbrecht (15).

Table 2. Estimated Concentrations of Wastewater Pathogens[a]

Pathogen	Number of Organisms/L			
	Untreated Wastewater	Primary Effluent	Secondary Effluent	Effluent Disinfected[b]
Salmonella	5.2×10^3	2.6×10^3	1.3×10^2	1.3×10^{-1}
E. histolytica	4.0×10^0	3.5×10^0	3.2×10^0	3.2×10^{-3}
Helminth ova	6.6×10^1	6.6×10^0	1.3×10^{-1}	1.3×10^{-4}
Mycobacterium	3.8×10^1	5.2×10^1	4.0×10^0	4.0×10^{-3}
Human enteriovirus (poliovirus,etc.)	1.0×10^4	5.2×10^3	5.2×10^2	5.2×10^1

[a]Adapted from Foster and Engelbrecht (15)
[b]Conditions sufficient to yield a 99.9% kill.

of enteric viruses (17), removal processes must be capable of achieving large reductions of virus before direct reuse.

Viruses are removed from water and wastewater physically and by inactivating or destroying the virus. Processes that remove viruses include sedimentation, adsorption, filtration, coagulation and precipitation. Viruses are inactivated by high pH, chemical oxidation by disinfectants like halogens, ozone, and photo-oxidation by certain dyes and ultraviolet light (45, 49). Processes that inactivate viruses are preferable to those that simply remove them since removal presents a problem of the disposal of potentially infectious materials.

There are several shortcomings to the evaluation of wastewater treatment plant methods. For example, the number of viruses entering a sewage treatment plant during a 24-hr period varies greatly, making temporal coordination of samples for treatment effectiveness difficult. Seeding experiments have been used to overcome this problem, but they too may yield misleading data because many viruses naturally present in sewage may be deeply embedded in or absorbed to suspended solids (3,4,30).

Primary treatment - Primary treatment of wastes, usually involving only settling and retention before discharge, seems to remove few viruses from sewage (10). Unfortunately, obtaining a true indication of the number of viruses removed by this process is difficult since there is no accurate method to assess the number of viruses embedded in and adsorbed on fecal material and other solids. Like most field studies, experiments with seeded poliovirus type 1 indicated little removal during primary settling (3). Also, settling in Imhoff tanks reportedly did not result in significant virus removal(3). Berg (3), however, believes that since many of the viruses present in raw sewage are probably solid-associated, long settling times of 12 to 24 hr should result in a considerable amount of virus removal. In any event, it is certain that large numbers of virus remain suspended in sewage after primary sedimentation.

Secondary treatment - The three most widely used biological processes for secondary sewage treatment are trickling filtration, activated sludge and oxidation ponds. Again, virus removal during these processes seems to depend largely on virus adsorption to solids (1,3,41), although biological antagonism may also be a factor (3,41). Trickling filter removal varied from 16 to 100% (3). Sherman, *et al.* (39) recently reported average virus removal of 9.0 and 18.9% through the trickling filter beds at two different plants. Berg (3) stated that viruses passing through trickling filters simply do not make good contact with adsorptive surfaces. However, rotary-tube trickling filters seem to be more efficient. Clark and Chang (11) reported 59 to 95% removal for three different enteroviruses in a bench-scale, rotary-tube trickling filter. Removal depended on the type of virus and the filtration rate. At medium filtration rates, virus removal approached that of the activated sludge process; high-rate filtration significantly reduced virus removal.

Enterovirus removals of 90% or greater have been observed during activated sludge treatment of sewage (1,3, 15,17). Results of several studies indicated that the efficiency of a treatment plant is closely related to the concentration of the mixed liquor-solids and its capacity to remove them (1,44). Clarke *et al.* (10) showed that Coxsackievirus A9 was removed much more effectively than poliovirus, although removal of both viruses was always greater than 80%. This indicates that each enterovirus will probably have different removel rate characteristics in the activated sludge process (44). More recently, Farrah *et al.* (14) found that rotaviruses, the major agent of gastrointestinal illness in children, adsorbed poorly to activated sludge flocs as compared with poliovirus type 1. On this basis, they speculated that wastewater treatment processes that are highly effective in the removal of enteroviruses may not be as effective in removing other viral groups, such as rotaviruses. Most field studies appear to confirm that a properly operated activated sludge treatment plant removes as high as 90% of the viruses, but little or no virus removal is not unusual in plants that are not operating efficiently (3).

Removal of viruses by waste stabilization ponds is erratic, and few studies have been reported. Shuval (40) reported virus removals ranging from 0 to 96% in ponds with a 20-day retention time. Malherbe and Strickland-Cholmley (31) found highly variable virus removals in a pond with a 19-day retention period, but they noted short-circuiting which may account for much of the great variability generally experienced with removal of viruses by stabilization ponds (3).

The time of year may influence the efficiency of virus removal from oxidation ponds. Nupen *et al.* (32) reported greater removals in the summer, when algal growth was abundant, than in the winter. Removals as high as 99.9% were achieved but viruses were still routinely detected in the pond effluent.

Tertiary treatment - The tertiary treatment of wastewater usually involves some type of physical-chemical processing, like coagulation with alum, lime, iron salts, or polyelectrolyte and/or passage through activated carbon or resins to remove residual organics. Coagulation seems to be a highly effective method for the removal of enteroviruses from wastewater. In laboratory studies, removals to 98% removals of poliovirus type 1 were obtained by precipitation of the phosphate in domestic wastewater, using either alum or calcium hydroxide (8). In pilot plant studies at an advanced wastewater treatment plant, 63% of seeded poliovirus type 1 was removed during alum treatment and greater than 99.7% during lime treatment of activated sludge effluent (49). Sattar *et al.* (35) found in laboratory studies that 99.99% or more of added poliovirus could be removed from raw sewage by lime treatment at pH 11.5 for 1 hour. In experiments using seeded virus, Nupen *et al.* (32) reported 3.7 to 4.5 log reductions during lime treatment with a retention time of 60 minutes. Virus removal was directly related to retention time and pH. Most enteric viruses are inactivated at high pH. However, these same investigators were always able to recover naturally occurring viruses in 1-l samples, even after exposure of the wastewater to pH 11.5 for 180 minutes. Thus, while seeded laboratory-grown virus studies indicate large reductions during lime treatment, naturally occurring viruses may be more resistant to high pH and not as efficiently removed.

Compared with metal salts, commercially available polyelectrolytes have only minor significance in virus removal from wastewater, but they do increase the effective range of virus removal by alum and iron salts. Shelton and Drewry (28) found only a 70% reduction of seeded f2 bacteriophage from wastewater by polyelectrolytes, but greater than 90% removal with metal salts.

Carbon adsorption of residual organics is often practiced in advanced wastewater treatment facilities, but while viruses are adsorbed to activated carbon, its adsorption capacity for viruses is limited (3,16), and breakthrough occurs after the passage of a few bed volumes (16).

The complete physical-chemical treatment of wastewater has received increasing interest as an alternative to the conventional biological treatment of wastewater. Sobsey *et al.* (42) studied the removal of poliovirus type 1 from a packaged physical-chemical treatment plant consisting of chlorination, activated carbon adsorption, alum flocculation, and vacuum filtration using diatomaceous earth as a filter aid. This system was capable of removing an average of 99.95% of the influent virus from raw sewage. The degree of virus removal in such systems generally seems to be superior to that of conventional primary and secondary wastewater treatment.

Disinfection - No sewage treatment process is principally designed to affect pathogen removal, except by disinfection. The sole purpose of disinfection is to reduce the level of any remaining pathogens to negligible levels from the public health standpoint. While most disinfectants are highly effective in removing bacteria from wastewater, far greater difficulty is encountered in the inactivation of viruses. The most commonly used disinfectant for wastewater is chlorine, although the use of other halogens, ozone, UV light, and photodynamic oxidation is currently being investigated (18,45).

Chlorine is an effective, rapid virucide in clean water devoid of ammonia and organic compounds and at neutral pH levels, where it exists as highly virucidal hypochlorous acid (4). However, in sewage effluents chlorine combines rapidly with ammonia to form chloramines and other forms of combined chlorine that have very small capacities for virus inactivation. In general, enteric viruses seem more resistant than bacteria to chlorine, although the sensitivity varies considerably by species, type, and even strain (45). One recent concern has been the demonstration that exposure of poliovirus type 1 to repeated sublethal doses of chlorine resulted in the surviving viruses becoming progressively more resistant to chlorine inactivation (2). Liu and McGowan (29), in laboratory studies using 25 different human enteric viruses, found that the time required for 99.99% inactivation varied from 2.7 to 120 minutes when viruses were exposed to 0.5 mg/l free chlorine in river water. Application of 8 mg/l chlorine to secondarily treated sewage resulted in no decrease of virus after 1 hour (3). With high doses (40 mg/l for 10 minutes), 99.9% of the viruses in secondary sewage were destroyed (3). A combined residual of 400 mg/l resulted in the inactivation of 99.99% poliovirus type 1 in 30 min in primary effluent, but such high levels of chlorine are costly and may produce potentially carcinogenic substances. Kott (24) found that after a 2-hr application of 8 mg/l chlorine, the level of naturally occurring enteroviruses in oxidation pond effluent decreased 46%.

Clearly, it is difficult to substantially reduce virus numbers in primary and secondary sewage by chlorine application, but in advanced wastewater treatment systems involving physical-chemical treatment, where the amount of organic matter, ammonia, and turbidity are greatly reduced, large reductions of virus are possible (12). For example, Nupen (32) reported reductions of 7 logs of virus after breakpoint chlorination of tertiary treated effluents.

Other methods are currently being investigated for disinfection of wastewater, such as chlorine dioxide, bromine chloride, UV light, ionizing, irradiation, ozone, etc., but these methods have been studied only in the laboratory or on a pilot scale (45). Ozone has received the most attention since it has been used to treat water supplies in Europe and Canada for over 60 years (45), but it is only now being studied for wastewater treatment. Recent pilot studies indicate that it is often difficult to disinfect secondary effluents with ozone and to consistently meet nominal bacteriological standards and that tertiary treatment may be required.

Our review indicated that some level of viruses will be present in the effluent from commonly used wastewater plants. Sproul (44) and Gerba *et al.* (17) previously estimated the percentage removal of viruses by various wastewater treatment systems. This information has been used to calculate the viruses remaining after various treatment sequences (Table 3). It should be emphasized that this information is largely based on work done with enteroviruses, and other viral groups may be removed more or less efficiently by these treatment processes. Based on the estimates of Clarke *et al.* (10) that raw sewage contains on the average 7,000 plaque-forming units (PFU) per l, Sproul (44) estimated that 350 viruses/liter would remain after activated sludge treatment, including chlorination and physical-chemical treatment involving phosphate precipitation and 6 PFU/liter remained after activated carbon adsorption and chlorination about. In other parts of the world, virus concentrations would be much higher. Calculations using the data of Buras (9) show that as many as 25,000 PFU of virus per l may be present after secondary treatment and chlorination (Table 3).

Removal of Suspended Solids During Pretreatment

The suspended solids content of wastewater is important in land treatment because of their effect on the soil infiltration rate. This is

Table 3. Estimated Enterovirus Reduction by Various Sewage Treatment Methods

Treatment	Percent virus removal expected	Viruses remaining in effluent: Clarke et al.(10)[a]	Buras (9)[b]
		PFU/l	
1. Primary treatment			
Sedimentation	0	7,000	500,000
Chlorination	50	3,500	250,000
2. Secondary treatment			
Stabilization ponds	90	700	50,000
Trickling filters	50	3,500	250,000
Chlorination (after trickling filters)	50	1,750	125,000
Activated sludge	90	700	50,000
Chlorination (after activated sludge)	50	350	25,000
3. Tertiary treatment (after sedimentation)			
Excess lime precipitation	90-99.99	700-0.7	50,000-50
Alum precipitation (after lime)	90	7-0.07	5,000-5
Chlorination (after lime and alum)	99-99.99	0.07-0.000007	50-0.0005
Activated carbon adsorption	0-50	--	--

[a]Based on an estimate of 7000 PFU/liter
[b]Based on a finding of 500,000 PFU/liter in wastewater in Israel.

particularly important in managing high rate infiltration systems because maintaining high infiltration rates over a long period of time is essential for an effective treatment system. Infiltration rates of 40 to 60 cm/day have been maintained during the 10 years the Flushing Meadows project was operated (1967-1976) (6,28). The soil at the site in a dry river bed contains about 89% sand, 8% silt, and 3% clay. The optimum flooding cycle was 2 weeks flooding, alternated with 1 to 2 weeks drying (1 week drying in summer, 2 weeks drying in winter). Infiltration rates declined linearly during flooding and were restored in sigmoid fashion during dry periods. Experiments with soil columns placed in a greenhouse showed that high infiltration rates could be maintained if the secondary effluent contained less than 10 mg/l suspended solids (34).

Experiments with soil columns placed in an insulated shelter with the top of the columns extending through the roof showed that high infiltration rates could also be maintained using primary sewage effluent (27). After the columns had been flooded on a schedule of 9 days flooding alternated with 5 days drying for 8½ months, the average infiltration rate had decreased by only about 3%. The average infiltration rate for the 8½ months of flooding with primary sewage effluent was 17.5 cm/day as compared with an average rate of 21.1 cm/day for the same columns for 10 months of intermittent flooding with secondary sewage effluent during the previous year. The suspended solids content of the primary effluent ranged from 51 to 181 mg/l. Evidently the suspended solids of primary sewage effluent reduced infiltration rates much less than the suspended solids of secondary sewage effluent. The reason for this may be that suspended solids of primary sewage effluent were degraded much more rapidly after they had accumulated on the soil surface than those of secondary sewage effluent.

The effect of flooding with a third kind of wastewater was studied in a field system with 16-ha recharge basins flooded with secondary sewage effluent from a 32-ha detention pond (27). The several days' detention time in the pond allowed a dense algal population to develop.

Infiltration rates ranged from 20 to 30 cm/day even though the soil was very similar to the area with 40 to 60 cm/day infiltration rates of secondary effluent in previous experiments. Preliminary studies indicated that the algal content of the water from the detention pond was probably the most important factor in reducing the infiltration rate and that much higher infiltration rates could be obtained with secondary effluent that had bypassed the 32-ha detention pond. This indicates that the algae of oxidation pond effluents may present a problem in maintaining high infiltration rates in a rapid infiltration system.

Thus, pretreatment to maintain infiltration rates is important mainly for rapid infiltration systems. Furthermore the available data indicate that only limited pretreatment (primary) is needed to insure high infiltration rates in coarse sandy soils. However, data are not available on the effect of different types and concentrations of suspended solids on infiltration rates of soils that may be marginal in providing high infiltration rates for rapid infiltration systems. The algal content of water from oxidation ponds or lagoons may be a serious problem in maintaining high infiltration rates. However, this effect is highly site specific and infiltration rates for waters from lagoons in different regions cannot be predicted without additional data.

Nutrient Removal

Conventional treatment systems are relatively ineffective in removing nitrogen and phosphorus from wastewater.

Primary treatment usually results in only slight removal while secondary treatment results in 10 to 20% removal. Therefore, conventional treatment has little effect on the nutrient load applied to land treatment systems. Various tertiary treatment systems can be used to remove most of the nutrients at considerable expense. Lagoon or oxidation ponds vary in their effectiveness depending upon the detention time and management system. For example, the storage lagoons of the Muskegon project reduced the total nitrogen (N) content from 8.2 to 5.6 mg/l and total phosphorus (P) from 2.4 to 1.4 mg/l (46). The Deer Creek Lake lagoons reduced the total N from 50.6 to 5.4 mg/l and P from 6 to 0.4 mg/l (25). Some of the reduction was due to dilution. Therefore, lagoons can significantly reduce nutrient loads when used as pretreatment before land application. If storage is required, nutrient removal will depend largely upon the detention time in the lagoon.

Heavy Metal Removal

Removal of heavy metals may be necessary to avoid injury to plants and to prevent harmful accumulations of metals in food chain crops and soils or movement to groundwater. Most heavy metals are removed by conventional treatment (Table 4) and even primary treatment is effective in producing wastewater adequate for land treatment (46). This is due to the association of most of the metals with the suspended solids. However, this would not be true for some effluents with large amounts of industrial wastes containing high metal concentrations.

Degree of Pretreatment Needed for Different Land Treatment Systems

Since the maximum loading rates, pathogen hazards, etc. are highly site specific, we cannot formulate definite rules for the degree of pretreatment needed. The following principles and alternatives should be considered in designing the various types of land treatment systems. Table 5 shows the problems associated with the different types of land treatment systems. A high degree of pathogen removal is required before wastewater can be used to irrigate edible crops. Since even disinfection does not produce a pathogen free effluent, excluding crops consumed raw from wastewater irrigation is more practical and less hazardous than extensive pretreatment. Irrigation of crops which are processed before sale should not be hazardous if at least primary treatment precedes land application. Irrigation with wastewater should not contaminate groundwater except when very coarse soils are used.

Only limited information is available on aerosols from irrigation systems. Sepp (36) detected coliform bacteria as far as 3 m downwind from spray limits in a sparsely vegetated area when ponded and chlorinated activated sludge effluent was applied by spray irrigation. Katzenelson and Teltch (22) isolated coliforms with an Anderson sampler 350 m from sprinklers applying raw sewage water. A Salmonella bacterium was isolated 60 m from the sprinklers.

Sorber (43) concluded that aero-

Table 4. Concentration of Metals in Various United States Wastewaters[a]

Element	Untreated Wastewater	Primary Effluents	Secondary Effluents	EPA Recommended Drinking Water Standards
	mg/l	mg/l	mg/l	mg/l
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]Adapted from Process Design Manual for Land Treatment of Municipal Wastewater (46)

solization of wastewater seems to range from 0.2 to 0.4%, depending on spray equipment and the prevailing meteorological conditions. Sunlight, temperature, and relative humidity affect virus survival in aerosols. Atmospheric stability is the most important meteorological variable affecting virus survival. Under the least desirable meteorological conditions, less than 200 m would be required to provide a reduction of three orders of magnitude in aerosolized virus concentrations (43). This would suggest that very low levels of enteroviruses would be present at 200 m when secondary sewage effluent was sprayed, but considerably more might be present at this distance if raw sewage was sprayed.

Katzenelson, *et al.* (23) reported that the incidence of enteric disease in agricultural communal villages in Israel near areas which were spray-irrigated with wastewater was higher than in similar villages not near spray-irrigated fields. However, the wastewater was concentrated sewage that was sprayed on fields after only a few days detention time in lagoons. Also, it was not determined whether the enteric diseases were transmitted by aerosols or by other means.

Thus, the limited data available indicate that aerosols from raw sewage used for sprinkler irrigation probably would be hazardous, while spray irrigation with secondary sewage effluent might present some hazard. The possible aerosol hazard from spray irrigation with secondary sewage effluent could be eliminated in several ways. Disinfection of secondary sewage effluent by chlorination should reduce to negligible levels the spread of pathogens by aerosols. However, it may be more practical to eliminate aerosols by applying the sewage by surface irrigation rather than by sprinkling. Buffer zones can also be used to reduce aerosol hazards. Disinfection is usually advisable when wastewater is used to irrigate golf courses or parks.

Contamination of crops is not a

Table 5. Potential Problems Associated with Different Land Treatment Systems

Types of Land Treatment Systems	Problems Associated with Land Treatment				
	Soil Clogging	Food Contamination	Ground-water Pollution	Surface Water Pollution	Aerosols
Overland flow	-	-	-	++	+
Irrigation (slow rate)	+	+	+	+	++
Rapid infiltration	++	-	++	-	-

\- Little or no potential problem
\+ Moderate potential
++ Considerable potential

problem with high-rate systems because crops are not usually grown on these lands. Movement of pathogens to the groundwater is the primary concern for high-rate systems. The available data indicate that both bacteria and viruses can move in large numbers through coarse sand and gravels to the groundwater.

The efficiency of virus removal by rapid soil infiltration has received only a limited amount of field study. Field studies at the Flushing Meadows rapid infiltration wastewater land treatment site near Phoenix, Arizona, indicated that at least 99.99% of the naturally occurring viruses in secondarily treated sewage could be removed by passage through 9 m of fine sandy loam soil(20). These results were confirmed by laboratory studies with the same soil (26). However, sporadic isolation of enteric viruses in groundwater below coarse sandy soils at some land treatment sites (37,47) indicated that efficiency of virus removal may be site dependent. There are indications that the nature of the soil, as well as environmental conditions (i.e., rainfall), can greatly affect the efficiency of virus removal by soil filtration (26). Management practices to reduce virus movement through soils probably can be developed but little research has been done on these practices. With soils with a loamy sand or finer texture, only a few fecal bacteria move through 2 m or more soil to the groundwater (7). The limited data available indicated that viruses would not move through these soils (13,28,50). The fecal bacteria seem to be eliminated by about 100 m of lateral movement in groundwater. Therefore, pretreatment could reduce the numbers of pathogens moving through rapid infiltration systems with very coarse soils. However, pretreatment would not be a factor if sufficient underground detention time could be provided before movement of the reclaimed water to wells or streams. Recently, Gerba and Lance (19) showed that equal amounts of seeded virus were removed from primary and secondary effluents by soil columns. Since these are general statements, each high-rate system must be considered separately. More research is needed to determine exactly which soils permit passage of some bacteria and viruses and which soils effectively remove these pathogens. One way to prevent the spread of pathogens in the groundwater is to intercept the renovated water and reuse it for irrigation, industrial, or recreational purposes. Water treated by most high-rate land filtration systems would meet pathogen standards for unrestricted irrigation and would require minimal treatment for industrial use or recreational lakes.

Pretreatment by conventional treatment plants has relatively little impact on nutrient removal by land treatment systems. Some nitrogen is removed by secondary treatment, but the carbon to nitrogen ratio of primary effluent is more favorable for denitrification in land application systems. However, storage lagoons can have considerable impact on the nutrient removed from wastewater. Lagoons can be managed by varying detention times, etc., to provide any additional nutrient removal needed above that provided by land treatment. The combination of lagoons and land treatment needed depends upon economics and local conditions. When extensive storage is required due to climate, it may be desirable to design and manage the storage lagoon to provide extensive nutrient removal.

If a land application system does not provide adequate phosphorus removal, pretreatment with materials like lime and alum can be used to precipitate phosphates (5).

In summary, the equivalent of primary treatment usually is needed to provide some stabilization and removal of suspended solids and heavy metals. In some cases, secondary treatment is needed to provide further stabilization and to minimize odors. Secondary treatment is designed primarily to remove BOD and accomplishes this goal. However, BOD loads are not a serious problem in land treatment systems because soils can handle high BOD loading rates. Secondary treatment also provides some pathogen removal but leaves relatively high pathogen populations in the effluent. Lagoons can be used to provide varying degrees of treatment and nutrient and pathogen removal could be an added benefit to storage systems. Due to its high cost, rarely would tertiary treatment be used before land treatment. In fact, land treatment often is used in lieu of in plant tertiary treatment. Most land application systems can be operated without using disinfection as pretreatment but sometimes a high degree of human contact with treated crops, soils, or aerosols may necessitate disinfecting wastewater before land application. The exact degree of pretreatment required depends upon the wastewater characteristics, soils, climate, degree of human exposure around the site and the intended use or disposition of the renovated water and should be determined separately for each system.

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PRETREATMENT REQUIREMENTS FOR LAND APPLICATION OF WASTEWATER

PREAPPLICATION TREATMENT FOR OVERLAND FLOW

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Overland flow, while well accepted for treating several industrial wastewaters, is a developing technology for treatment of domestic wastewater. The research team at the EPA laboratory in Ada, Oklahoma, has been conducting studies on overland flow treatment of domestic wastewaters since 1971. These studies included a series of field-plot studies to compare the effects of preapplication treatment on the quality of product water produced by overland flow treatment.

The results reported compare the treatment achieved for 3 levels of preapplication treatment and 2 loading rates for each level of preapplication. Overland flow treatment of raw domestic wastewater at a loading of 5.2 M/yr is an effective advanced treatment process. Overland flow at 10.4 M/yr preceded by primary treatment gives comparable results for removal of total suspended solids (TSS) and total organic carbon (TOC) but less effective removal of total nitrogen and total phosphorus. Overland flow at 20.8 M/yr preceded by secondary treatment achieves better removal of TSS and TOC but substantially less effective removal of total nitrogen and total phosphorus.

INTRODUCTION

Treatment of wastewater by allowing it to flow as a thin sheet over gently sloping ground is commonly referred to as "spray runoff," "grass filtration" or "overland flow." Overland flow has been adopted by the United States Environmental Protection Agency (U.S. EPA) and will be used in this discussion of preapplication treatment as a design factor. Overland flow is not a new technology for disposal of wastewater. It has been in use for many decades for disposal of industrial wastewaters and municipal wastewater on a limited basis. Conversely, the concept of overland flow as a treatment process is relatively new. Research to understand the removal mechanisms and consequently the control of these mechanisms by design and operation is a rapidly advancing science.

Increased scientific interest in the use of the overland flow concept by food processors in the United States appears to originate from experiences with a sprinkler disposal system reported by Luley(1) in 1963. Subsequently, the concept has been further developed and installed as the treatment process at other food processing plants (2)(3). Direct involvement at several of these successful industrial locations and knowledge of results at scattered municipal systems around the world stimulated EPA researchers in Ada, Oklahoma, to initiate a program on overland flow treatment of raw sewage in 1970. The program has covered many aspects of overland flow technology in its eight year history.

The Ada research team became particularly interested in preapplication treatment of municipal wastewaters while conducting a study at a food processing plant. Law, Thomas, and Meyers(4) reported that this system was producing an effluent with a biochemical oxygen demand (BOD) of 9 mg/l while being loaded with a raw wastewater having a BOD of 5 mg/l. This direct experience, coupled with other reported results on treatment of high BOD wastewaters, led the research team to conclude that overland flow treatment of raw domestic wastewater had higher potential as a low cost treatment alternative.

The purpose of this paper is to highlight the principal results of seven years of research by the U.S. EPA team at Ada. The paper summarizes study of overland flow as a complete treatment process and as a polishing process following conventional primary or secondary treatment. Emphasis is placed on a comparison of effluent quality obtained with or without the preapplication treatment. The principal comparison is for no preapplication treatment (raw sewage), conventional primary preapplication treatment, and conventional secondary preapplication treatment.

EXPERIMENTAL SITE CONDITIONS AND OPERATIONS

The study site at Ada, Oklahoma, is suitable for year round operation with minimal considerations due to severe weather conditions. Much of the annual precipitation of about 100cm occurs during high intensity storms of short duration. There is an average of 26 days per year with more than 1.25cm of precipitation. The long summers are warm with a high percentage of clear days. Average minimum temperatures are above freezing for all months except January when the average minimum temperature dips to -1.0°C. The seven year study period was characterized by typical variations in weather conditions.

All experiments were conducted on plots measuring 11 meters by 36 meters with slopes of 2 - 4%. Raw domestic wastewater was obtained directly from the city sewer main and the primary and secondary effluents were obtained from the city treatment plant. Secondary treatment was provided at the city plant by a trickling filter. Figure 1 shows a schematic of the handling system for the raw wastewater. The system for handling primary effluent and secondary effluent was similar except that the settling tank was replaced with a flow equalization tank and the mutrator was eliminated from the system. Flow measuring and sampling was provided for each of the seven plots that were used to compare many different design and operating variables. Wastewater was applied to the plots through the rotating distributor shown in Figure 2. These distributors applied the wastewater over a circle 10 meters in diameter at the upper edge of the plot. This circular area of $78.5m^2$ was about 20% of the total plot area of $396m^2$. The 11 x 26 meter area downslope from the actual application area received the wastewater by sheet flow from the upslope area.

RESULTS AND DISCUSSION

This 12 month study was devised to evaluate the effect of increasing the level of preapplication treatment on the quality of effluent after overland flow and on the amount of wastewater that could be applied to the overland plots. The operating conditions for this study included 3 levels of preapplication treatment and two amounts of wastewater loading to the overland flow plots. The schedule of applying the wastewater was the same for all plots. Each plot was dosed for 5 consecutive days and then rested for 2 days in winter while being dosed for 6 consecutive days and then rested for 1 day in summer. The average annual loadings for the 3 levels of preapplication treatment were as follows: raw domestic wastewater 5.2 and 10.4 M per year, primary 7.8 and 10.4 M per year, and secondary 13.0 and 20.8 M per year. Daily and weekly loadings were seasonally adjusted by as much as 15% more in summer and 15% less in winter for some of the test periods.

The chemical composition of the raw domestic wastewater, the primary effluent, and the secondary effluent is summarized in Table 1 for selected parameters. The data show the raw domestic wastewater to be a medium strength source for the United States. The concentration of suspended solids and organic carbon in the primary and secondary effluents were comparatively low in comparison to typical values. This would indicate that the treatment plant was operating very efficiently during this period of study.

Chemical composition of the renovated wastewater collected after overland flow treatment is summarized in Table 2. All of the data presented show overland flow to be an effective process for removal of suspended solids, organic carbon, and total nitrogen. Removal of phosphorus is substantially less and is more dependent on the level of preapplication treatment and the amount of wastewater loading. There was not a consistent improvement in the removal for all parameters as the level of preapplication treatment was increased to secondary. Neither was the best removal achieved for all four parameters with the lower amount of annual loading.

The concentration of suspended solids after overland flow treatment did decrease with an increasing level of preapplication treatment even though the wastewater loading for the highest amount of secondary effluent loading was four times the amount for the lowest loading of raw wastewater. When applying raw domestic wastewater, doubling the wastewater loading resulted in a doubling of the suspended solids in the system discharge. A 33% increase in the primary effluent loading and a 60% increase in the secondary effluent loading did not change the suspended solids concentration after overland flow treatment. These results indicate that removal of suspended solids from primary effluent or secondary effluent by overland flow is comparable for substantially different wastewater loadings whereas removal of suspended solids from raw wastewater was sensitive to the amount of wastewater loading.

The concentration of total organic carbon after overland flow treatment also decreased as the level of preapplication treatment increased to secondary. In contrast to the results for suspended solids removal there was no appreciable differences between the low and high loadings regardless of the level of preapplication treatment. These results indicate that none of the loading rates selected for study exceeded the capacity of the system to stabilize the organic carbon. Since the capacity of the system to stabilize organic carbon was not exceeded at the high wastewater loadings, it is logical to assume that solids removal through the primary and secondary treatment processes reduces the amount of slowly oxidizable organic carbon in the wastewater. Proof of this assumption would require fractionation and identification of the organic carbon in the applied wastewaters and the overland flow product water.

Biochemical oxygen demand (BOD) was measured during the early phases of these overland flow studies to establish a ratio of BOD to TOC. Once a firm ratio was established the time consuming BOD test was dropped from the analytical program because BOD could be accurately estimated from TOC data. A total of 40 sets of data were used to establish the ratio of BOD to TOC for the renovated wastewater following overland flow treatment. On the basis of these data sets BOD equals 0.54 times the TOC with a very narrow range of 0.47 to 0.62 times the TOC. This ratio, 0.54 times TOC, was used to calculate the BOD values in Table 2 which show that BOD values were generally less than 10 mg/l.

The concentration of total nitrogen after overland flow treatment varied with both the amount of wastewater applied and the level of preapplication treatment. Increasing the level of preapplication treatment resulted in an increase in the concentration of total nitrogen after overland flow treatment. Similarly, increasing the loading resulted in an increase in the concentration of total nitrogen for all three levels of preapplication treatment. These results show that nitrogen removal by the overland flow process is very dependent on the source of the wastewater and the amount of wastewater loading. Previous results reported by Thomas, Jackson, and Penrod ([5]) attribute nitrogen removal to microbial denitrification. The results of this study are consistent with the hypothesis that the nitrogen is being removed by microbial denitrification. The higher oxygen demand in the raw wastewater makes it easier to deplete available oxygen and sustain a low oxygen environment which promotes denitrification following nitrification in aerobic areas. Increasing the loading of this wastewater too much depletes oxygen to the point where nitrification is blocked and high concentrations of reduced nitrogen forms (ammonia and organic) remain after overland flow treatment. The secondary level of preapplication treatment reduces the oxygen demand in the wastewater to where denitrification is limited by higher oxygen concentrations. In this case higher concentrations of oxidized nitrogen forms (nitrate) remain after overland flow treatment. The application of 34.5 mg/l of reduced nitrogen forms in the raw wastewater results in 5.5 mg/l after overland flow treatment and removal of 85% of the applied nitrogen. Doubling the loading of the raw wastewater to 10.4 M/yr started to block nitrification or to exceed the nitrification/denitrification capability and the removal of nitrogen dropped off to 73% and 9.2 mg/l remained after overland flow treatment. The application of 15.3 mg/l of partially oxidized nitrogen in secondary effluent results in up to 9.8 mg/l of oxidized nitrogen (nitrate) after overland flow treatment and removal of 36% of the applied nitrogen.

The concentration of total phosphorus after overland flow treatment increased as the amount of wastewater applied was increased regardless of the level of preapplication treatment. There was also a general decrease of phosphorus removal by overland flow treatment with greater preapplication treatment even though more phosphorus was removed by each level of preapplication treatment. The best phosphorus removal, 52%, was for the lowest loading of raw domestic wastewater while the poorest removal, 11%, was for the highest loading of secondary effluent. Many researchers (2)(4)(6) have reported that phosphorus removal is comparatively low for overland flow while Thomas, Bledsoe, and Jackson (7) have reported on enhancement of phosphorus removal by chemical addition. Phosphorus removal by soil and plants is obviously a function of the quantity applied, the available reaction surface, and the time of exposure to the reaction surface. Overland flow, unlike other land treatment systems, provides a short contact time over a limited reaction surface. Because of these operating conditions phosphorus removal will be comparatively low unless the rate of the precipitation reaction or the reaction surface is augmented in some manner.

SUMMARY

These data, extracted from parts of seven years of study, show that overland flow can provide excellent treatment of raw domestic wastewater without primary or secondary preapplication treatment. The product water obtained by direct application of raw domestic wastewater is representative of advanced treatment and equal to or better than the renovated water obtained by providing either primary or secondary preapplication treatment. The cost of overland flow as a process unit with a loading of 5.2 M/yr is approximately equal to the cost of conventional primary treatment. It is obvious that providing primary or secondary preapplication treatment increases the total cost regardless of the fact that the loading can be increased substantially by providing more preapplication treatment.

Overland flow for treatment of domestic wastewater is a developing technology for which practical considerations of location and site conditions have a strong influence in the decision process. Seasonal preapplication treatment will always be needed where climatic conditions prevent year round operation. Since the concept is new and comparatively unproven, gaining acceptance by state agencies, the consulting profession, and the public will be difficult. It is natural for most people in all walks of life to accept standard practice rather than to be a pioneer for something different. Research and pilot system results are establishing overland flow as a low cost and energy saving alternative for wastewater management. The transition from technical proof of the process to general acceptance will be frought with controversy until there are several operating systems that demonstrate and verify the findings of the research studies.

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Table I. Composition of Wastewaters for the Preapplication Treatment Study

Parameter	Raw Domestic Wastewater	Primary Effluent	Secondary Effluent
number of samples	30	34	32
TSS	170	52	12
TOC	90	31	16
total N	34.5	19.0	15.3
total P	8.1	7.3	6.6

Table II. Chemical Quality after Overland Flow Treatment - Mean Concentration, mg/l

	Wastewater source and annual loading, M/yr					
	Raw Domestic Wastewater		Primary Effluent		Secondary Effluent	
Parameter	5.2	10.4	7.8	10.4	13.0	20.8
TSS	8	16	7	7	5	5
TOC	17	19	14	14	11	10
BOD(1)	9	10	8	8	6	5
total N	5.5	9.2	6.7	6.7	7.6	9.8
total P	4.2	4.7	4.7	5.2	5.0	5.9

(1) calculated as 0.54 times TOC based on 40 sets of BOD/TOC data

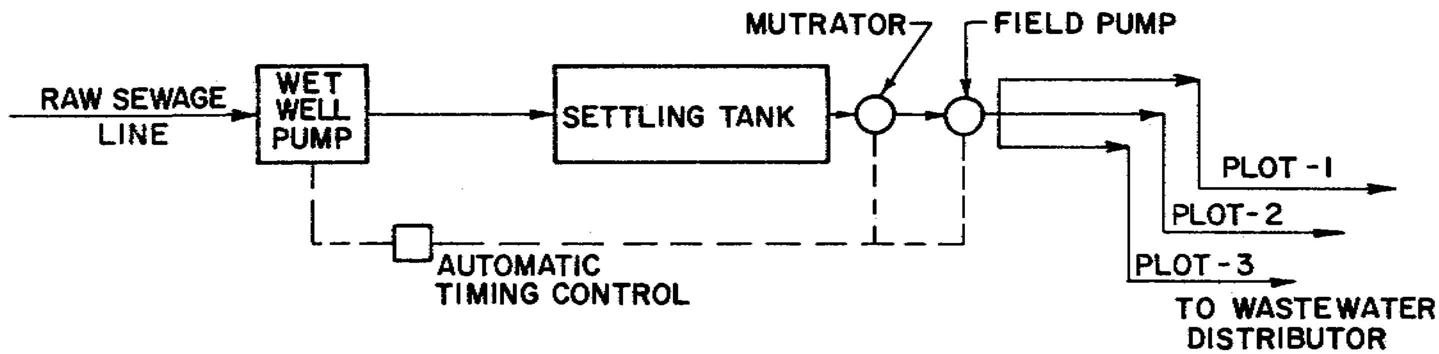

FIGURE 1 - SCHEMATIC OF WASTEWATER HANDLING SYSTEM

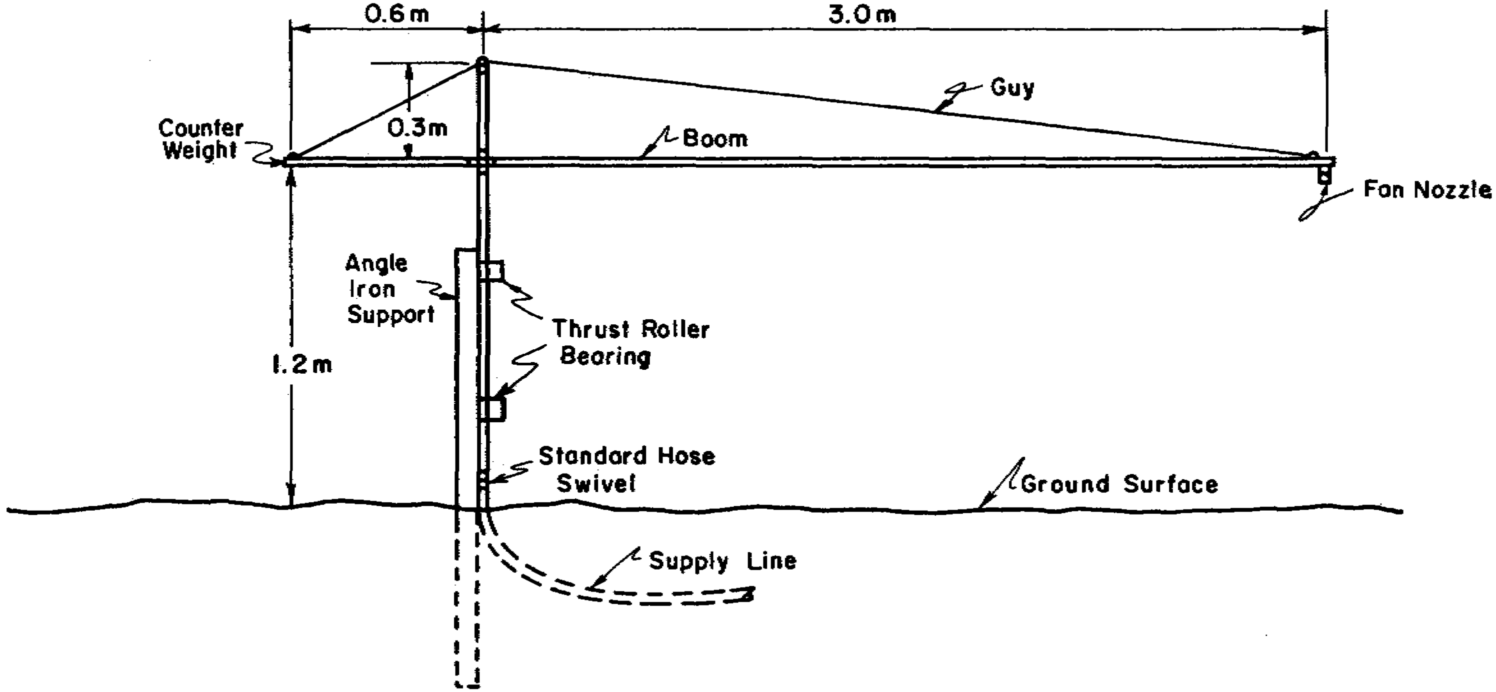

FIGURE 2 - DETAIL OF WASTEWATER DISTRIBUTOR

AGRICULTURE AND FOREST USE

AGRICULTURAL PRACTICES ASSOCIATED WITH LAND TREATMENT OF DOMESTIC WASTEWATER

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Important factors in the success of land application of wastewater systems are the field design and agronomic management during operation. Crop selection should be based on their adaptability to the region, marketability, tolerance to wet soil conditions and soluble salts, disease resistance, and their nitrogen uptake rates. Two or more crops allow more freedom in timing of operations and flexibility in scheduling irrigations and may offer agronomic advantages.

Crops that remove large amounts of nitrogen are desirable to maximize removal of nitrogen from the wastewater. We present an equation that relates concentration of nitrogen in percolate water to concentration of nitrogen in the wastewater, amount of wastewater applied, nitrogen uptake by the crop, rainfall, and evapotranspiration.

On many soils underground drainage will be required because of high natural or perched watertable conditions. Deeper and more closely spaced drains than the normal design will be needed to achieve the required unsaturated soil depth.

Irrigation designs for application of wastewater range from a continuous mist to surface flooding. Factors, like land topography, prevailing winds, soil-type, operator-skill requirements, and available labor must be considered. In many wastewater application systems, the soil will seal at the surface and materially reduce infiltration. Tillage, residue, and cover-crop management systems are suggested to minimize sealing during production of row crops.

Irrigation design and schedules must consider the hydraulic properties of the soil, rainfall, and evapotranspiration and irrigation must be scheduled to allow drying time before crop planting and harvesting. Drying times will depend upon soil and climatic characteristics.

Planting of row crops in tilled strips of sod crops has the advantages of a longer season for nutrient uptake by plants and less surface-soil sealing. In many climates, two or more crops may be planted in 1 year, which also will enhance nutrient removal.

INTRODUCTION

The success or failure of a land application of wastewater system will greatly depend upon soil and crop management. This is true, if an adequate design and site have been selected.

Field management can be prescribed in broad outlines, although daily alterations will often be necessary because of variations in weather and crop systems. Soil microorganisms, agronomic crops, and plant diseases are biological systems that are highly influenced by weather conditions and also are responsive to management by man.

The object of this report is to outline field management practices and how they are used to maximize the nutrient renovation of wastewater when it is recycled through a soil-crop-system.

CROP SELECTION

Selecting the proper crops for a wastewater irrigation system is a key factor in the success of land treatment. Land treatment is a biological renovation system, and when we select crops that will perform the needed function, e.g., transfer nitrogen from wastewater to harvestable portions of the plant, under the conditions created by land application we must consider both the nature of the plant and the environment in which it is grown. Crop selection should be based largely on local climate and market conditions, the need for a variety of crops, their tolerance to total salts or specific ion components of the wastewater, their disease resistance, and their nitrogen-uptake rates into plant tissue.

Crops well adapted to local conditions (irrigated in dry areas) should be used for initial screening for wastewater-irrigation systems. Crops adapted to local conditions of temperature, humidity, day length, radiation, and season length are integrated into a range of crops that are known to produce economic yields in the region. This list of adapted crops can be narrowed to those with an existing or potential market in the area.

Crop selection should not be narrowed to a single crop because a variety of crops will enhance market stability, provide some variation in timing and types of cultural operations, e.g., planting, tillage, and harvesting; benefit soil physical and chemical conditions; and provide a variety of timing and amounts of wastewater irrigation. Variety in irrigation timing is a benefit to wastewater-irrigation systems. For example, a system with both forages and corn can be used so that the corn is irrigated while the soil in the forage systems is drying for harvesting. Forages can be irrigated early and late in the season before the corn is planted and after it is mature, as well as during the corn growing season.

Crop selection will also be governed by the crop's tolerance to total salts and specific ion components of the wastewater. Typical municipal secondary effluent is classified high saline and low to medium sodium and, thus, is not a sodium problem on most soils but might require plants with good salt-tolerance. Reduced crop production by salts from irrigation water is usually due to an increased osmotic potential of the water which reduces the availability of water to plants. Additional detrimental effects of salts in irrigation water are due to specific ion-toxicities. The USDA Salinity Handbook (8) is an excellent source book for selecting crops with high degrees of tolerance for total salt and specific ion problems.

Crops selected for wastewater application must be highly resistant to disease, since frequent wastewater applications produce conditions favorable to disease organisms.

A major factor to be considered in selecting a crop is its ability to remove nutrients from the wastewater. In most land-treatment systems nitrogen will be the main nutrient of concern, since most other constituents are filtered or sorbed within the soil or are not major environmental concerns. A crop's ability to remove nitrogen from wastewater will depend on many environmental factors as well as on the crop's physiological requirements.

Table 1 gives a range of nitrogen-uptake values that might be expected for various agronomic crops. These (Table 1) values were compiled from considerable data in the literature for above-average

Table 1. Maximum yield and nitrogen uptake of various species for three regions of the U.S.A.

	North		Central		South	
Crop	Yield	Uptake	Yield	Uptake	Yield	Uptake
	mt/ha					
Alfalfa 1/	10	.300	18	.540	--	--
Coastal bermuda grass 2/	--	--	--	--	24	.600
Kentucky bluegrass 2/	9	.225	12	.300	--	--
Bromegrass 2/	10	.250	18	.450	--	--
Reed canarygrass 2/	10	.250	18	.450	--	--
Tall fescue 2/	10	.250	18	.450	24	.600
Quackgrass 2/	10	.250	--	--	--	--
Corn (fodder) 3/	15	.180	--	--	--	--

1/ Assumes 3.0% N
2/ Assumes 2.5% N
3/ Assumes 1.2% N

yields under excellent climate and soil (including fertility) growing conditions.

Nitrogen uptake by a crop in a wastewater land treatment system is somewhat a function of the amount of nitrogen loading (3). Nitrogen uptake in Minnesota by reed canarygrass ranged from 230 to 400 kg N/ha for applications ranging from 230 to 800 kg N/ha/yr. Corn had a much narrower range of nitrogen uptake with little increase due to increased nitrogen loading. Assuming that denitrification, mineralization, change of storage within the soil, ammonia volatilization and other nitrogen reactions have no net effect on nitrogen uptake and leaching, an expression may be written as:

$$C = \frac{(En - U) \times 10}{E + R - ET} \qquad (1)$$

Where C = average concentration of nitrogen in percolate water (mg/liter); En = nitrogen applied in wastewater (kg N/ha); U = nitrogen uptake by crop (kg N/ha); E = wastewater applied (cm); R = rainfall (cm); ET = evapotranspiration (cm). Two examples for reed canarygrass with effluent at 20 mg/liter and R = ET: (1) E = 200, En = 400, U = 325, and C = 3.8; (2) E = 300, En = 600, U = 400, and C = 6.7. In these examples a 50% increase in effluent application produced a 75% (3.8 vs. 6.7) increase in percolate nitrogen concentration. If crop uptake had remained constant at 325 kg N/ha, the percolate nitrogen concentration would have increased 140% to 9.2 mg/liter.

DRAINAGE

Soil drainage to intercept and remove excess water will probably be required for most land-treatment systems. Under many agricultural conditions, permanent or perched watertables will occur when heavy wastewater loads are applied, and underdrainage is necessary.

Interception of percolated water may be desirable to meet regulatory requirements or to maintain normal streamflow. Once intercepted, it can be further treated, or returned to surface waters. Interceptor drains can be subsurface perforated plastic tile, open ditches, or interceptor wells. Background information for design of such systems can be obtained from Drainage of Agricultural Lands (6).

It may be necessary to remove water from the biologically active zone of the soil under the land-treatment fields to achieve good crop growth and wastewater renovation. High watertables under wastewater irrigation fields will lead to poorly aerated soils with root growth and biological activity restricted. Under most agricultural production systems, the watertable should be maintained between the 0.6 and 1.5-m depth. Drainage design requirements to achieve the unsaturated soil depth are illustrated in Figure 1. The equation relating soil characteristic K (hydraulic conductivity) and the loading rate I (infiltration rate) to the dimensions of the system (shown in Figure 1) is:

$$Hc^2 = Hd^2 + IW(W + 2L/K) \qquad (2)$$

This equation is a steady-state equation since with a given set of dimensions a continuous inflow of I would produce watermounding, characterized by Hc. The equation can be applied to intermittent application by using an equivalent steady-state inflow rate and determining

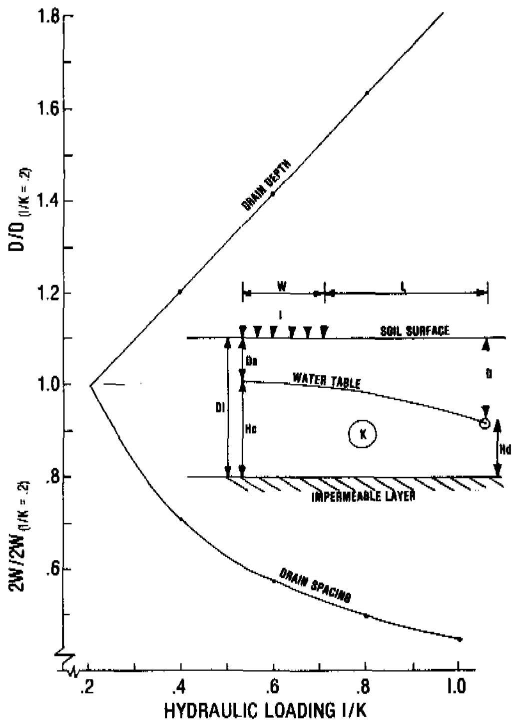

Fig. 1. Relative tile drain spacing (2W) and depth (D) as a function of relative hydraulic loading (I) in order to maintain a satisfactory aerated soil depth (Da). Curves based upon height of mounding equation (Eqn.2) with L = 0, Da = 1.52m; Di=15.2m; K=.10m/day.

the spacing and depth of drain-placement requirements. As hydraulic loading rate increases, drain spacing (2W when L = 0) must decrease and/or the depth of drain placement must increase. Figure 1 shows relative drain depth or drain spacing required to maintain an aerated zone of constant depth as the average infiltration rate (hydraulic loading rate) increases from 0.2 to 1.0K. Drain depth cannot be increased beyond about 2 to 3 m since trenching equipment has limited depth capacity. Qualified drain installers of an area can give recommendations on drainage requirement based on considerable practical experience.

IRRIGATION

Method of Irrigation

Wastewater-distribution systems and application methods are of major concern in designing and managing land-treatment systems. Application methods range between a continuous mist to surface flooding and are alternatives that must be considered. Factors like land topography, prevailing winds, soil type, operator skill requirements, and available labor must be considered. Engineering aspects of design have been adequately discussed elsewhere and will not be reexamined here (5,9).

Method of application influences some soil-management considerations. In a ridge-furrow irrigation system the soil must be shaped and occasionally reshaped into ridges for water conveyance. Weed control may be particularly difficult in a ridge-furrow system.

Ridge-furrow or other surface-flooding systems require considerable operator skill to obtain uniform applications. It is important to initially have a large stream flow until runoff starts and then cut back stream size to avoid excessive runoff and minimize leaching at the upper end of the field. Surface-application systems will also require a tailwater recycling system to retain wastewater within the system until it percolates through a soil profile.

Sprinkler-type application requires a minimum of operator skill since the system can be controlled mechanically. The operator, however, must observe the system periodically to detect malfunctions. A major problem with sprinkler-type irrigation is the action of falling water drops destroying inherent soil structure and forming a dense relatively impermeable crust or seal at the soil surface. The formation of a seal depends on the momentum of the water drops, i.e., both drop size and their height of fall. The sealing action can be minimized by mist-type applications with very small water droplets. However, small water droplets are extremely susceptible to wind drift which can lead to nonuniform application and to considerable movement of airborn material.

The intensity of seal formation depends on the structural stability of the soil. Most agricultural soils will seal when exposed to rainfall or sprinkler irrigation and, under high application of wastewater, will develop reduced intake capacity as the season progresses. Soils with low organic matter contents and high silt content are particularly susceptible. Design considerations for application rates should be based on sealed conditions and site management should be geared to minimize seal formation. Techniques like plant-residue mulches, cover crops, are useful to minimize seal formation.

Irrigation scheduling and loading

Most slow rate systems irrigated with typical secondary effluent will accept more water than nitrogen renovation requirements will allow. Factors to consider in nitrogen loading can be illustrated by use of a nitrogen balance equation:

$$Rn + F + En + M + S = U + Dn + V + N + I \qquad (3)$$

where Rn = rainfall nitrogen; F = fertilizer nitrogen; En = effluent nitrogen; M = nitrogen mineralized from soil organic matter; S = nitrogen returned in crop residue; U = crop uptake; Dn = nitrogen drainage below root zone; V = ammonia volatilization; N = denitrification; I = nitrogen immobilization (all units in kg/ha/time). The major components influencing nitrogen loading are the amount and quality of the effluent applied, crop uptake, and drainage losses. The other components of nitrogen balance will be small and may be self-compensating in a typical slow rate system. Therefore, a simplified nitrogen-balance equation may be written as:

$$En = U + Dn + XEn \qquad (4)$$

where X is fraction of applied nitrogen removed from wastewater by the net effects of all other components of nitrogen balance, and En, Dn, and U were previously defined.

A water-balance equation may also be written (assuming no surface runoff) as:

$$E + R = Dw + ET \qquad (5)$$

where E = effluent loading; R = rainfall; Dw = drainage water; ET = evapotranspiration (all units in cm). Combining equations 4 and 5 and substituting the relationships:

$En = .1\ ECi$

$Dn = .1\ DwCo$

gives an equation for effluent loading as:

$$E = \frac{10U + Co(R - ET)}{(Ci - Co)(1 - X)} \qquad (6)$$

where Ci = input concentration of nitrogen; Co = percolate water nitrogen concentration (units in mg/liter). Equation 6 is a simplified design equation in terms of input nitrogen concentrations, the normal climatic balance between rain and evapotranspiration, the net losses of nitrogen by other mechanism, crop uptake of nitrogen, and the desired (or regulated) nitrogen concentration of drainage water.

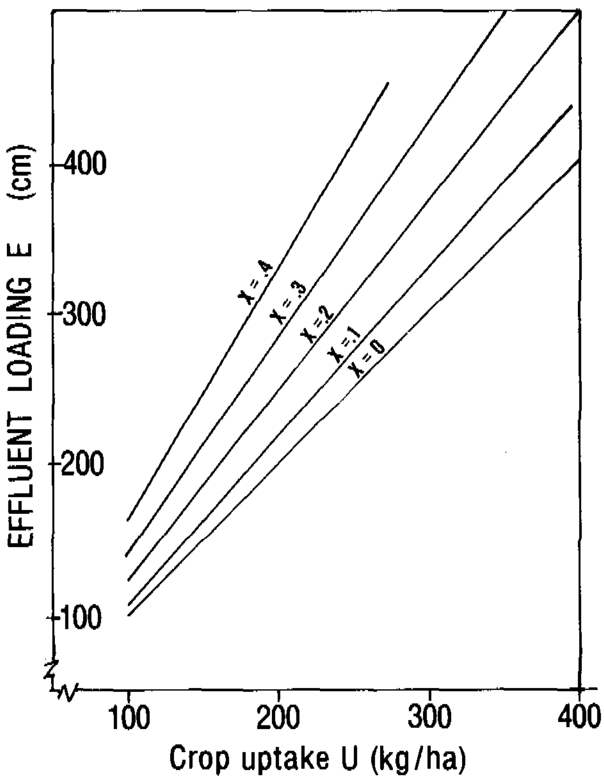

Fig. 2. Annual effluent loading rate (E) as a function of crop uptake of nitrogen (U) for various values of nitrogen lost (expressed as a fraction of the nitrogen applied) by other means (X). Curves based on effluent loading equation (Eqn. 6) with Ci = 20. Co = 10 (units in mg/liter) and R = ET.

Table 2. Climatic data used to establish biweekly effluent loading rates at SW Agr. Exp. Station, Lamberton, Minn., 1961-1976.

Biweekly period beginning	ET1/	Rain	Effluent Loading2/
	---cm---		
4/1	4.0	2.1	10.1
4/15	5.1	3.1	12.7
4/29	6.2	3.7	15.4
5/13	7.0	3.9	17.9
5/27	7.7	4.0	19.7
6/10	8.3	4.3	21.3
6/24	8.6	4.7	21.9
7/8	8.1	4.9	20.1
7/22	7.4	4.3	18.6
8/5	6.7	2.9	18.0
8/19	6.1	2.6	16.1
9/2	5.4	3.6	13.2
9/16	4.7	4.0	10.4
9/30	3.9	3.6	8.4
10/14	3.1	2.5	7.2
Season Total	92.2	54.2	231.0

1/ ET = evapotranspiration ≅ 0.8X (pan evaporation)

2/ Effluent loading assumptions and calculations: Crop uptake = 170 kg/ha; Effluent-N concentration = 20 mg/l; Drainage-N concentration = 10 mg/l; Fraction nitrogen loss other than crop uptake = 0.1; Seasonal effluent loading = 231 cm (Eqn. 6); biweekly loading = ((231 + 54)/92)ET-Rain, (Eqn. 7).

Figure 2 presents some simplified solutions to the loading-rate equation (Eqn. 6) indicating that effluent loading may be increased by increasing crop uptake, U, or the fraction lost by other mechanisms, X. Optimizing conditions for maximum nitrogen removal through crop selection and management will maximize nitrogen renovation of wastewater more than optimizing nitrogen losses by other means.

The distribution of the annual loading rate into daily, weekly, or monthly applications can be determined by considering water balances for shorter time periods. Variation in evapotranspiration, ET, can be used to distribute the annual loading between biweekly periods since plant growth (and, subsequently, nitrogen uptake) increases and drainage volume decreases as ET increases. Table 2 shows a typical variation in ET demand during the growing season and a typical rainfall distribution for Minnesota. Weekly effluent loading can be obtained from such a relationship (Table 2) by using the simple ratio:

$$\left(\frac{E + R}{ET}\right)_{season} = \left(\frac{E + R}{ET}\right)_{period} \qquad (7)$$

where: E, R, and ET were previously defined as irrigation, rainfall and evapotranspiration amounts (units in cm) respectively.

Table 2 shows some sample calculations of biweekly effluent loading based on the normal climatic data. Such a distribution of effluent-irrigation amounts will create a distribution of drainage matched to ET demands and maximize nitrogen renovation by taking advantage of higher soil temperatures and biological activity.

After establishment of weekly, biweekly, or monthly application amounts, it will also be necessary to distribute this water into smaller time intervals of hours or days. Two somewhat opposed principals should govern this distribution. First, maximum residence time within the soil is obtained by applying very low rates continuously. Secondly, short application times allow drying periods for disease and insect control and biological activity. Continuous applications require considerably more management skill and labor than intermittent (daily, every other day, every third day) applications and may also lead to continuously waterlogged soils with restricted oxygen supply. Continuous applications must be at a rate below the unsaturated hydraulic conductivity at 10% residual air saturation (RAS) to meet aeration and oxygen requirements of roots. Figure 3 illustrates some typical data of continuous application rates vs. RAS. The data from which Figure 3 was developed was obtained from a controlled experiment in which soil-water contents and hydraulic gradients were measured periodically during and following an effluent irrigation. The application rate for this

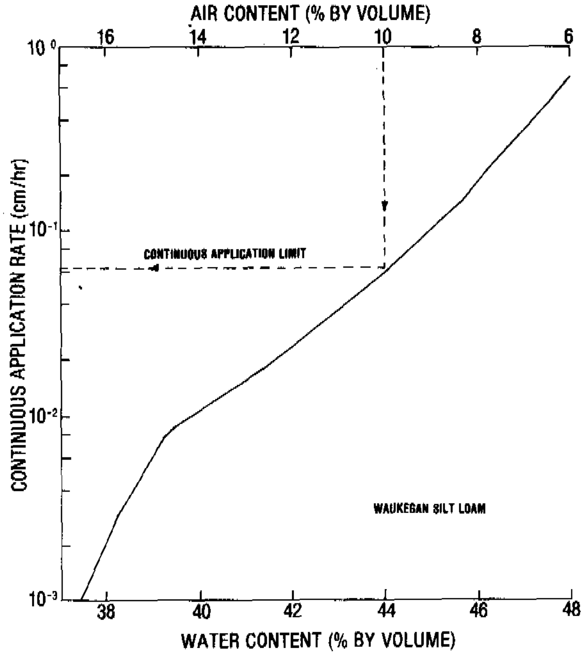

Fig. 3. Hydraulic conductivity or continuous application rate as a function of water and air contents of a Typic Hapludoll.

silt loam soil must be below 0.065 cm/hr to maintain a 10% RAS. Such a restriction on application rate limits weekly loading to about 10.9 cm of wastewater. Intermittent applications could be at higher rates because during drying the matrix potential in the soil increases and, thus, water enters the soil at rates exceeding the saturated hydraulic conductivity. Intermittent irrigations at higher intensities produce wetter soils during the irrigations but drier soils between irrigations.

Experimental data with reed canarygrass indicated that irrigation scheduling with an ammonia form nitrogen wastewater from daily to weekly applications (with equivalent weekly loading) had little effect on the nitrogen renovation capacity of the soil-crop system.

The final design irrigation schedule is then:

1. Intermittent irrigations with 1 or more days between irrigations.
2. Application intensity limited to the soil's capacity to intake water over extended periods of time.
3. Weekly, biweekly, or monthly loading rates based on ET demands and the annual loading.

4. Annual loading rates determined by nitrogen and water balance considerations.

Irrigation scheduling must maintain some degree of flexibility so that the schedule can be interrupted for significant rainfall events and cultural operations. A rule-of-thumb practice might be to increase each irrigation amount by 10% so that about 10% of the irrigations could be omitted from the schedule due to rainfall or cultural operations.

Undue compaction of the soil by tillage or harvesting equipment must be avoided as it will reduce water intake and crop growth. The wetter the soil the greater the degree of compaction from a given load. One to several days of drying after irrigations may be necessary, depending on the soil type, drainage, and ET demand, for the soil to support farm equipment. Longer drying periods would be required in the spring and fall, and for larger equipment.

Figure 4 presents a typical irrigation frequency diagram obtained from a study in Minnesota. When conditions were favorable (high ET and no rainfall), daily irrigations were applied. Early and late in the season and during harvest periods, several days elapsed between irrigations.

CROP AND SITE MANAGEMENT

Close-seeded forages

Perennial forages are high in protein, they generally require minimal management skills and they provide continuous vegetative cover for the soil surface which enhances the infiltration capacity of the soil. Selected forages are well adapted to wet soil conditions and continuous application of small amounts of nitrogen. To optimize production and, thereby, remove the most nitrogen from the soil-crop system, a good stand of the selected forage must be established and the crop must be harvested at appropriate times.

Establishing perennial forage will involve: (1) weed control (including grass); (2) preparation of a good seedbed; (3) selection and planting good quality seed at a recommended rate; and (4) limited irrigation until the plants are established.

Timely harvest, use of insecticides as needed, control of soil pH, and application of required fertilizer will assure maintenance of the stand for optimum production.

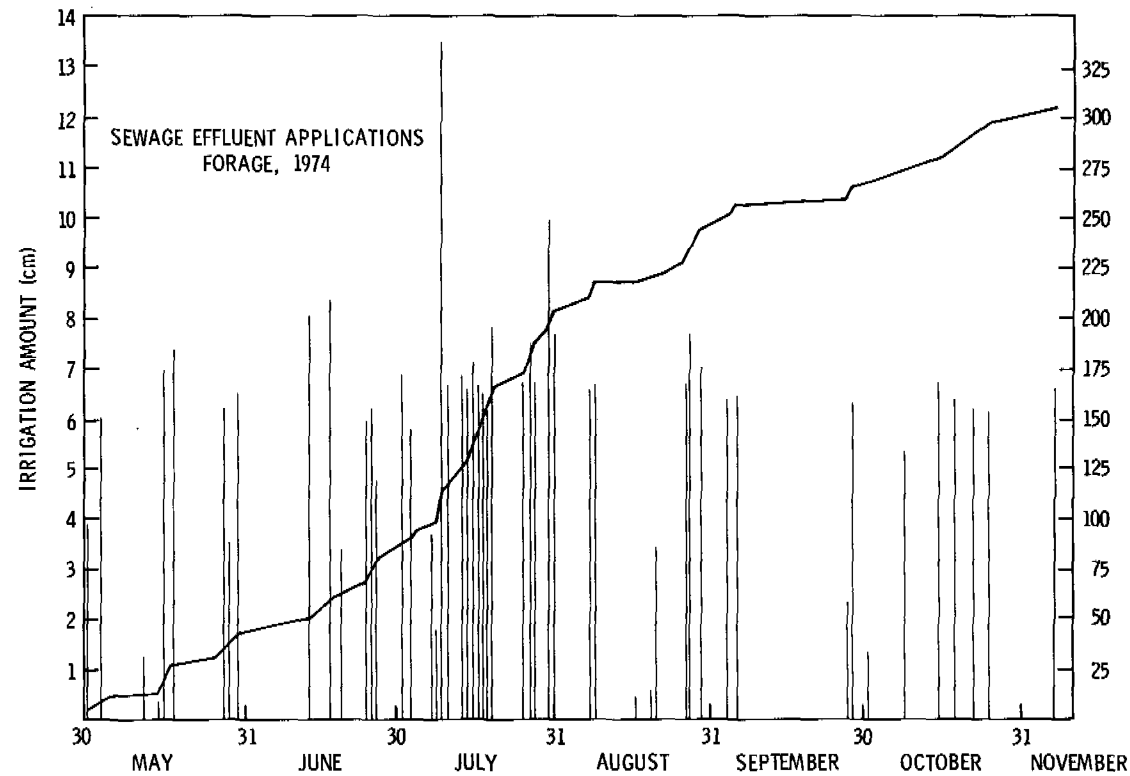

Fig. 4. Daily and accumulative effluent irrigation amounts during an application season.

Row-crop management

Several important soil properties are the result of the interactions of the kind of crop cover, plant roots, plant residue, tillage, irrigation, and climatic conditions. Bulk density, porosity, water conduction and holding capacity, and heat exchange and storage properties, and soil strength--all are properties affected by the interaction of external forces and the crop on the basic soil matrix. The processes of root growth and decay, freezing and thawing, rainfall and irrigation wetting and water-drop impact, interception of water by a crop canopy, and the presence or absence of plant residue material at the soil surface or within the soil are particularly important to the formation and maintenance of soil structural features and soil physical properties. Table 3 gives some typical data obtained from soil sampling following irrigations on density, water, and air contents after 24 hr of drainage and infiltration characteristics of a soil with a perennial forage crop as compared with annual row-cropped corn.

Reduced infiltration capacity, resulting from the formation of a surface seal and a denser soil-surface layer, will probably be the major management problem under row-cropping systems. Low infiltration rates will limit effluent loading rates, enhance the chances for undesirable runoff losses, and also limit soil aeration by restricting air flow and oxygen diffusion through the soil surface layer. Thus, it is desirable to maintain the soil at high-infiltation-capacity condition, which is naturally obtained under a perennial sod crop. The presence of plant residue, tillage, and crop cover will be discussed as techniques for maintaining a high-infiltration-capacity soil.

Tillage loosens soil material, creates large pores and a roughened soil surface, and enhances infiltration capacity (2). Infiltration enhancement from tillage is usually temporary since the soil tends to recompact, become smoother, and decrease in infiltation capacity when exposed to natural or irrigation wetting. Generally, one to several centimeters of rainfall or irrigation are sufficient to reduce the infiltration capacity on freshly tilled surfaces to less than the pretillage condition. If the freshly tilled soil is protected by crop residue, it will take 2 to 10 times more rainfall or irrigation to reduce the infiltration capacity to pretillage conditions.

Crop residue mulches will enhance and help maintain high infiltration capacity conditions (4). Surface residue mulches protect the soil from direct water drop impact. The protective effect of crop residue mulches diminish with time because they are subject to microbial decomposition and will usually be undetectable by the end of a growing season. Under effluent irrigation systems, the microbial decomposition of plant residue is more rapid than under ordinary irrigation systems. Plant residue will be most effective protecting a high-infiltration-capacity condition from water-drop impact, but may also help develop increased infiltration capacity by promoting earthworm activity. Residue on the surface in the spring and fall may significantly enhance earthworm activity and thereby increase infiltration capacity, but may not be important during a single growing season. Alternative or additional infiltration enhancement procedures may be necessary in order to eliminate low infiltration problems during part of the year.

Table 3. Soil physical properties under corn and perennial forage crops.

Crop	Bulk Density	Porosity	Field Capacity (24 hour): Volumetric Water Content	Field Capacity (24 hour): Air Filled Porosity	Final Infiltration Rate (60 min)
	g/cm^3	%	%	%	cm/hr
Corn	1.40	47.1	36.4	10.7	0.8
Forage	1.17	55.9	43.2	12.7	3.3

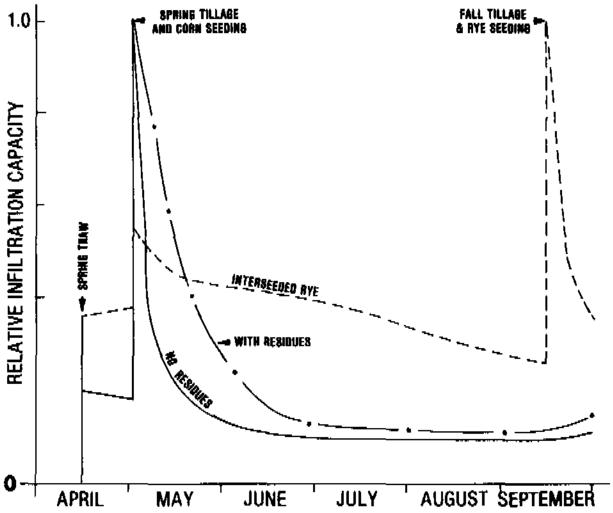

Fig. 5. Relative infiltration capacity (relative to a freshly tilled condition) as a function of time for various management schemes.

Perennial sod crops enhance infiltration conditions and they have been grown between corn rows to enhance and maintain high-infiltration-capacity conditions (1). A live grass interrow crop, like rye or reed canarygrass, has been very effective in maintaining high-infiltration-capacity conditions throughout the application season in our experiment. The grass interrow crop provides year-long coverage and continual root growth and decay processes as well as some residue material to promote soil biological activity. The interrow grass is competitive with corn and some setback (limit growth of grass) procedures may be necessary. Herbicides, partial tillage, and grass cutting are all possible means of setting back the growth of grass, while not entirely killing the established grass crop.

Figure 5 presents a summary of probable infiltration capacity conditions for various residue and tillage management systems throughout the season based upon extensive infiltration data in the literature and observations on our own experiments.

SUMMARY

Agronomic management information is necessary to design a land-treatment system, and sound agronomic decisions are also essential to successfully operate the system. Crop selection based on climatic conditions, salt tolerance, disease resistance, and nitrogen uptake of the crops are important. Irrigation systems based on annual nitrogen loading, seasonal variation in evapotranspiration demands, soil conditions, operator skill, and drying periods are essential. Proper drainage to intercept water and maintain an unsaturated biologically-active soil zone will be necessary. Timely cultural operations and harvests will also be necessary. Special attention should be given to maintaining the soil in a high-infiltration-capacity condition by using crop residue, tillage, winter-cover crops, and dual-cropping systems. Daily modifications to a well-designed system based on observation will be a critical factor in the successful operation of a land-treatment system.

CROP IDENTIFICATION

Alfalfa (*Medicago sativa* L.)

Coastal bermuda grass (*Cynodon dactylon* L.)

Corn (*Zea mays* L.)

Kentucky bluegrass (*Poa pratensis* L.)

Quackgrass (*Agropyron repens* L.)

Reed canarygrass (*Phalaris arundiancea* L.)

Rye (*Secale cereale* L.)

Smooth bromegrass (*Bromus inermis* Leyss)

Tall fescue (*Festuca arundiancea* Schreb.)

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AGRICULTURE AND FOREST USE

RENOVATION OF WASTEWATER AND RESPONSE OF FOREST ECOSYSTEMS: THE PACK FOREST STUDY.

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ABSTRACT

Wastewaters from a secondary treatment facility have been irrigated on both seedlings and an established 45-year-old Douglas-fir forest year round at the rate of 5 cm/wk. Both barren and a grass plot have also been irrigated with wastewater during this period to provide comparison with another vegetative type and a non-vegetative control.

Ion fluxes within the soil and nutrient utilization by the vegetative cover were monitored monthly during this study period. Our data allow calculation of the efficiencies of these forest systems in renovating applied wastewaters.

Results from the first three years of our study demonstrate that wastewaters can be added at 5 cm/wk to these forest systems and the drainage waters from them will be well within the 10 ppm nitrate standards set by EPA for drinking water. In addition the Douglas-fir and poplar seedlings show a marked increase in production and nitrogen uptake after wastewater irrigations. For example, during the third growing season total production on the Douglas-fir seedling plot is 6-fold greater than the control and the poplar plot is 13-fold greater. This increase in growth yields a nitrogen renovation rate in excess of 80% on the seedling plots and 90% on the established forest. Phosphate renovation is complete (99.9%) on all wastewater plots.

INTRODUCTION

Application of wastewater from municipal sewage treatment facilities on forest sites is not a new approach. It holds certain advantages over other land disposal alternatives. Forest lands are abundant, comprising about a third of the land area of the United States, and generally are less costly than other land types suitable for effluent disposal. Forests are farther removed from man's food chain than are agricultural areas; consequently, the public health aspects and social acceptability of effluent application to such land may be more easily managed. Many forests, especially in the Pacific Northwest, are nitrogen deficient and adding nitrogen results in rapid nitrogen uptake and increased tree growth (Gessel _et al._, 1969). Another potential benefit has direct implications for drinking water standards since the nitrification process seems to be inhibited in some forest soils (Overrein, 1971; Cole, _et al._, 1975).

In the United States perhaps the best known of such studies are those of Kardos and Sopper (1973) in Pennsylvania and Urie (1973) in Michigan. These studies have now been complemented in the Douglas-fir region of the Pacific Northwest of the United States. This program of wastewater application by spray irrigation was started in June 1973 about 120 km south of Seattle, Washington at Pack Forest, an experimental forest of the College of Forest Resources, University of Washington. In this paper the results from the Pack Forest wastewater renovation project are synthesized and compared to those from other forest sites where the slow infiltration of wastewater has been studied. The results are discussed in three parts: the effect of wastewater application on the chemistry of the soil solution and drainage waters; growth response of the forest to these application rates; the efficiency of wastewater renovation under forest conditions.

PACK FOREST - THE STUDY SITE

Pack Forest is south of Seattle within the Puget Sound lowlands. As such it has a temperate, mild climate. Mean annual precipitation is about 100 cm, most of which is in the form of rain. A summer drought of two to three months is common. The annual temperature is about 10°C with the winter temperature remaining relatively mild (January mean temperature is 3°C).

The wastewater study site is on a glacial outwash terrace, deposited during the most recent period of glaciation about 10,000 years ago. The soil is an Everett series, a Dystric Xerochrept composed of over 90% gravel (<2 mm).

Five plot types were established in the study area: (1) an area kept free of all vegetation, (2) an area planted with Douglas-fir seedlings (*Pseudotsuga menziesii*), (3) an area planted with Lombardy poplar rooted cuttings (*Populus nigra*), (4) an area planted with grass, primarily reed canary (*Phalaris arundinacea*), and (5) an existing 45-year-old Douglas-fir forest. Naturally occurring grasses invaded the two seedling plots and remain the dominant form of vegetation on the Douglas-fir plot after three years of irrigation.

Within each vegetation plot, one subplot receives municipal wastewater and one receives water from the nearby Nisqually river. The river water subplot provides the control in the experimental design.

The wastewater is supplied by the Renton, Washington, Sewage Treatment Plant operated by the Municipality of Metropolitan Seattle (METRO). This 30 MGD treatment plant provides secondary treatment by an activated sludge process. Seventy-five percent of the inflow is estimated to be of domestic origin.

Wastewater is applied once a week in one eight-hour period. About 5 cm is sprayed on all experimental plots, using irrigation sprinklers for two minutes of each 30-minute period. Except for a few times when the pumping equipment malfunctioned or the sprinkling system froze, the barren, grass, poplar, and Douglas-fir seedling plots have been irrigated on this weekly schedule since October 1974. The 45-year-old Douglas-fir plot first received wastewater in December 1975.

Soil solutions moving through the soil profile are continuously sampled by tension lysimeters (Cole, 1968). Water flow through the profile and evapotranspiration are estimated from the outflow of containment lysimeters adjacent to the experimental plots (Fritschen *et al.*, 1973). The containment lysimeters, 2 m deep x 7.5 m diameter, have vegetative covers equivalent to those on the experimental plots and received simultaneous wastewater irrigation. The outflows from these lysimeters provide accurate estimates of the volume of soil solution flowing through the profile and of evapotranspiration from the four vegetative types. These data provide a second estimate for quantifying nutrient flux through the soil profile to the groundwater.

Further details on the field installation, methods used for the analysis of the soil solutions, and soil and vegetative materials are reported elsewhere (Breuer *et al.*, in press).

RESULTS AND DISCUSSION

Effect of Wastewater on the Chemistry of the Soil Solution and Drainage Waters.

Since the overriding reason for wastewater application to forest ecosystems is for disposal, it is critical that these systems be effective in such a role. Such systems should, as a minimum, be able to alter the chemical composition of wastewater to meet safe drinking water standards. Nitrogen and phosphorus are the two elements of greatest concern in this regard, nitrogen because of its toxicity in the nitrate form and phosphorus because of its activity in aquatic productivity. Removing these nutrients from wastewaters occurs in two ways: physical-chemical removal by adsorption, fixation, and ionic exchange, and biological removal through the plant uptake process and by denitrification.

Forest ecosystems seem to remove phosphate from wastewater rapidly and completely. The deepest that wastewater phosphate has leached on any of the wastewater irrigated plots during the first two years of irrigation, including the barren control plot, has been about 10 cm (Table 1). Less than 0.1% of the applied phosphate leached passed the 10 cm depth. Clearly this removal is physical-chemical and not biological, because removal is equally complete on the plot kept free of vegetative cover (barren plot). In addition, the adsorption isotherms demonstrate that this soil is capable of removing such levels of added phosphorus (Johnson and Cole, 1977).

Table 1. Phosphorus leaching over a 2-year period associated with wastewater (WW) and river water (RW) irrigations (kg/ha).

	Input	Depth 10cm	60cm	180cm
		Barren		
WW	195	1	<1	<1
RW	2	3	<1	<1
		Grass		
WW	188	12	<1	<1
RW	2	3	<1	<1
		Poplar seedlings		
WW	186	6	<1	<1
RW	2	1	<1	<1
		Douglas-fir seedlings		
WW	174	2	1	<1
RW	2	1	<1	<1

Sopper (1975) found similar results in the Penn State Project. After a decade of wastewater application at a 2 to 5 cm per week rate he reported removal of from 96 to 99% of the added phosphorus within the top 120 cm of soil. Urie (1973) also found the added phosphorus associated with wastewater irrigation in a Jack Pine (Pinus banksiana) forest remained largely within the top 60 cm of soil and did not reach the ground water table.

When losses of nitrogen through leaching occurred the form was almost exclusively as the nitrate ion (Table 2). This was very evident for the barren plot. In 1977 a total of 177 kg/ha of nitrogen leached past the 180 cm soil depth; 96% was in the nitrate form with less than 1% as ammonium. To a lesser extent the same relationship held true for the Douglas-fir seedling and grass sites. When the annual leaching of nitrate was observed to be at a minimum level, as in the case of the poplar seedling plot (4 kg/ha, 0.3 mg/L) the Douglas-fir forest plot (10 kg/ha, 0.7 mg/L) the leaching of nitrogen was correspondingly decreased (Table 2). Thus the leaching of nitrogen is directly linked by the nitrification process.

Table 2. Leaching losses at the 180 cm soil depth associated with wastewater irrigation during 1977.

NO_3 mg/L	NO_3 kg/ha	Total N mg/L	Total N kg/ha
	Barren		
9.8	165	10.2	171
	Grass		
5.6	95	6.1	103
	Poplar seedling		
0.3	4	0.7	10
	Douglas-fir seedling		
6.6	92	7.0	97
	Douglas-fir forest		
0.7	10	0.9	14

The monthly trends associated with nitrate leaching losses for each of the vegetative types are shown in Figure 1. These data show that the forest areas become more efficient in renovation of wastewater as the seedlings mature and occupy more of the site. This was particularly true in the case of the poplar plot where during the initial year of irrigation 47% of the applied nitrogen leached past the 180 cm soil depth resulting in a peak of 38 mg/L of nitrates in these drainage waters. In the second and third years only 5% of the applied nitrogen reached the 180 cm depth with the average concentration dropping to less than 0.2 mg/L.

Applying wastewater to the 45-year-old forest also yield excellent nitrogen renovation for the two-year period that this phase of the study has been conducted. The maximum nitrate concentrations observed in the 180 cm drainage waters for this plot has been 5 mg/L. Typically the concentrations have remained below 1 mg/L (Figure 1) and have averaged 0.7 mg/L during this two-year period.

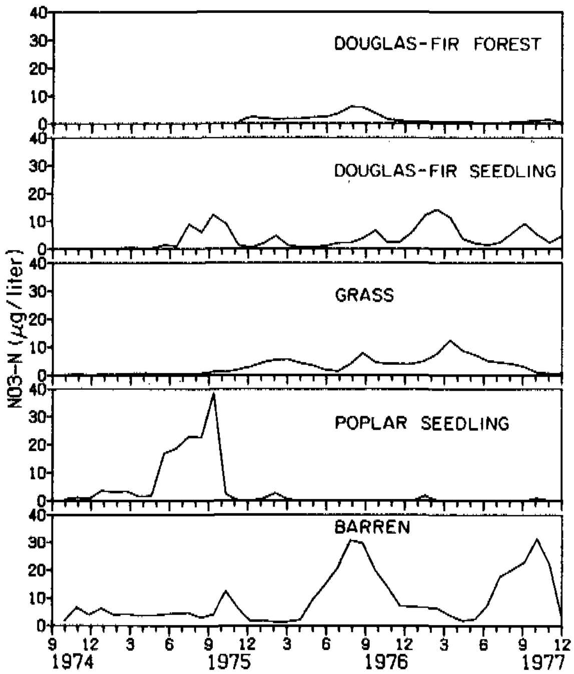

Figure 1. Nitrate-nitrogen concentration in soil solution collected at 180 cm under five different vegetation covers undergoing wastewater irrigation.

The leaching of anions, including nitrate and phosphate, through the soil profiles of these various vegetative types is illustrated in Figure 2 and tabulated in Table 3. The removal of phosphate in the surface 10 cm and the mobility of nitrate, chloride and sulphate through the soil profile is evident. There is a sharp decrease in the bicarbonate ion concentration with soil depth caused by a corresponding decrease in the soil solution pH. It can be seen from Figure 2 that the leaching of cations (Ca^{++}, Mg^{++}, K^{+}, Na^{+}) is regulated on a stoichiometric basis with the leaching of the anions. Thus any process that increases the anion concentration of the soil solution, such as nitrification, also increases the potential for losses of an equivalent amount of cations.

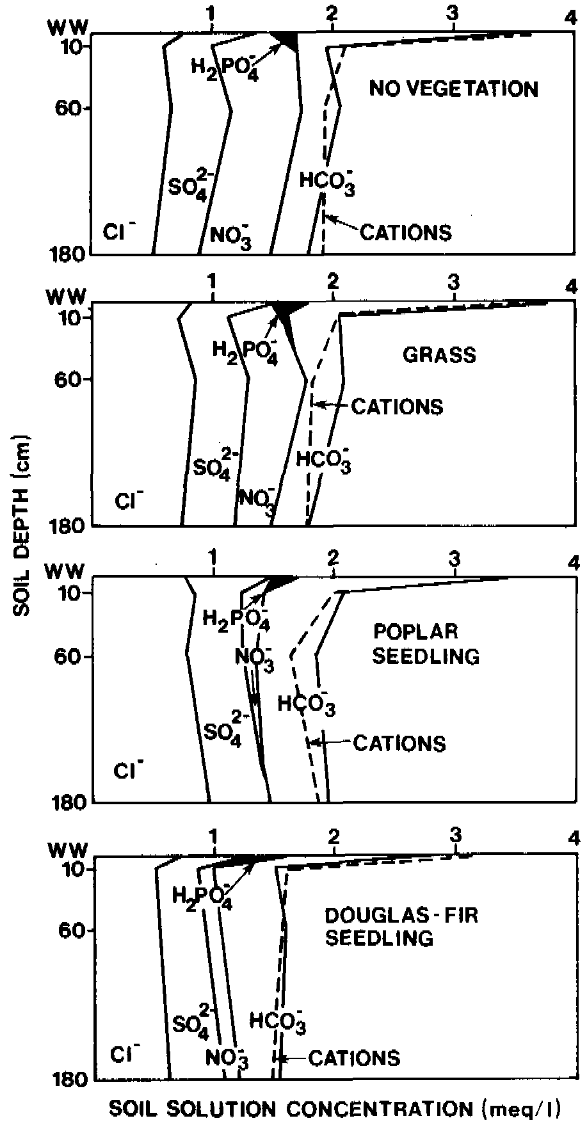

Figure 2. Anion and cation fluxes (meq/L) within the soil profile associated with wastewater irrigation at Pack Forest.

These relatively low rates of nitrate leaching following wastewater irrigation on forest sites are not always found. Sopper (1975) and Urie (1973) both reported a significant quantity (in excess of 10 ppm) of nitrate at times in leachates or groundwaters after extended periods of wastewater irrigation. This was especially evident at the 5 cm/wk application rate. This can be partially accounted for by the fact that the forest cover did not utilize nearly so much of the applied nitrogen in either of these two studies

Table 3. Ion fluxes (eq/ha) in wastewater (WW) irrigation and soil solution passing 10 cm, 60 cm, and 180 cm (October, 1975 through August, 1976).

	$H_2PO_4^-$	SO_4^{2-}	Cl^-	NO_3^-
		No Vegetation		
WW	4,435	13,085	15,086	593
10 cm	16	9,632	15,477	17,706
60 cm	3	11,998	16,759	14,755
180 cm	3	8,833	12,825	15,553
		Poplar Seedling		
WW	4,253	12,307	14,414	530
10 cm	89	8,385	18,611	4,157
60 cm	3	10,033	16,870	2,093
180 cm	4	10,862	20,685	773
		Douglas-fir Seedling		
WW	3,315	9,641	11,195	441
10 cm	22	6,006	8,783	2,639
60 cm	27	7,141	9,013	2,382
180 cm	2	7,948	10,848	2,212
		Grass		
WW	4,280	12,504	14,578	556
10 cm	334	7,979	14,030	9,284
60 cm	2	8,925	16,637	9,384
180 cm	2	8,884	14,100	5,660

as we have observed at Pack Forest. This was particularly true when wastewater was applied to the coniferous stands of these study areas. It is also possible that the nitrification process is inhibited in forest soils we have studied (Cole et al., 1975, Heilman, 1974) in that Breuer, et al. (in press) found less activity of these nitrifying bacteria associated with the lower solution nitrate levels.

Growth Response Resulting from Wastewater Applications

Wastewater renovation by forest systems partly depends on uptake and storage of the wastewater nutrients by the forest cover. Initially some removal, perhaps for the first several years, could be accomplished by soil microorganisms due to the high carbon to nitrogen ratios in these forest soils. Ultimately, however, the removal must be either by uptake by the forest cover or by denitrification. Plant uptake could be an effective mechanism for decreasing nitrogen leaching losses through these soils if the uptake rates are high enough. For trees to be effective agents in this removal process, the annual uptake of nutrients must be substantially increased over the 10 to 30 kg/ha reported for Douglas-fir on these sites (Heilman and Gessel, 1963, Cole et al., 1967).

In order to achieve a nutrient uptake rate more in line with the rates of application, it is necessary to realize a marked increase in productivity on the wastewater-treated sites. The extent of the growth response after three growing seasons is shown in Table 4. Poplar cuttings showed the greatest response to wastewater (18-fold) followed by grass (10-fold) and Douglas-fir seedlings (3-fold). By including the subordinate vegetation, primarily grass, into the production calculations a more uniform response picture emerges between these three plots. This gives the poplar plot a total production of 13 times the wastewater control, grass 10 times the control, and the Douglas-fir seedling six times the control. The difference in grass production on the poplar and Douglas-fir seedling plots is due primarily to shading by the seedlings although competition for nutrients may be a factor in the poplar plot. After three years of growth the poplar cuttings are more than 7 meters tall while the Douglas-fir seedlings are about 3 meters tall.

Table 4. Aboveground biomass production by vegetation irrigated with wastewater (WW) and river water (RW) (kg/ha).

	1976 Age 2 Yr		1977 Age 3 Yr	
	WW	RW	WW	RW
		Poplar		
seedling	8540	1020	28170	1220
grass	8000	1600	5970	120
total	16540	2620	34140	2340
		Douglas-fir		
seedling	2860	930	4910	1620
grass	8500	1600	8440	760
total	11360	2530	13350	2380
		Grass		
grass	8500	3000	9130	910

This increase in biomass production is reflected in a substantially greater nutrient uptake in the wastewater-treated sites (Table 5). Nitrogen uptake was increased 10-fold on the poplar plot, 5-fold on the Douglas-fir seedling plot, and 12-fold on the grass plot. A maximum nitrogen uptake of 303 kg/ha was observed in 1977 on the poplar plot. This is nearly three times larger than the maximum nitrogen uptake value reported in the literature for forest species and accounted for 62% of the applied nitrogen. Sixty-seven percent of the applied nitrogen was accounted for by uptake in the Douglas-fir seedling plot while only 37% of the applied nitrogen could be accounted for on the grass plot during the same period.

Table 5. Nitrogen and phosphorus uptake by three-year-old vegetation irrigated with wastewater (WW) and river water (RW)(kg/ha).

Plot Component	Nitrogen		Phosphorus	
	WW	RW	WW	RW
		Poplar Seedlings		
Seedling	213	15	42	2
Grass	90	16	21	2
Total	303	31	63	4
		Douglas-fir Seedlings		
Seedling	43	16	7	2
Grass	181	30	31	3
Total	224	46	38	5
		Grass		
Grass	179	15	36	2

Biomass production and nutrient uptake information will not be available from the 45-year-old Douglas-fir plot until the end of the 1978 growing season. However, preliminary observations of these older trees indicate that they are also experiencing a substantial increase in growth and nutrient uptake.

Sopper (1975) also noted a significant tree growth response in his Penn State Project. The greatest increases he observed were with hardwood species rather than conifers: red pine (Pinus resinosa) and white spruce (Picea glauca). However, the response he reported was not as great as we noted at Pack Forest; thus Sopper found substantially less nitrogen and phosphorus uptake. For example, the poplar stand at Pack Forest removed 212 kg/ha of nitrogen during its third year of growth while an established hardwood forest in the Penn State study removed only 94 kg/ha.

Efficiency of Wastewater Renovation under Forest Conditions

The capacity of a forest ecosystem to utilize and store nutrients associated with wastewater irrigations has been clearly demonstrated at Pack Forest. Viewed in terms of wastewater renovation efficiencies, we can also see that these forested systems are functioning effectively. The wastewater renovation capacity of the four vegetative types and barren control at Pack Forest during the past three years is shown in Table 6. These data clearly show that all of the vegetative covers provide substantially greater renovation of wastewater than the barren control. For this three-year period, an average of 84% of the nitrogen was removed by the vegetated plots through both soil retention and plant uptake. Without any vegetative cover the removal percentage was substantially less, an average of 56% over the three-year period. By examining the yearly trends in these data (Table 6) we see that an established forest cover increases this renovation capacity. For example, wastewater applications to the 45-year-old forest resulted in a nearly complete removal of nitrogen (94%). Applications on the poplar plot show a similar removal efficiency after the poplar became established in 1976 and fully occupied the site. Because of initial slower growth, Douglas-fir seedlings have not yet fully occupied the plot after three years of growth. Consequently this site still has a major component of grass, 64% of the 1977 production. Since the Douglas-fir are now about 3 meters tall on the wastewater-treated site, we would expect that this area will be essentially dominated by trees at the end of the 1978 growing season. This should in turn cause an increase in the renovation capacity, bringing this plot more in line with the poplar and 45-year-old forested sites.

Monthly patterns of renovation by each vegetative type and the barren control over this three-year period are illustrated in Figure 3. A pattern of seasonality emerges from this analysis.

Table 6. Addition (kg/ha) and renovation (%) of nitrogen associated with wastewater irrigation at Pack Forest.

1975 kg/ha	%	1976 kg/ha	%	1977 kg/ha	%	Total kg/ha	%
Barren							
404	69	438	34	489	65	1331	56
Grass							
404	93	433	77	480	79	1317	83
Poplar Seedling							
404	53	429	95	406	98	1239	82
Douglas-fir Seedlings							
404	85	320	83	333	71	1057	80
Douglas-fir Forest							
--*	--*	400	88	429	97	829	92

*wastewater application did not begin until December 1975.

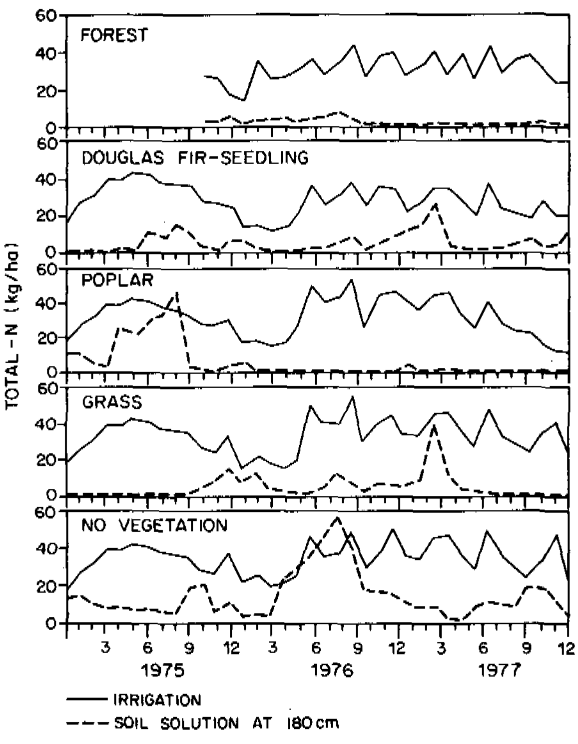

Figure 3. Nitrogen addition by wastewater irrigation and loss through deep leaching (180 cm) for the five vegetative types at Pack Forest.

The greatest loss of nitrogen occurs from the barren site from May to October when the soil temperatures are highest. This loss coincides with the period of greatest nitrification (Figure 2). In contrast nitrogen renovation on the vegetated plots was lowest during the winter and early spring periods. This was not unexpected because nitrogen uptake by these systems is probably maximum during summer and fall. Where very high rates of nitrogen uptake were found, such as in the case of the poplar and forest plots, all indications of seasonality in renovation were lost.

The removal of nitrogen from the wastewater irrigation in the 45-year-old forest could be related to the microorganisms. The high carbon content and wide C/N ratio associated with the forest floor and surface A horizon of this established forest theoretically provide a storage potential of over 500 kg/ha of nitrogen. Although this is not a large storage reservoir compared to the annual additions by wastewater, it is sufficient to provide a storage buffer during the initial year before rapid growth response.

The renovation efficiency for nitrogen by the tree species tested in the Penn State Project (Sopper, 1975) is not as encouraging. This could be partly due to the testing of some slower growing species such as red pine and also to the harsh winter climate of that site. Sopper (p. 233) concluded that "trees are not as efficient renovating agents as agronomic crops." Our results at Pack Forest suggest that this conclusion may be invalid for species and sites having greater growth potential.

CONCLUSION

The study at Pack Forest and those in Pennsylvania and Minnesota demonstrate that forest ecosystems can play a major role in renovating municipal wastewater. There are several advantages in using forests for this purpose. The results from the Pack Forest project clearly show that forest ecosystems can be as effective in renovation as agricultural crops. Renovation rates comparable to agricultural crops have been achieved. It was also found that wastewater could be applied to the forest sites at 5 cm/wk on a year-round basis and still maintain EPA drinking water standards for solutions passing 180 cm depth.

The key to accomplishing this renovation of wastewater seems to be a high nitrogen recovery rate and growth response of the vegetative cover. The mild winter climate at the Pack Forest site undoubtedly was an important factor in allowing these year-round irrigations. Using a perennial plant, such as a tree, is also critical in this regard.

Poplar appears to be an ideal recipient of wastewater because of its rapid early growth and high uptake rates of nutrient elements. Douglas-fir seedlings take longer to occupy a site initially, but such a rapid growing coniferous species might ultimately be a better choice because of its capacity to transpire water and take up nutrients in the winter months, as well as its higher final market value.

The outstanding response of trees to these wastewater applications leads us to speculate on the feasibility of combining wastewater management with forest management. Such a prospect is further enhanced by the current interest in utilizing forests as energy plantations to help meet the world energy needs.

ACKNOWLEDGMENTS

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UTILIZATION OF DOMESTIC WASTEWATER IN FOREST ECOSYSTEMS THE PENNSYLVANIA STATE UNIVERSITY LIVING FILTER PROJECT

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ABSTRACT

Long-term nitrate-N renovation efficiency of several forest ecosystems on the Penn State Renovation and Conservation Project were evaluated. During the past 15 years (1963-1977) secondary treated municipal sewage effluent was spray irrigated in mixed hardwood forests, a red pine (Pinus resinosa), and a white spruce (Picea glauca) plantation established on an abandoned old field. Effluent was applied at rates ranging from 2.5 to 7.5 cm per week over various lengths of time from the growing season to the entire year. Soil water samples were collected weekly at the 120 cm depth with suction lysimeters and analyzed for selected nitrogen forms. Results indicated that satisfactory wastewater renovation was achieved when mixed hardwood forests and red pine plantations were irrigated with sewage effluent at 2.5 cm per week during the growing season. An old field-white spruce ecosystem irrigated at 5.0 cm per week during the growing season also provided satisfactory renovation. A red pine plantation, spray irrigated with effluent at 5.0 cm per week during the growing season did not provide adequate wastewater renovation. However, when the red pine plantation was replaced by a first-stage succession of herbaceous vegetation and pioneer shrub and tree species, renovation efficiency was restored. Year-around irrigation of a mixed hardwood forest ecosystem on a sandy loam soil at an application rate of 5.0 cm per week did not provide adequate wastewater renovation. Increasing the sewage effluent application rates by 50 percent after 9 years of operation resulted in the collapse of all forest ecosystem renovation systems. Recovery of these systems was extremely slow requiring 1 to 3 years. Recovery was most rapid when wastewater applications were terminated completely rather than continued at a lower application rate.

INTRODUCTION

The fate of nitrogen in one of the most important aspects which must be considered in regard to land application of municipal wastewater. This is particularly true for "Living Filter" systems where the primary goal is renovation of wastewater for direct recharge of the groundwater reservoir. Renovation, in terms of nitrogen, means that the concentration of nitrate-N in soil water leaving the root zone should not exceed 10 mg/l (U.S. Public Health limit for potable water).

Under the "Living Filter" concept, the renovation of wastewater is achieved through the removal and degradation of organic and inorganic constituents by microorganisms, chemical precipitation, ion exchange, biological transformation, and biological absorption through the root systems of the vegetative cover. Most ecosystems have an initial high capacity to accept and renovate wastewater satisfactorily. However, the continual application of large volumes of water with high amounts of nutrients will ultimately affect ecosystem stability and all renovation processes. Most of these changes occur slowly and may not become obvious for a decade. Some of the more important changes that may influence wastewater renovation efficiency in forest ecosystems are vegetation species composition and density, foliage chemical composition, leaf litter and forest floor decomposition rates, humus mineralization rates, physical and chemical properties of the soil, microbial and earthworm populations, and microclimate.

Wastewater renovation has been intensively investigated in several forest ecosystems as a part of the Penn State Wastewater Renovation and Conservation Project. Earlier results have been reported by Pennsypacker et al (1967), Kardos and Sopper (1973), Hook and Kardos (1977, 1978), Parizek et al (1967), and Sopper and Kerr (1978). The purpose of this report is to summarize the results of these earlier studies and to discuss the long-term effects of spray irrigation of sewage effluent on the renovation efficiency of several forest ecosystems.

During the past 15 years (1967-1977) secondary treated municipal sewage effluent from the University wastewater treatment plant has been spray irrigated in several forest ecosystems. The ecosystems included two mixed hardwood forests, a red pine (Pinus resinosa) plantation, and a sparse white spruce (Picea glauca) plantation established on an abandoned old field. Detailed descriptions of these areas have been previously reported by Parizek et al (1967). The two soils present are ultisols. The Hublersburg soil (Typic Hapludalf) has a silt loam topsoil with a silty clay or clay subsoil. It has moderate to moderately rapid permeability and is well drained. The Morrison soil (Ultic Hapludalf) is a sandy loam with a sandy clay loam subsoil and has a moderately rapid to rapid permeability and is well drained.

Application rates ranged from 2.5 to 7.5 cm per week over various lengths of time ranging from the growing season (April to November) to the entire year. The normal weekly application rates were 2.5 and 5.0 cm per week (0.63 cm per hour), throughout the study period except for 1972 and 1973. During these two years the application rates were increased by 50 percent to evaluate the effects of chronic applications of wastewater on ecosystem collapse and recovery. During the period 1963 to 1977, the total amount of sewage effluent applied was 1064 cm on the areas irrigated at 2.5 cm per week and 2090 cm on the areas irrigated at 5.0 cm per week during the growing season on the Hublersburg soil. The hardwood forest on the Morrison soil which was irrigated at 5.0 cm per week the entire year received a total of 2362 cm during the period 1965 to 1977.

The mean concentration of nitrogen forms in the sewage effluent is given in Table 1.

Table 1. Mean Concentration of Nitrogen Forms in the Sewage Effluent

	mg N/l
NO_3-N	9.4
NH_4-N	8.5
Org-N	4.6
Total-N	22.5

There has been a change in the concentrations of nitrogen forms over the 15 year period and this change must be considered in relation to the renovation process. In 1970, the sewage treatment plant modified its operation to lower nitrogenous-biological oxygen demand of the effluent since only 20 percent was being applied on land and 80 percent was being discharged into a local stream. As a result there was a shift in the predominant nitrogen form from ammonium to nitrate. Such a shift could influence the amount of nitrate leaching, particularly during the winter. During winter irrigation ammonium added should be held on exchange sites of soil colloids and with low nitrification, should accumulate in the soil. Nitrate would continually leach during the winter. During the summer months, the exchange phenomena should hold ammonium near the soil surface as water percolates downward during irrigation, carrying much of the nitrate with it. It is possible that the ammonium held in the surface soil layer might be converted to nitrate and then would be subject to leaching with irrigation the following week. However, in any event, this nitrogen would remain in the topsoil for a longer period of time which would prolong its availability for plant uptake, microbial assimilation, or denitrification before the next irrigation. While mean annual total N concentration was almost constant, there was a distinct seasonal cycle in mean monthly total N concentrations (Hook and Kardos, 1978). Mean monthly total N concentrations ranged from a low of 18 mg/l in August to a high of 35 mg/l in March. The increases during the winter months were primarily due to ammonium increases and were probably in response to changes in the student population served by the sewage treatment plant.

METHOD OF STUDY

Samples of the sewage effluent were collected in each forest ecosystem during each irrigation period using rain gauges or plastic containers placed 1 m above the ground. Suction lysimeters were installed in all areas at the 120 cm soil depth to obtain samples of percolating soil water. Soil water samples were collected, whenever possible, after each weekly application of wastewater. Nitrate-N was determined by the brucine method (APHA, 1965a) from 1963 to 1969 and by the ion-selective electrode method (Milham et al, 1970) from 1970 to 1976. Ammonium was measured by direct Nesslerization method (APHA, 1965b) from 1963 to 1972 and by ammonia gas electrode (Orion, 1971) from 1973 to 1976. Organic N was determined by a semimicro-kjeldahl digestion followed by direct Nesslerization (Kelley et al. 1946) or by ammonia gas electrode.

RESULTS AND DISCUSSION

Nitrogen

The efficiency of the forest ecosystems to renovate the wastewater and to reduce nitrogen concentrations in soil water has been variable. Mean annual concentrations of nitrate-N in soil water at the 120 cm soil depth for all forest ecosystems during the period 1965 to 1977 are given in Table 2. Data for 1963 and 1964 are not presented since soil water samples were only collected at the 30 and 60 cm depths. However, nitrate-N concentrations in soil water at these shallower depths were all equal to or less than the values reported for 1965 at the 120 cm depth.

Forest Ecosystems on Hublersburg Soil

Mixed Hardwood Forest. Sewage effluent was irrigated in this ecosystem during each growing season at the application rate of 2.5 cm per week throughout the study period (1963-77) except for 1972 and 1973 when the application rate was increased to 3.8 cm per week. Results presented in Table 2 indicate that this forest ecosystem was able to satisfactorily renovate the wastewater. The mean annual concentrations of nitrate-N in the soil water were consistently below the U.S. Public Health Service (USPH) limit for potable water (10 mg/l) during the first nine years of operation (1963-1971). However, there appears to be a decreasing trend in renovation efficiency starting in 1968 after the first six years of operation. This may be related to other ecosystem changes reported by Richenderfer and Sopper (1978) and Dindal et al (1978). Richenderfer and Sopper found that sewage effluent irrigation caused a significant increase in forest floor decomposition rates, and, thus, higher humus mineralization rates. Dindal reported significant increases in microbial and earthworm populations. All of these changes would tend to favor higher leaching of nitrate-N. The data in Table 2 also indicates that forest ecosystems may be quite sensitive to wastewater application rates and may have a low threshold of collapse in terms of renovation. Ecosystem collapse being defined as that point in time when the ecosystem is no longer able to remove nitrogen and the concentration of nitrate-N in soil water at the 120 cm depth exceeds 10 mg/l. In 1972, when the application rate was increased from 2.5 to 3.8 cm per week, the forest ecosystem renovation system collapsed and the soil water mean annual nitrate-N concentration increased significantly to 23.9 mg/l. However, it should also be noted that some of this increase in nitrate-N leaching was due to tropical storm "Agnes." During the period June 21-25, 1972 there was a total of 24.05 cm of rain. This large amount of rainfall also significantly increased the mean annual concentration of nitrate-N in the soil water of the control forest (0.1 to 4.7 mg/l). The increased application rate was applied again in 1973 and the results obtained are more indicative of what might happen under normal climatic conditions. Wastewater renovation was still unsatisfactory,

Table 2. Mean Annual Concentration of Nitrate-Nitrogen in Soil Water at the 120 cm Depth in the Forest Ecosystems.

Year	Red Pine I Hublersburg Soil cm per week		Red Pine II Hublersburg Soil cm per week		Hardwood Hublersburg Soil cm per week		Old Field Hublersburg Soil cm per week		Hardwood Morrison Soil cm per week	
	0	2.5	0	5	0	2.5	0	5	0	5
	mg/1	mg/1	mg/1	mg/1	mg/1	mg/1	mg/1	mg/1	mg/1	mg/1
1965	0.9	2.2	0.9	3.9	--	0.0	0.3	8.0	--	--
1966	0.1	2.1	0.1	9.3	0.1	0.2	0.1	5.0	0.1	10.6
1967	0.9	1.7	1.8	13.8	0.3	1.4	0.3	6.1	1.4	19.2
1968	0.9	2.7	1.6	20.0	0.1	8.0	0.2	3.7	0.1	25.9
1969	0.2	4.2	0.5	24.2	0.1	7.2	0.2	2.3	0.3	23.7
1970	<1	5.3	<1	8.1	<1	5.0	<1	3.5	1.0	42.8
1971	2.6	8.3	2.6	2.1	0.5	5.8	0.5	3.8	0.8	17.6
1972	6.0	21.8*	6.0	14.5†	4.7	23.9*	3.2	11.8†	4.7	22.9
1973	0.5	13.7*	0.5	8.7†	3.0	14.7*	0.5	13.5†	1.3	17.3
1974	0.7	16.1	0.7	7.8	1.5	14.5	0.5	10.9	0.5	14.3
1975	1.3	11.9	1.3	5.1	1.7	11.6	0.8	12.9	0.8	9.0
1976	0.7	9.8	0.7	4.3	1.2	12.5	0.8	8.4	0.6	4.8
1977	0.7	5.0	0.7	3.7	1.4	9.4	0.9	6.9	1.1	5.1

*Application rate increased to 3.8 cm per week.

†Application rate increased to 7.5 cm per week.

however, the mean annual nitrate-N concentration in soil water was only 14.7 mg/l. During the period 1974 to 1977 the application rate was reduced to 2.5 cm per week. It is quite obvious that ecosystem recovery after chronic applications of wastewater is extremely slow. Mean annual nitrate-N concentration remained above 10 mg/l until 1977 when it decreased to 9.4 mg/l.

Red Pine I. Sewage effluent was irrigated in this ecosystem during each growing season at the application rate of 2.5 cm per week throughout the study period except for 1972 and 1973 when the application rate was increased by 50 percent. Results presented in Table 2 indicate that this forest ecosystem was able to satisfactorily renovate the wastewater. The mean annual concentrations of nitrate-N in the soil water were consistently below 10 mg/l during the entire period 1963 to 1971. Similar to the mixed hardwood forest there appears to be a decreasing trend in renovation efficiency starting in 1969. As in the mixed hardwood forest, the explanation for this decreasing renovation efficiency may be the ecosystem changes previously discussed. Increasing the application rate in 1972 resulted in a similar collapse of the ecosystem renovation processes and the mean annual nitrate-N concentration in the soil water significantly increased to 21.8 mg/l. During the period 1974 to 1977 the application rate was reduced to 2.5 cm per week. Ecosystem recovery was extremely slow with nitrate-N concentrations remaining above 10 mg/l until 1976 when it decreased to 9.8 mg/l.

Red Pine II. Sewage effluent was irrigated in this ecosystem during each growing season at the application rate of 5.0 cm per week throughout the study period except for 1972 and 1973 when the application rate was increased by 50 percent. Renovation results are given in Table 2. Results indicate that this forest ecosystem was only able to satisfactorily renovate wastewater during the first four years of operation (1963 to 1966). Mean annual concentration of nitrate-N in soil water gradually increased from 3.9 to a peak of 24.2 mg/l in 1969. In November of 1968, a snow storm accompanied by high winds resulted in a complete blowdown of the plantation. In 1969 the area was clearcut and all trees were removed. Sewage effluent irrigation was continued and immediately a dense cover of herbaceous and shrub vegetation developed. With the development and growth of this perennial herbaceous vegetative cover, the mean annual concentration of nitrate-N in the soil water decreased from 24.2 mg/l in 1969 to 8.3 mg/l in 1970 and to 2.9 mg/l in 1971. Increasing the application rate from 5.0 to 7.5 cm per week in 1972 increased the nitrate-N concentration in the soil water to 14.5 mg/l. Even though this concentration exceeds 10 mg/l, it is obvious that the new developing ecosystem composed of invading pioneer species of herbaceous and tree vegetation was very efficient in renovating wastewater. In 1972, tropical storm "Agnes" increased the mean annual concentration of nitrate-N in the soil water on the control area from 2.6 to 6.0 mg/l. On the irrigated area, in 1973, with the increased application rate under normal climatic conditions, the nitrate-N concentration in soil water was only 8.7 mg/l. It is obvious that the pioneer vegetation ecosystem was able to recovery quickly from the chronic applications of wastewater and that the renovation efficiency of the new ecosystem is much greater than the original red pine ecosystem. Even though the red pine ecosystem was very inefficient in renovating the wastewater during the period 1963 to 1969, reaching a peak concentration of 24.2 mg/l of nitrate-N in soil water, the new developing pioneer vegetation ecosystem was extremely efficient as indicated by the fact that mean annual nitrate-N concentration in the soil water was only 3.7 mg/l in 1977. This is quite significant when one considers that almost 21 meters of sewage effluent had been applied on this area as of 1977. The results obtained from this area dramatically illustrate the interrelationship between the application rate, the type of vegetation, and the system of management.

Old Field-White Spruce. Sewage effluent was irrigated in this ecosystem during each growing season at the application rate of 5.0 cm per week throughout the study period except for 1972 and 1973 when the application rate was increased by 50 percent. Nitrogen was added at an average rate of 310 kg N/ha per year. In 1972 and 1973, 430 kg N/ha per year was added.

Renovation results are given in Table 2. This ecosystem has been somewhat exceptional in terms of nitrogen renovation in comparison to the other forest ecosystems. It received the highest application rate on the Hublersburg soil and yet has consistently maintained nitrate-N concentrations in soil water below 10 mg/l throughout the period 1963 to 1971. Increasing the application rate in 1972 resulted in a slight increase in concentrations of nitrate-N in the soil water. However, the ecosystem quickly recovered. Average leaching losses of N at the 5.0 cm per week irrigation rate was 109 kg N/ha per year or 35 percent. At the increased irrigation rate (7.5 cm per week) the leaching loss of N was 323 kg/ha/yr or 75 percent. In 1963 at the start of the project the area was primarily an open field with a few scattered white spruce saplings (1 to 2 meters in height). Although the trees are now more than 9 m in height, the spruce stand is still sparse with fairly large open areas occupied by perennial herbaceous vegetation and shrubs. It appears that the annual and perennial vegetation that occupy these open areas during the growing season (irrigation period) provide temporary storage for nitrogen and hence reduce the nitrate-N leaching losses. In the fall, vegetation growth ceases and nitrate-N is again available for leaching. However, since irrigation has ceased by this time, the concentration of nitrate-N in soil water remains at an acceptable level. The desynchronization effect of nitrate-N application, vegetation utilization, soil water leaching is one of the primary factors contributing to the renovation efficiency of this ecosystem.

Forest Ecosystem on Morrison Soil

<u>Mixed Hardwood Forest</u>. Sewage effluent was irrigated in this ecosystem during the entire year at the rate of 5.0 cm per week (0.42 cm per hour) throughout the period 1965 to 1974. Renovation results are given in Table 2. Results indicate that irrigation throughout the entire year of forest ecosystems on sandy soils with sewage effluent is not feasible. Mean annual concentration of nitrate-N in soil water continually increased reaching a peak within five years (1970) of 42.8 mg/l. During the study period unknown amounts of liquid digested sludge were periodically injected into the sewage effluent and spray irrigated on the area. These sludge applications probably account for the unexplained fluctuations in the mean annual nitrate-N concentrations in the soil water during the period 1968 to 1971. The increase in nitrate-N concentration in soil water in 1972 was partially the result of tropical storm "Agnes." At the end of the growing season in 1974 it was decided to cease sewage effluent irrigation in this forest ecosystem and to evaluate the rate of ecosystem recovery in terms of nitrate-N renovation efficiency. Within one year the mean annual concentration of nitrate-N in soil water decreased from 14.3 mg/l to 9.0 mg/l. In 1976 and 1977 sewage effluent was again applied during the growing season at the rate of 5.0 cm per week. Results indicate that the forest ecosystem is again providing satisfactory renovation of the wastewater. Nitrate-N concentrations of percolating water at the 120 cm soil depth were below 10 mg/l (4.8 mg/l in 1976 and 5.1 mg/l in 1977).

CONCLUSIONS

Several forest ecosystems were irrigated with treated municipal wastewater over a 15-year period. Wastewater application rates ranged from 2.0 to 7.5 cm per week and were applied over various lengths of time ranging from the growing season to the entire year. Soil water samples were collected at the 120 cm soil depth for chemical analyses. Results of these analyses were used to determine the long-term renovation efficiency of each forest ecosystem. The results reported indicate the following conclusions:

1. The mixed hardwood and red pine forest ecosystems on the Hublersburg soil that were irrigated during the growing season at 2.5 cm per week for nine years (1963-1971) provided satisfactory renovation of the wastewater. The mean annual concentration of nitrate-N in soil water at the 120 cm soil depth never exceeded the U.S. Public Health limit of 10 mg/l for potable water.

2. The old field-spruce ecosystem on the Hublersburg soil that was irrigated during the growing season at 5.0 cm per week for nine years (1963-1971) provided satisfactory renovation of the wastewater. The average annual concentration of nitrate-

N in soil water at the 120 cm soil depth never exceeded 10 mg/l.

3. The red pine forest ecosystem on the Hublersburg soil that was irrigated during the growing season at 5.0 cm per week for nine years (1963 to 1971) did not adequately renovate the wastewater. Average annual concentrations of nitrate-N exceeded 10 mg/l by 1967 and reached a maximum of 24.2 mg/l in 1969.

4. Conversion of the red pine forest, following blowdown in 1968, to a first-stage succession of herbaceous vegetation and pioneer shrub and tree species resulted in restoration of renovation efficiency. Within one year the average annual concentration of nitrate-N in soil water decreased from 24.2 to 8.3 mg/l and to 2.1 mg/l by the second year.

5. Year-around irrigation of the mixed hardwood forest ecosystem on the Morrison soil at 5.0 cm per week did not provide adequate renovation of the wastewater. Within one year the average annual concentration of nitrate-N in the soil water exceeded 10 mg/l and peaked at 42.8 mg/l within 5 years. These results indicate that it is not feasible to irrigate mixed hardwood forests with municipal wastewater at an application rate of 5.0 cm per week on a continuous year-around basis.

6. Results indicated that a 50 percent increase in the weekly application rate (from 2.5 to 3.8 and 5.0 to 7.5 cm per week) resulted in a collapse of the forest ecosystem renovation system in all areas on the Hublersburg soil.

7. Recovery of the ecosystem renovation systems following chronic applications of wastewater is extremely slow, requiring one to three years. Recovery is the most rapid if wastewater applications are terminated rather than continued at a lower application rate.

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AGRICULTURE AND FOREST USE

INFLUENCE OF IRRIGATION WITH CLARIFIED CATTLE FARM OUTFLOW ON LUCERNE CROPS

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Kupavna, Ministry of Amelioration and Water Utilization, All Union Scientific Research Institute for Utilization of Waste Water in Agriculture, 1975.

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The conversion of animal husbandry to an industrial base in Belgorodskaya Oblast was undertaken in 1964. In 1974 there were 20 hog breeding complexes, 9 young cattle offspring growing and feeding complexes, and 20 dairy farms. The construction of about 40 dairy complexes is nearing completion. The successful operation of animal husbandry complexes largely depends on the utilization of liquid manure, i.e., improperly stored and utilized, clarified outflow could become a source of pollution of surface and ground water.

One of the efficient methods for the utilization of animal husbandry outflow is its use for irrigating farm crops, perennial grasses in particular. In Belgorodskaya Oblast lucerne is the main crop used in fodder crop rotation, accounting for about 60% of the area. Compared with other leguminous grasses, lucerne contains more digestible protein in its green mass and in a well-prepared protein-vitamin grass meal, hay, and haylage. It is rich in macro- and microelements and in many other vitamins needed by the animals.

With suitable agrotechnology lucerne yields high crops. A total of 700 to 1000 quintals of green mass or 150 to 200 quintals of hay may be obtained per hectare of irrigated land with 4 to 7 or more mowings. Lucerne growing has a positive agrotechnical importance. With high yields it has a favorable impact on soil properties. In order to obtain high and stable lucerne yields it would be expedient, along with the vegetation irrigation, to have autumn or early spring water supply irrigation on droughty areas. Such irrigation contributes to the lucerne's fast growth and development.

Until recently studies were concentrated essentially on the doses, types, and times of application of chemical fertilizers and manure in lucerne crops while the question of irrigating the lucerne with clarified outflow was totally ignored. The purpose of this study was to determine the optimal norm of application of clarified outflow in lucerne growing. The experiment was organized in 1972 at the kolkhoz imeni Sverdlov, Belgorodskaya Oblast.

The lucerne (Pavlovskaya-7 strain) was planted in the autumn of 1972 with a SZTN-46 grain seed drill; the seeding rate was 14 kg per hectare planted at a 2-cm depth. Following the planting the soil was rolled. In the period of the germination of the seed and the appearance of the sprouts the crop was cultivated with a light harrow to break up the soil crust.

The content of the nutritive substances in the clarified cattle outflow used for irrigation, with a 99.5% moisture, was as follows: general nitrogen, 0.03%; phosphorus, 0.015%; and potassium, 0.05%. Table 1 shows the rate and times of lucerne irrigation with waste water by experiment year.

According to Table 2 superphosphate in variants Nos 5 and 6 was applied at the beginning of the vegetation period. The irrigation of the experiment sector was conducted with an ANZh-2 machine using flexible capron hose. The experiment was repeated three times. Plots were chosen at random, covering areas of

Table 1. Basic Lucerne Irrigation System Indicators.

Year of study	Irrigation norm, m^3/ha IV	V	VI	VII	VIII	IX	Irrigation period, days	Sprinkling norm, m^3/ha
1973	400	400	800	350	350	500	123	2800
1974	400	400	800	350	350	500	123	2800

Table 2. Influence of Clarified Cattle Outflow on Lucerne Crop (green mass).

N	Variant	Content in outflow and chemical fertilizers, kg			Yield, quintals/ha		Average yield,	Additional yield from fertilizing and irrigation	
		N	P_2O_5	K_2O	1973	1974	quintals/ha	quintals/ha	%
1	Control (no irrigation	-	-	-	389	409	399	-	100
2	Clarified outflow 2800 m^3/ha	840	420	1400	740	702	721	322	180
3	Clarified outflow 2000 m^3/ha	600	300	1000	650	646	648	249	162
4	Ibid, 1500 m^3/ha	450	225	750	649	602	626	227	156
5	Ibid, 1400 m^3/ha + water, 1400 m^3/ha + 180 kg P_2O_5	420	390	700	625	617	621	222	158
6	Clarified outflow, 1400 m^3/ha + water 1400 m^3/ha	420	210	700	610	603	606	207	151
7	Clarified outflow, 700 m^3/ha + water 2100 m^3/ha+90 kg P_2O_5	210	195	350	612	600	606	207	151
8	Clarified outflow 600 m^3/ha	180	90	300	490	529	509	110	131
9	Clarified outflow 300 m^3/ha + water 2500 m^3/ha	90	45	150	535	559	547	148	124
	HCP 0.05, quintals/ ha				23.0	30.0			

120 square meters with a control area of 50 square meters. The full experiment system is shown in Table 2.

In the course of the experiment weather conditions varied greatly: in 1973 rain precipitation in the vegetation period was about 280 mm, compared with 324 mm in 1974. Precipitation was uneven. The June-July 1974 precipitation accounted for 62% of the annual amount. Within that period the average daily above-freezing temperatures ranged between 24.5 and 26°C. The plots were mowed 3 times annually.

Observations indicated that the lucerne particularly needs moisture and uses more of it in its growth period (in spring and after mowing) and in the course of its intensified growth, usually taking place before and during the bud formation period. During the vegetation period the greatest need for water was observed in the first half of the summer. Toward the autumn moisture outlays declined as a result of the lower evaporation from the soil surface and the lucerne plants. The initial irrigations took place at the end of

April and ended with the irrigation season in September; subsequent ones were applied 4-5 days after mowing. Irrigation rates ranged from 350 to 500 cubic meters per hectare.

Water supply irrigation at the rate of 500 cubic meters per hectare was applied every year to accumulate moisture at the beginning of the vegetation (in September). Along with improvements in the water regime of the soil the lucerne was supplied with easily accessible nutritive substances from irrigation outflow. Thus, in the second variant the amount of nutritive substances applied (N, P_2O_5, and K_2O) reached, respectively, 840, 420 and 1400 kg per hectare.

The study of the lucerne green mass crop by year indicated that with all doses reliable additions compared with the control variant (Table 2) were achieved. The green mass yield treated with clarified outflow in the experimental variant for the second year averaged between 399 and 721 quintals per hectare. The highest yield was obtained in the variant with a maximal waste water application rate (2800 cubic meters per hectare), reaching 721 quintals. Diluting this amount with an equal amount of water lowered the yield to 606 quintals per hectare, while an 800% dilution lowered it to 547 quintals per hectare. This way the use of animal husbandry outflow in lucerne growing enables us to use huge volumes of liquid manure and obtain high guaranteed fodder yields regardless of weather conditions, and protect the surrounding territory and nearby water bodies from pollution.

AGRICULTURE AND FOREST USE

THE USE OF WASTEWATER FOR IRRIGATION IN THE UKRANIAN SSR (ABSTRACT)

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A large part of the Ukranian territory is located in the zone of insufficient precipitation. Therefore, in order to ensure a guaranteed production of grain, artificial irrigation is performed. One use of the irrigation process is wastewater from cities and animal farms.

In 1977 wastewater was used for irrigation on more than 70,000 ha. City wastewater mixed with industrial wastewater was subjected to full biological purification before application to the land. Using this technique the wastewater does not have a large fertilizing value, and is used primarily during the harvest period for irrigation of agricultural crops, and during the fall-winter periods water runoff is allowed to enter surface water. Due to the large temperature fluctuations in the Ukranian climate and the absence of irrigatable lands in the proximity of the cities, the wastewater runoff must be collected.

The collected residential effluents are used to irrigate 61,000 ha. The largest irrigation system is the BORTNICHSKAIA system with more than 23,000 ha. The wastewater is applied by a sprinkler irrigation system. The irrigation rates range from 2,000 to 4,000 m^3/ha. The type of agricultural crops are limited by sanitary rules. The crops grown are mainly used for feed purposes. Grass harvest range from 80-100 centners/ha of dry mass, tuberous crops from 500-600 centners/ha.

Animal waste runoff is used to irrigate close to 9,000 ha. Manure removal is conducted by hydraulic rinsing. The solid fraction is separated from the liquid and is then removed by mobile transportation. The liquid fraction is stored in holding ponds and is diluted with fresh water 15-20 times before being applied to the field by a hydraulic method.

The largest irrigation system is located in the KALITIANSKAIA with an area of 1,800 ha. The rate of irrigation depends upon the climatic zone and ranges from 1,000 to 4,000 m^3/ha. In connection with the increased deficit of pure water, irrigation with wastewater should obtain high development.

MONITORING REQUIREMENTS FOR LAND TREATMENT SYSTEMS

AN OVERVIEW OF MONITORING LAND TREATMENT SYSTEMS

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Monitoring of a well designed and operated land treatment system should be undertaken in a manner designed to serve as an early warning system and to help optimize the operation of the system's individual components, as well as providing performance monitoring data to fulfill regulatory requirements. Such considerations generally prove to be of much more value to the system's operators and managers, as well as to regulatory officials, than undertaking the more detailed monitoring associated with research and demonstration projects. The results of this type of monitoring can help avoid the onset of major operational problems and may lead to significant O&M savings at the same time.

INTRODUCTION

When we speak of land treatment systems, apparently certain individuals envision situations where wastewater is "indiscriminately dumped on the land," and they respond accordingly by enumerating an almost endless list of anticipated problems, including potential hazards to human health and the environment. Such "opponents" to land treatment systems often include landowners of or neighbors near the selected site and possibly more importantly, a variety of individuals trained in and experienced with conventional treatment and discharge technologies - including many of our local, state and Federal regulatory officials. Even after undertaking carefully planned and executed steps to explain the more or less natural physical, chemical and biological processes involved with land treatment systems and the detailed project design and operating/management plans incorporating the latest information available, many of these technically minded opponents to land treatment projects are not reassured that well designed and managed land treatment systems can reliably "treat" wastewater without creating "more of a nuisance than they're worth." In fact, one of the major problems with expanding the use of land treatment systems has been the institutional acceptance of such systems for water pollution control (Freshman, 1976).

One mechanism that has been widely used to gain the acceptance of specific land treatment projects is the incorporation of a detailed monitoring system design to address the potential problems that could potentially occur if conditions allowed and if built-in design safeguards failed. These very detailed research oriented monitoring systems generally give considerable attention to the internal workings of the land treatment system, while conventional treatment and discharge systems generally are only required to monitor influent and effluent quality to gain acceptance. They frequently incorporate the latest highly sophisticated sampling and analytical techniques that require the assistance of university research scientists to be properly implemented. Detailed data and information are generated by such monitoring programs, which are aimed at gaining initial acceptance of the project rather than optimizing the system's overall performance. However, in many cases, the monitoring becomes more of a research effort than one of performance monitoring and little of the data generated serves any purpose other than confirming what has been demonstrated many times before by carefully operated and managed land treatment systems.

It is time that the monitoring programs for land treatment systems be designed and implemented in a manner that serves the useful functions of providing an early warning of individual process failures and helping to optimize the overall performance of the entire system and not treat each new land treatment system as a research project. Land treatment technologies are established wastewater treatment alternatives and should be so treated. While project specific in nature, the monitoring programs for new land treatment systems should reflect this attitude.

MONITORING RATIONALES

The monitoring of land treatment systems as with the monitoring of conventional treatment and discharge systems may be undertaken for numerous reasons. In general, however, monitoring programs are normally implemented for one or more of the following objectives: (1) to meet regulatory requirements; (2) to generate detailed R&D information; (3) to serve as an early warning system and provide data to help optimize the system's operation. The level of detail required and the cost of monitoring undertaken in response to these objectives can differ greatly.

Regulatory Requirements

Some type of regulatory monitoring requirements are placed on nearly all land treatment systems, but they may differ considerably from project to project, being somewhat dependent upon the level of preapplication treatment. EPA's technical design basis for land treatment alternatives is provided in the "EPA Design Manual on Land Treatment" (EPA 625/1-77-008), which includes guidance on the level of preapplication treatment prior to storage or actual application to the land, and the criteria for protection of groundwater set forth in 40 FR 6190, February 11, 1976.

In general, the level of preapplication treatment required by EPA is not one of a uniform level for all land treatment systems (EPA, 1977), but is described as "an internal process decision made by the designer to ensure optimum performance of the land treatment processes." It ranges from screening or comminution for isolated overland flow systems with no public access to biological treatment by lagoons or inplant processes with additional BOD or SS control as needed for aesthetics plus disinfection for sprinkler application in public access areas such as parks and golf courses.

The role of adequate industrial pretreatment to protect the natural biological processes involved with land treatment systems is also stressed. As with projects applying sewage sludge to the land (EPA 430/9-77-004), it only makes sense to test wastewaters being applied to the land to determine their levels of both undesirable (pathogens, toxic organics and certain heavy metals) and desirable (nutrients, micronutrients, and biodegradable organics) pollutants in addition to the traditional pollutants (BOD, SS, pH, and coliforms) to protect the system from toxic upsets or other problems, to allow accurate accounting of nutrients applied, and to assure that discharge permit requirements are met.

State regulatory agencies, however, have frequently issued regulations more restrictive than those promulgated by Federal agencies. Data compiled by the National Commission on Water Quality and by several more recent studies (e.g., Corps of Engineers, 1975; Morris & Jewell, 1976) indicate that while preapplication treatment requirements for wastewaters prior to land application vary widely throughout the United States (depending on the site, ultimate fate of the applied waters, crops grown and other conditions -- anything from very limited to "tertiary" treatment of effluents may be required), a majority of states have required secondary treatment or secondary treatment plus disinfection prior to land application of ("disposal" in most cases) sewage effluents. Generally, the states have simply applied these "land disposal" requirements to land treatment systems which are designed to "treat" wastewaters and not merely to dispose of them on land rather than into surface waters. In some cases, state guidelines have been so restrictive that they clearly discourage if not prohibit land treatment as a practicable treatment technology mainly due to their requirements for very extensive and expensive preapplication treatment.

The differences in state preapplication treatment requirements in certain cases have also been seen in their monitoring requirements. Several states have quite flexible project monitoring requirements based upon site specific factors including application rates, water quality, crops to be grown, groundwater hydraulics and quality, soil characteristics, etc. Other states have employed uniform monitoring requirements for all land application projects or for various categories; these have generally been based upon water quality relative to any crops grown. Many of the latter situations were based upon the attitude that the wastewater was being applied or disposed of onto the land as an irrigation water rather than being "treated" by a well designed and operated land treatment system.

In general, the state monitoring requirements have ranged from analyses of BOD, SS, etc., in the irrigated water and in any effluent discharged to surface waters, as with conventional treatment and discharge systems, to specific numbers and locations of groundwater sampling wells, baseline data collected before operation of the facility, and routine analyses of effluents and underlying groundwater for such parameters as: conductivity, chlorine residual, TDS, nitrate-, nitrite-, and ammonia-nitrogen, phosphorus fecal coliform bacteria, and methylene blue active substance. In addition specific analyses of soils, crops grown, and even aerosols have been required. Under certain circumstances heavy metals, toxic organics, specific pathogen and virus analyses have been required to gain project approval. The reporting requirements for such analyses as described above have varied from one time analyses to routine daily/weekly/monthly/annual or periodic data submittals.

Since the issuance of the EPA policy on land treatment of municipal wastewater and the Land Treatment Design Manual in October 1977, there has been an apparent change in the acceptance of land treatment technologies by certain state agencies. We eventually expect this to also be reflected in the level of preapplication treatment and monitoring required by the states, including those states that have strongly discouraged or prohibited land treatment systems from being utilized. We hope that such changes will come about from an increased understanding and acceptance of land treatment as an available alternative treatment technology and not simply due to a tighter control of the Federal construction grants funding available for municipal wastewater treatment facilities.

Detailed Research Information

In many cases where extensive monitoring requirements have been placed upon land treatment systems, these requirements have been adopted from research and demonstration projects that have attempted to answer detailed questions about the physical, chemical and/or biological mechanisms at work in land treatment projects. This may be due in part to the joint funding of research efforts as a part of full scale demonstrations of land treatment (e.g., Muskegon, MI, and Phoenix, AZ). Experimental efforts have been underway for some time to address such detailed questions as the longevity of pathogens and viruses in effluents sprayed on the soil and crops, the impact of infiltrated and percolated effluents upon groundwater quality, the effect of a system's age on hydraulic efficiency and renovation efficiency, the effect of natural soil chelation processes on metal availability, movement and availability of organic and inorganic P and N, the impact of application rates upon nitrification/denitrification processes, etc. The types of detailed monitoring required to provide the data necessary to address such research areas are generally far more detailed than what should be expected from non-research oriented but well designed and operated land treatment systems.

The Musekgon Land Treatment Project (slow rate system using center pivot irrigation) is an example of a jointly funded research and demonstration project. The operation of the Muskegon system has included an extensive series of research programs (involving over \$2.5 million to date) to determine the efficiency of the system, to ensure that the quality of the underdrain water meets discharge standards, to investigate the detailed physical, chemical and biological mechanisms at work renovating the wastewater, to study impacts on groundwaters underlying the system, and to follow the response of surface waters into which the renovated water recovered from beneath the system is discharged (Bastian, 1973). The research monitoring program at Muskegon was designed to evaluate the influent, biological preapplication treatment, storage, irrigated water, lagoon seepage, groundwater quality and movement, surface water, soil and crop characteristics in detail. Samples were 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 were analyzed for at least one of 25 wastewater constituents. In addition, groundwater was sampled monthly to twice yearly from over 300 wells (at both onsite and offsite locations) for analysis (Demirjian, et. al., 1978; Walker, 1976; Culp and Hinricks, 1976; EPA 625/1-77-008).

This major research funded monitoring program required the services of nine full-time laboratory personnel at the facility plus several outside research groups. The results to date have shown that no significant negative effects on the groundwater under the project site have occurred. At the same time, improvements are becoming evident in the area lakes that had received primary and secondary effluent discharges before the land treatment project was implemented. Finally, no significant negative soil impacts have occurred and no limitations to the sale of the corn crop have occurred as a result of detailed crop quality monitoring. This is not meant to imply that the project has not had problems, but only that the project has performed well within its design expectations.

The Phoenix "Flushing Meadows" rapid infiltration research and demonstration project was constructed and has been operated in a highly successful manner for about ten years as an experimental pilot project to demonstrate the feasibility of renovating secondary effluent for unrestricted irrigation and recreational uses. A scaled-up project was constructed in 1974 based upon the engineering criteria developed at Flushing Meadows. In addition to groundwater level and quality, once fully operational this project will also follow the direction of groundwater movement by monitoring TDS concentrations in observation wells outside the infiltration basins. Continuous 24-hour samples will be taken of effluent entering the infiltration basins. Additional research at the Flushing Meadows and scaled-up facilities has emphasized how air pressure build-up in the soil beneath the large infiltration basins affects infiltration rates, effects of high algae loadings on surface clogging, and the fate of viruses and nitrogen in the wastewater as they enter the soil. In all, over 30 parameters have been closely monitored at the Phoenix research and demonstration facilities. The renovated water from the projects has been certified by the Arizona Health Department for unrestricted irrigation and recreational uses.

The detailed research oriented monitoring being undertaken at joint research and demonstration projects such as the Muskegon and Phoenix projects should not be required at all land treatment systems for a number of reasons including cost, accuracy and value of detailed data to the systems operation, and the lack of agreement within the scientific community as to the interpretation of the research monitoring data.

Interpretations of experimental results from such detailed studies should, however, help provide the basis for determining new system design factors, including built-in safeguards to avoid both short-term and long-term operating problems. Summaries of research findings are published quite frequently in conference proceedings (such as this one), technical journals and reports (Appendix A). This detailed information should serve as the basis for updating design manuals, guidance and regulations to allow appropriate system design rather than intensive system monitoring to address the most recent findings of detailed research and demonstration studies.

Early Warning & System Optimization

In general, the monitoring of well designed and managed land treatment systems can be undertaken in a manner quite similar to the monitoring of conventional treatment systems, with early warning of individual process failures and optimization of system performance in mind. This type of system emphasizes the monitoring necessary to protect the system's vital functions against unforeseen or unpreventable failures as well as fine-tuning of individual processes. It isd a basic necessity for a well managed land treatment system's successful operation. The equipment that is utilized in preapplication treatment, pumping stations, transmission systems, and irrigation systems are especially vulnerable to mechanical failure and vandalism in spite of routine maintenance programs. In addition, monitoring for uncontrollable slugs of toxics that could enter a land treatment system can prove to be invaluable by allowing the operation of the system to adjust and thereby prevent major process failures that might lead to poor water renovation or long-term harmful effects to soils or crops.

By monitoring for highly mobile ions such as chlorides and nitrates, the actual movement of the wastewater can be detected. If these indicators are detected at elevated levels in areas of groundwater where levels above background are not expected, further detailed monitoring to determine the actual source of the indicators should be initiated and, where necessary, changes in the system design and/or operation can be undertaken to mitigate or prevent future problems.attributable to the system. Fairly inexpensive though efficient monitoring of indicators has proven to be an effective early warning system to handle potential groundwater impacts problems.

In addition, monitoring of the water quality parameters identified in the system's operating permit at the beginning or end of each major step of the land treatment system (e.g., influent, preapplication treatment, storage, point of application to the land, and point of recovery, discharge or entry into groundwater) may allow for modifications, perhaps only seasonal, to the system's operation that can lead to sizable O&M savings. For example, storage lagoons can also act as effective stabilization and treatment mechanisms that could allow for at least seasonal cutbacks in preapplication treatment by mechanical aerators, etc., and save both significant energy and maintenance expense. Higher-than-design loadings may be possible during high flow periods as long as final effluent quality meets permit requirements, thereby preventing bypass problems and expenses. Chlorine residuals for disinfection purposes may be avoidable where natural die-off is demonstrated to meet acceptable levels. Monitoring of crop condition and tissue analyses during the growing season will allow adjustments to be made in fertilization or pest control programs that prevent possible crop failures and resulting losses in levels of wastewater treatment or revenues from crop sales.

This type of monitoring involves the observation of significant changes that result within the land treatment system as well as at the point of discharge. Such monitoring data can serve to confirm predictions and to determine where corrective action may be necessary to protect the environment, maintain the renovative capacity, or improve the operation efficiency of the land treatment system.

MONITORING PROGRAM DEVELOPMENT

The development of a practical monitoring program for a specific land treatment system should initially focus upon the objectives you wish to achieve from monitoring the system. Once these objectives are established, be they regulatory, research, operations or otherwise oriented, you can proceed to develop a monitoring plan to meet your needs and the project's specific site conditions.

Since land treatment systems are very site specific in nature and design (e.g., high rate vs low rate; public access vs no public access; varying in climatic, soils, geology, hydrology, crop, influent and required effluent quality characteristics), this variability will also be evident in a particular land treatment system's monitoring needs and program design. There are very few, if any, general rules to go by that would dictate a particular number of groundwater monitoring wells, soil samples or even sampling frequency for all land treatment systems due to this project specific nature.

Because land treatment systems involve wastewater interactions directly with soil and often vegetation, and indirectly with groundwaters and surface waters, monitoring of the system's environmental impacts is often concentrated on these interactions. Additional monitoring may also involve the project's effects upon marketed crops, wildlife, odor and aerosol production. Also, due to these environmental interactions, as well as operational and regulatory considerations, it is very important to monitor the water quality characteristics of the system's wastewater influent and ultimate discharge to surface or groundwater.

In most situations regulations will require at a minimum that land treatment projects monitor the water quality of the influent and renovated water for traditional pollutants. In addition, most systems will be required to monitor the quality of the applied wastewater and the groundwater underlying the project site. Many systems will also be required to analyze soils and vegetation on a periodic basis. Systems with complex industrial inputs in their wastewater should also place a major emphasis on monitoring these industrial waste sources and their pretreatment activities.

In general, it is necessary to monitor new projects more intensively during their initial period of operation to be assured that the system's processes are effectively operating and the design criteria are met. With experience and a data base demonstrating successful performance, even the regulatory officials will become more comfortable with these relatively "natural" systems; their sampling frequency and analyses requirements may even be relaxed. Eventually the monitoring program for a new land treatment project may even be allowed to function as a tool for the system's operators and managers rather than a regulatory crutch.

EQUIPMENT AND DETAILED PROGRAM DESIGN

Once the decisions have been made concerning what is to be monitored to meet the objectives of a planned monitoring program, a logical question that comes to mind is "What procedures should be used?" The Land Treatment Design Manual (EPA 625/1-77-008) addresses many helpful procedures in the detailed appendices. While detailed "standard methods" exist for sampling and analyzing many water, soil and biological constituents, a very useful publication titled "Sampling and Analysis of Soils, Plants, Wastewaters and Sludge; Suggested Standardization and Methodology" (Ellis, et. al., 1975) is available that addresses sampling and analyses procedures as they apply to land application systems. The use of standard samples, sampling, sample handling and preservation and analytical procedures are all covered in a readily understandable format. References providing sources of additional information are also provided. Other sources of detailed information include professional societies publication (such as those of the Soil Science Society of America, the American Society of Agronomy, the American Society of Agricultural Engineers, the Society of Analytical Chemists, and others).

Considerable information on adopting standardized methods to handle specific project requirements is available in the literature. True insights into the problems encountered when carrying out monitoring programs, however, can best be obtained from operating project personnel responsible for their system's monitoring program. Further information on techniques for monitoring different types of land treatment systems will be presented in the papers to follow in this session.

CONCLUSIONS

Monitoring of land treatment systems may be undertaken for a variety of reasons. These may include the collection of data for submittal in reports required by regulatory agencies and for investigations of detailed process mechanisms as a part of research efforts. Monitoring can also serve as an early warning of process or mechanical failures and to provide information that allows operators and managers to make operating adjustments in an effort to optimize their system's performance.

In the past land treatment systems have often been required to develop and implement very detailed monitoring programs based upon the type of monitoring performed at research and demonstration projects to obtain project approval by regulatory officials. A clear distinction, however, should be made between the monitoring needs of individual operating land treatment systems and the detailed monitoring of research and demonstration projects. Monitoring of operating systems should be limited when possible to that required to comply with realistic regulatory requirements and to serve the system's operators and managers as a tool for early warning of unforeseeable process failures and for system optimization purposes.

Land treatment technologies have been repeatedly demonstrated to provide reliable engineering processes for wastewater management. The monitoring of these systems should, therefore, be undertaken with the same attitude in mind as the monitoring of other accepted wastewater treatment systems. Detailed research oriented monitoring should be limited to research and demonstration efforts or special situations where it is of real value while practical monitoriing programs designed specifically for local site conditions and with specific objectives in mind should be the normal course for well designed and operated land treatment systems.

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APPENDIX A

Examples of Sources Providing Detailed Research Data Summaries

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MONITORING REQUIREMENTS FOR LAND TREATMENT SYSTEMS

DESIGN OF WATER QUALITY MONITORING SYSTEMS FOR LAND TREATMENT OF WASTEWATER

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ABSTRACT

This paper introduces the subject of monitoring of water quality resulting from land treatment of wastewater. Omitted are those aspects which are common to both land treatment and other forms of treatment, such as aerosols, odors and discharges to surface waters. The paper concentrates upon soils, plants, soil water and ground water. Typical models for movement of ground water are presented. Comparison of the steadiness of the land treatment system with the steadiness of the conventional system is made. Field methods for measuring soils permeability and transmissivity are discussed, as are the conventional model of agricultural soil systems and conventional sampling and chemical analyses of soils. Samplings and analyses of plant and water systems are also discussed. Finally, a bibliography for further study is presented.

INTRODUCTION

General Approach

This paper is not concerned with those aspects of monitoring which would be common to both conventional and land treatment of wastewater. For example, the monitoring of aerosols is not discussed. One would use the same techniques for the monitoring of aerosols from a land treatment system as for a conventional treatment system. Aerosols produced by trickling filters or aeration tanks travel in the same way through the air as aerosols from a spray irrigation system, and are affected in the same way by atmospheric conditions. The same condition would apply to the monitoring of odors or the monitoring of discharges to surface waters, something which is often the final step in a land treatment system.

Ground and Soil Water Impacts

Impacts of land treatment of waste wastewater upon ground and soil water can be and are monitored by a variety of methods. Methods of sampling and analysis are examined here. Methods to control the direction of movement of ground water are also discussed, such controls being an effective way to provide assurance that the ground water under neighboring properties will not be affected.

Soil Impacts

Over a long period of time, the soil of a land treatment system will be impacted by the process. Most commonly, these impacts are beneficial, such as the increased humus content of the upper soil horizon. There are concerns, such as possible accumulation of heavy metals in the soils, which are sometimes expressed as potential hazards. This subject will also be discussed.

Plant Impacts

Related to the possible accumulation of heavy metals in soils receiving applications of wastewater is the concern about possible adverse effects upon plants grown on these soils. The viewpoint that such concerns are only hypothetical and without any basis in objective observations is presented.

SOIL AND GROUND WATER MODELS

Soil Water Mechanics

Definition. The term "soil water" is used for the water present in the nonsaturated layer of soil above the elevation in the soil which is saturated with water, the water in the saturated zone being called "ground water". In the nonsaturated zone, the soil water is present in the form of films of water which cling to individual soil particles. The void space between soil particles not filled with water is then available for atmospheric gases, including oxygen. This oxygen is essential for certain natural soil bacteria which function to solubilize plant nutrients which are present in the soil.

Surface Tension. The property of water which dominates the mechanics of water movement in the zone of soil water is the property of surface tension. This is the same property that dominates the phenomenon of capillarity, as in the action of a wick in drawing fluid up against gravity. Surface tension is so dominant with finely grained soils that the effect of gravity in the zone of soil water can be largely ignored. On the other hand, as soils with progressively coarser structures are examined, one observes a systematic increase in the relative importance of the gravitational forces with respect to the forces of surface tension.

The effect of surface tension is responsible for the fact that not all of the water in a typical soil can be drained out of it by gravitational forces. The finer the soil texture, the larger the fraction of the soil water which cannot be so drained. Other forces, such as those developed by osmosis or by evaporation must be relied upon to remove water from such soils.

Infiltration. When sufficient water is available on the surface, infiltration into the soil occurs. The rate of infiltration is also a function of soil texture. Fine-grained soils have small rates of infiltration, and coarse ones have large rates. When the rate of infiltration is in excess of the quantity required to maintain the minimum of soil moisture, then a general downward movement of the water in the soil occurs, extending from the surface to the ground water. This is the mechanism whereby the ground water is most commonly "recharged".

Further Study. Those interested in the mechanics of water movement in the zone of soil water should do further reading in the field. Book (1) in the Bibliography at the end of this paper is suggested as a starting point. Much work has been done in studying soil water mechanics, particularly in the field of agricultural engineering.

Ground Water Mechanics

Definition. Ground water exists in the saturated portions of earth which lie beneath the zones of soil water as discussed above. In this zone, all of the interstices between the particles of soil are filled with water, generally with very little dissolved gas in it, though often with considerable quantities of dissolved solids.

Gravitational and Viscous Forces.

The dominant properties of water which govern the movement of it in the ground water phase are those of fluid weight and fluid viscosity. The gravitational effect which results in fluid weight provides the motivating force which produces fluid motion. The viscosity of the fluid is the property which produces forces which resist motion. The balance of these forces is achieved when the motion of the fluid has reached an equilibrium, steady state.

Viscosity of water is a function of the temperature, and to a minor extent a function of the materials which may be admixed with the water. In the practical case, there are no admixtures in the ground water which significantly affect the viscosity, and the temperature of the water is nearly a constant, so the ordinary assumption is made that the viscosity is a constant. This is the reason why one fails to see a viscosity term in the mathematical models most often used to describe the movement of ground water.

Gradients. In accordance with the second law of thermodynamics, ground water moves in the direction of the hydraulic gradient, this being the direction of maximum negative gradient of the piezometric potential. At the surface of the zone of saturation the interface between the zone of soil water and the zone of ground water, commonly called the "water table" - the pressure is considered to be atmospheric or zero. At this location, the piezometric potential is equal to the elevation. Rather commonly, one measures the locations of the "ground water table" and from this deduces what the pattern of piezometric potential must be. Less commonly, one also measures the piezometric potential at other locations below the "ground water table", and uses these measurements also in constructing a map of piezometric potential. The common way of showing such measurements is in terms of maps with contours of equal piezometric potential. The maximum piezometric gradients are then found to be perpendicular to the contours of equal piezometric potential.

Flow Nets. The network which can be made up of lines of equal piezometric potential and the lines of maximum piezometric potential gradient perpendicular to those lines is called a flow net. Often the flow net is constructed graphically, using the principles that the "squares" of the net should be as nearly square as possible, and that all intersections should be at right angles. As the net becomes more and more finely subdivided, these requirements should be increasingly more accurately met.

The flow net is a useful tool to calculate the rate of flow of water through saturated permeable materials. The area diagrammed is bounded by the "ground water table" at the top and by an aquiclude at the base. This aquiclude is a relatively impervious layer of earth which is continuous and functions to prevent the flow of water through it. A partially permeable aquiclude is sometimes called an "aquitard", a layer with less permeability which retards the flow through it. More accurately, the aquitard simply requires a steeper piezometric gradient to achieve a given rate of flow. If the gradient is not possible, then the flow rate through the aquitard is correspondingly small.

Ground Water Models. The actual mass of soil which is saturated with water is by its nature ordinarily very complex. The characteristics are distributed in a nonuniform manner throughout the mass, except in unusual cases. Such a situation is typical of any problem in hydraulics of natural earth formations, whether the problem be one of surface or of ground water hydrology. In such instances, one employs the technique of substituting for the real condition a "model" which is less complex and more amenable to mathematical description. As the complexity of the model approaches the complexity of the real situation, the correspondence of model behavior with actual behavior comes closer to unity. However, it is often found that a relatively simple model of a naturally complex system will give such a high degree of correspondence as to be very useful in analysis. This fact results from the dominance of certain aspects of the natural situation, so that by taking these dominant aspects into account in the model and ignoring the others one can achieve a model which closely approximates in its behavior under a given set of inputs the actual behavior of the natural system under the same set of inputs.

The validity of the model is demonstrated by a process called "calibration", in which the same inputs are used in both model and prototype (the latter being the name given to the real situation in nature) and the "calibration" is considered completed when the outputs of model and prototype agree within acceptable limits.

Specialists who work with models of ground water systems, like specialists in any field of work, acquire a skill in assessing very rapidly what are the dominant aspects in any given situation. They also acquire a skill in making a selection of the appropriate model, such that they are very often correct in the very first model which they propose to use. The techniques used by these specialists involve both physical and mathematical models, the latter often being of the digital type for use with an electronic computer. Examples of physical models are actual models using water and earth materials in a laboratory and the use of electrical analogy models in which voltage corresponds to piezometric potential, and current flow to flow of water. The explanation of how models are constructed and operated is a subject far beyond the scope of this paper. Suffice it to say that such techniques and specialization exist, and it would behoove anyone who contemplates setting up a monitoring system to secure the services of a skilled hydrologist who can produce a model of any desired degree of sophistication.

Controlling Ground Water Movement. In some instances, it is desirable or necessary to control the direction of movement of the ground water. Control methods can best be devised after the model of the particular ground water situation has been developed. The principle of control is always the same: produce gradients of piezometric potential of the desired direction and magnitude. Gradients are produced by adding or withdrawing water from a system. Additions can be by surface application, or by injection wells. Withdrawals can be by evapotranspiration, or by drainage. The latter can be accomplished by either horizontal or vertical drains, the latter commonly called wells. The Muskegon County Wastewater Management System No. 1 used both wells and horizontal drainage pipes and ditches for control of the piezometric gradients, and hence the control of the ground water movement. The aquifer was a dune sand 20 feet to 160 feet in thickness underlain by a clay aquiclude.

Controls of ground water movement are ordinarily designed to permit variation of inputs in response to observed outputs. If the movement is either too fast or too slow, one can change the inputs to achieve the velocity desired. Likewise, if the direction is not exactly what one desires, one relocates the inputs to alter the direction in the desired manner.

Chemical Reactions

Aerobic Conditions. The conditions in the zone of soil water are generally aerobic. The chemical reactions which occur in this zone are not the same as those which occur in the anaerobic conditions which generally exist in the ground water below. Thus, nitrogen in the soil water is generally in the nitrate form, such ammonia as may be present being converted to this nitrate form by the action of aerobic bacteria which are present in the zone of soil water. If the nitrate is not taken up by plants and moves on down into the ground water, there is the possibility of a different action taking place as is discussed below.

The soil chemistry in the aerobic zone has been the subject of much s study and research.

Anaerobic Conditions. In the ground water there is usually a deficiency of oxygen, and anaerobic conditions prevail. Here a different set of chemical reactions take place. For example, if nitrate nitrogen enters the ground water and sufficient carbon source is also present, denitrifying bacteria which operate under anaerobic conditions can separate the nitrogen from the oxygen in the nitrate, releasing nitrogen gas which eventually escapes to the atmosphere.

Not as much is known about chemical reactions in the anaerobic ground water zone, as none of the agricultural processes take place in this zone. Only recently is knowledge of this zone being expanded, with the research at the Muskegon County project being one source of information. Here, the soils under the storage lagoons and under the dikes surrounding these lagoons permit steady percolation of water leaking out of the lagoons and into the interception ditch which is provided for that purpose. During the first 5 years of operation of these lagoons, this soil mass - which is roughly 60 feet in average thickness, 500 feet in width, and 40,000 feet long - has been filtering an average of 16 million gallons of water per day. The filtered product meets the highest standards for any effluent to be discharged into Michigan streams, and is so discharged without further treatment. For example, the toal phosphorus is consistently changed from about 2 mg/l in the storage lagoons to about 0.1 mg/l in the water in the ditch. Furthermore, much of the total P in the ditch water comes from the biological community which lives in the ditch. There is reason to believe that the total P content of the percolating water before it enters the ditch is about 0.2 mg/l. This is an example of the type of chemical change which is taking place in the anaerobic conditions which exist in the saturated sands through which this leakage is taking place.

Long before the mechanics of the soil chemistry is understood, it is possible that sufficient information would be obtained from the actual operation of the Muskegon system to permit other systems to be designed to take advantage of what appears to be a very reliable yet inexpensive method for removing phosphorus from the percolating water (probably the precipitation by iron in the soil).

COMPARISON WITH CONVENTIONAL MONITORING

Steadiness

It is instructive to compare the steadiness of the monitoring of land treatment systems with the steadiness of monitoring of the alternative conventional systems. Conventional systems do not have the very large storage facilities that commonly comprise a part of a land treatment system. Hence, they are subject to relatively rapid changes in conditions as the quality of the incoming sewage may change abruptly. If one is then to monitor properly a conventional treatment system, the number of samplings must be very large. The quality of the incoming sewage may change from hour to hour. Such frequent sampling is usually considered too expensive and compromises are made which result in missing the sampling of sudden changes in quality resulting from an industrial spill, for example. The result appear as anomolies in the final data, where one could find more nitrogen in the effluent than in the influent, for example.

The inherent unsteadiness of a conventional treatment system combined with the high cost of sampling to measure the actual variation in quality with time result in monitoring systems which are not as comprehensive as one would desire.

By contrast, the land treatment system with sufficient storage is a very steady system. The variation in results from hour to hour, day to day, or even week to week is relatively small. Although the quality of the imput may vary just as widely as it would in a conventional system, the fact that these variations must blend with a much large mass of water which has accumulated over a long period of time necessarily results in very small variation in the quality of the water in the storage lagoon, for example.

This steadiness in land treatment systems makes possible a careful monitoring with many fewer samples to be analyzed than would be required to obtain the same degree of knowledge about a conventional system.

Reliability

The land treatment system is inherently a very reliable system. It is difficult to discharge untreated water from the system; in comparison, the conventional treatment system does not require much in the way of an upset or a temporary overload to result in a discharge of polluted water directly into a waterway.

Again, the reason for the difference lies in the use of large volumes of storage in the land treatment system. If sudden changes in quantity of quality of sewage occur, they are absorbed with little or no difficulty by the large masses of water already in the system.

The writer has often proposed that a comparison be made of the relative reliabilities of treatment systems, with the Muskegon County System, for example, being compared with any other system in the State of Michigan, particularly with conventional systems such as exist at Detroit, Grand Rapids, Lansing and elsewhere. It is believed that such comparisons would reveal a very much higher degree of reliability in the land treatment systems.

Now this reliability is very important to the achieving of the clean streams for which the entire water pollution control program is striving. If a stream receives shock loads of occassionally untreated sewage, such shock loads would certainly nullify the effects of good treatment performance between shock loads. The possibility of shock loads would make the stream unreliable as a source of water for reuse and as a stream which could be stocked with fish for sport fishing.

(Because it is difficult if not impossible for conventional systems to achieve reliability, this problem is largely ignored by the environmental engineering community. It is one of those vexing problems - like the handling of polluted storm water run-off - which is being "postponed" until some future date. One would do well to ponder if there is any conceivable answer to either of these problems save in the use of massive amounts of storage.)

Again, because land treatment systems are inherently reliable, it is possible to achieve sufficient monitoring of results with very many fewer samplings than would be true for a conventional system. This fact is not recognized by most authorities in the field, however, possibly because past experience with conventional plants has shown them to have insufficient sampling and analysis to fully monitor what has happened.

PERMEABILITY AND TRANSMISSIVITY MEASUREMENTS

Mechanical Properties of Aquifers

Systematic mechanical analysis of soils and aquifers begins with assembling of previously available information about the region. Aerial photographical maps, logs of borings of exploratory drillings and of wells, and geological reports are all sources of such information. On the basis of such information, one constructs a preliminary model of the land and soil mass to be involved in a land treatment system and develops a program of further exploration. This can take the form of drilling and sampling of the soils in the conventional manner, supplemented in some instances by seismic explorations to map interfaces between formations of substantially different densities.

Laboratory tests of samples would include size distribution analyses, with particular emphasis upon the distribution of the finer particles. It is usually true that the finest 5% to 10% of the soils particles determine the permeability of the mass. The number of such samples would depend upon the amount of variation of soil types in the region involved.

Hydraulic Properties of Aquifers

The best way to measure the hydraulic properties of an aquifer is through the making of actual field tests. This can best be done by means of well pumping tests. In each test, a pumping well is located in a strategic location and a number of observation wells are constructed around it. The water level in each well before pumping is measured. Then the pumping well is operated for a long period of time - for several days up to 2 weeks - at a steady rate without interruption. The rate of pumping should be the maximum that can be sustained. The rate of pumping is measured and recorded at regular intervals. At the same time, regular and systematic measurements are made of the water levels in the observation wells. The initial measurements are made relatively soon after the well pumping begins. Successively longer intervals elapse between times of water level measurement as the test goes on.

The resulting data permit one to calculate both the storage coefficient and the transmissivity of the formation in the vicinity of the well. By making a number of such tests, one can determinethe variation of transmissivity of the quifer with location throughout the region of the proposed land treatment system. The cost of one well pumping test is likely to be in the range of $15,000 to $50,000, depending upon the depth and capacity of the aqifer. If four such tests would be used on a large treatment site as for Muskegon, the cost would be as much as $200,000. Spread over a large site of 10,000 acres, this would amount to $20 per acre, which is not a large fraction of the total project cost. Yet these tests provide essential information by which the design of the ground water monitoring system is controlled.

SOIL SYSTEM ANALYSIS

Agricultural Soil Horizons

Bibliographies (1) and (2) provide rudimentary information about the standard way of classifying the soil horizons from the point of view of agriculture. This system should be recognized by the persons making analyses of soil systems to be used for land treatment, as both the agronomical and soil chemistry considerations will be much aided by the agricultural perspective of a soil system.

Suffice it to say here that the agriculturalist classifies the soil into the A, B, and C horizons, with the progression being downward from the surface of the ground. Within the A horizon, one finds at the top the plow layer, underlain by successively less and less fertile layers in the typical situation. Much of the important action in a land treatment system takes place in the plow layer, so most of the attention in monitoring is paid to this layer.

Sampling and Analyses

Typical sampling is done by hand using special stainless steel push-probes which are used to take samples typicall 1/2" in diameter and seven inches long. Ordinarily a number of these samples is taken to comprise a composite sample of the soil in a given general location, spread out over about one acre. A number of such composite samples - about one per 4 acres - would be used to characterize a 40-acre field.

The samples are classified visually by the person taking the sample, and sealed in tight plastic containers to preserve the moisture content. They would be carefully tagged and labeled and an accurate record would be kept.

Soil Chemistry

Reference (2) in two volumes is a source of information which concentrates on the organic aspects of soil chemistry. This field of work is so different from the chemistry that is ordinarily applied to environmental engineering work as to call for entirely different procedures and training. Much waste has occurred and much misinformation has been generated by laboratories inexperienced in soil chemistry which have attempted to perform assignments in this field. Responsible professional engineers have a duty to insure that any analytical laboratory which they use for the analysis of soil samples is fully qualified in that field.

Likewise, the interpretation of the laboratory results must be performed by persons experienced in soil chemistry. Such persons can quickly spot laboratory problems and errors and if brought into the picture at an early date can greatly multiply the value of the laboratory work which is done.

Typical Results

Samples of soil involved in land treatment system are often analyzed for changes in organic content, changes in nutrient content and for accumulations of metals such as zinc lead and cadmium. Such changes occur most prominently in sludge apllications to land, where the amount of solids applied to the soil is perhaps 10 to 100 times as much as would be applied in a year's application of wastewater. Results from testing soils to which large amounts of sludge have been applied have shown no problems with the crops subsequently grown on these soils, the many papers which have raised the spectre of such problems notwithstanding. The serious investigator will do well to probe deeply the claims of the persons who warn of problems of heavy metal accumulations or uptake by plants and he will find that such problems are entirely conjectural. In no actual field study has such a problem been observed. This warning is necessary in view of the popular misconception that problems of heavy metal accumulations actually do exist in real situations.

Because the testing and analyses of soils subjected to heavy dosages of sewage sludge have shown these soils to have benefited from such applications and not been degraded, one is inclined to discount even further the possibility that soils would be adversely affected by the application of the very much more dilute sewage effluent to them. For this reason, it is prudent to do the soils sampling and analytical work very carefully and very well, even for a few samples, rather than to attempt to do a large number in a careless and perhaps misleading manner. To discover an adverse impact on soils resulting from the application of sewage to them would be startling, in view of the total lack of such evidence at present.

PLANT SYSTEMS ANALYSES

General Observations

Sampling of plants to determine deficiencies in nutrients and micro nutrients is a well practiced art among sophisticated agronomists. Refined laboratory techniques have been developed to aid in this work, and the securing of such a qualified laboratory will greatly facilitate the obtaining of valid results.

This is again a field of considerable specialization and one is advised to secure the services of qualified specialists with considerable experience.

Typical Results

Again one would be surprised to find that the irrigation of crops with sewage effluent would in any way create an adverse impact on such crops, unless the amount of water applied is excessive, such as to drown the essential soil bacteria. Such an adverse impact could of course be produced by plain water, so that the use of effluent would be of no consequence. Aside from such foolish approaches, there would probably be no way that one could produce an adverse impact, assuming the sewage to have been given biological treatment prior to irrigation. (This biological treatment gives assurance of the absence of biological poisons in the sewage.)

For this reason, one would ordinarily conduct a rather economical program of monitoring of plants produced by irrigation with wastewater. There is no known way in which adverse impacts could be produced, saved by deliberate sabotage or outright foolishness.

WATER SYSTEMS ANALYSIS

Soil Water

Soil water is sampled by means of lysimeters which intercept soil water as it percolates downward toward the ground water table. These are ordinarily installed in pits in the ground and utilize horizontal or slightly tipped trays to intercept the flow and to deliver it to sampling containers. Samples are removed at intervals and taken to laboratories for analysis.

Tests of soil water are made solely for research purposes. One seeks to determine the rate at which changes in the chemistry of the percolating water is taking place as a function of the thickness of the soil through which it has passed. By taking samples first at one and then at another depth, one can answer questions.

From the point of view of an operating land treatment system, it is important to know the quality of the final product and this if found in the ground water under the irrigation site.

Ground Water

Samples of ground water may be withdrawn with the aid of wells or by removing water from the horizontal drains of the regular drainage system. In the Muskegon County project, both techniques are used.

Samples of ground water are generally found to be devoid of suspended solids and bacteria. Such pollutants as may be present are all in dissolved form. With ordinary loading rates on good agricultural soils, testing for virus usually shows it to be absent in the ground water, even though it may be present in the applied water. Only in instances in which the soil is relatively coarse - such as gravel or course sand - would one expect a few virus to reach the ground water. Such soils may not be suitable for the land treatment of wastewater.

When the ground water is allowed to empty into open ditches as it is in the Muskegon County project, it picks up from the biological community of such ditches a new complement of suspended solids and bacteria. Thus, the final effluent as discharged to surface streams does not have the same high quality that one would find in the ground water itself. For example, the ground water at Muskegon County may be compared with the ditch water as follows:

Parameter	Ground Water	Ditch Water
Suspended solids	0	4
Total P	0.02	0.10
Bacteria	0	100 MPN

These results are merely illustrative of the type of results to be expected. The major differences are in those materials which the soil is effective in removing. Soluble materials such as nitrates and chlorides remain unaffected.

Frequency of Sampling

Because of the relative steadiness of a land treatment system, there will be very little variation in quality of the ground water from month to month. Thus the frequency of sampling can be low. Just how low one could determine from actual experience with a given site. One should be prepared to find that perhaps 4 samples per year at a given location would suffice.

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MONITORING REQUIREMENTS FOR LAND TREATMENT SYSTEMS

DESIGN OF SOIL-PLANT MONITORING PROCEDURES FOR LAND TREATMENT SYSTEMS

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The philosophy of a monitoring program of soil and plants in a land treatment system is discussed, and recommendations of suggested analyses and frequency of sampling are given. Since a monitoring program is site specific, no firm recommendations are possible. Interpretation of the data is difficult due to sample variability and regional climate and soil differences. For constituents which are tightly sorbed on soil particles, long-term trends in soil profile distribution and plant uptake will be needed to establish potential impact. Yearly sampling of the surface soil is recommended mainly to evaluate soil fertility and yearly harvested plant material to evaluate crop quality and N and P removal efficiencies. We feel the difficulties in properly monitoring the NO_3-N concentration of the percolate below the root zone, which often is essentially at the ground water surface, are such that it is not justified in a soil monitoring program. Rather NO_3-N must be included in a ground water or underdrain monitoring program.

INTRODUCTION

A monitoring program for a land treatment system should be designed to provide continuing evaluation of system effectiveness and provide early warning of soil, water or plant growth problems not anticipated in the system design. The amount of monitoring required should be the minimum necessary to attain these objectives since monitoring is expensive. In this regard, it is important to differentiate between a monitoring program designed strictly for continued evaluation of system performance and one in which research and development is also a component of the project. The latter would require more extensive data collection than we recommend here.

Monitoring programs must of necessity be site-specific. Among the factors to be considered are: (a) wastewater characteristics and variability of these characteristics; (b) application method [slow rate (SR), rapid infiltration (RI) or overland flow (OF)]; (c) crops grown and ultimate use of the crops; (d) ground water or surface water regime and potential for contamination of the receiving waters; (e) climate; (f) size of the system; and (g) availability of transferable performance data.

In this paper, we have limited our discussion to nutrients, metals and persistent organics in the soil-water system. Ground water and pathogen monitoring is discussed elsewhere in the symposium. Further, we make no attempt to review in detail the numerous biochemical, chemical and physical transformations of the constituents considered here. These are discussed elsewhere in this symposium and in several other reviews and symposia proceedings.

CRITICAL CONSTITUENTS

Nitrogen

Most SR and OF systems are designed to take advantage of the nutrients of the wastewater to partially or totally supply nutrients needed for food or fiber crops. This enables an economic return from the system to offset some of the costs and permits removal of these nutrients in the crop. In contrast, agronomic crops are seldom a component of RI systems.

Because of the requirement that the percolate below the root zone of land treatment systems contain less than 10 mg/l of NO_3-N (13), N removal is of critical importance in most land treatment systems. Luxury consumption of N, leading to excessively high NO_3-N in forage or silage crops and potential animal health problems, must also be considered (31,42). Crop species selection, particularly for SR systems, usually is made based on their relatively high N requirement. The most commonly used crops are corn, corn interseeded with an annual grass such as rye, or forages (11,13,20,28,34). Forest ecosystems also have been utilized, but often are considered less desirable due to their lower capacity to assimilate N (17,19). Even though the total wastewater N applied over a growing season may meet or exceed annual crop requirements, in some cases a N deficiency may occur. This is because the wastewater may have too low a concentration of available N to supply the crop needs during the period of rapid growth and N uptake. Supplementary fertilizer N may be required as was demonstrated in the SR Muskegon project (14).

Denitrification may also be an important mechanism of N removal in land treatment systems (16). Lance et al. (26) have demonstrated the potential for managed denitrification to remove up to 70% of the effluent N in RI systems. Nitrification-denitrification at the plant-soil interface is apparently responsible for much of the N removal in well-managed OF systems (13). The potential for denitrification in SR systems is considered much less than in RI or OF systems since aerobic conditions must be maintained for optimum plant growth, and relatively coarse-textured soils are used to obtain satisfactory infiltration rates. The current design manual (13) uses 20% denitrification of effluent in the calculation of N loading rates with SR systems. This is an empirical value based on N balances obtained with fertilizer N studies, and true denitrification in SR systems could be significantly greater or less than 20%. In fact, the seasonal distribution and total amount of denitrification in SR systems is extremely difficult to predict, and virtually impossible to monitor.

Phosphorus

The amount of P added in land treatment systems will usually exceed crop needs. Thus, soil attenuation becomes the important P removal mechanism for SR and RI systems over the site lifetime, and monitoring programs are needed only to evaluate the depth of P leaching, especially on coarse-textured soils where P retention would be relatively low. Phosphorus toxicity to crops is extremely unlikely, but imbalances, particularly of micronutrients could possibly be induced by accumulation of excess available P (9,33). Overland flow systems normally are less efficient in removing P from wastewater; removals of 35 to 60% are commonly reported (13).

Potassium, Calcium and Magnesium

These nutrients dominate the cation exchange sites of most soils (except sodic soils). While of no concern from a pollution standpoint, they can be of importance in system performance if they are not present in sufficient amounts for normal plant growth. While municipal wastewaters likely contain sufficient Ca and Mg that these elements will always be present in sufficient amounts, K deficiency in heavily cropped municipal wastewater treatment sites is possible (36). Agronomic crops commonly remove about the same amount of K as they do N. Fortunately, most wastewaters also have about a 1:1 N:K ratio, but much more N can potentially be lost by leaching or denitrification. Therefore, K deficiency in most systems is unlikely, but soil available K should be routinely monitored (33).

Sodium, Chloride, Sulfur and Excess Salts

Sodium is of main concern with regard to its effects on soil permeability; excess Na on the exchange sites will cause loss of soil structure (flocculation), with drastic negative effects on the ability of the soil to accept and transmit water while excess salts decrease crop growth (13,40). Experience in irrigated regions of the arid west has shown that use of irrigation waters high in Na relative to other cations (principally Ca and Mg) can cause sodicity problems (15,25,40); problems with excess Na or salts are much less likely in humid regions. Sodicity is evaluated by measuring exchangeable Na relative to Ca and Mg. Chloride, an essential plant nutrient, can be present in sufficient amounts to reduce crop growth (phytotoxicity). These situations can be avoided by proper site management and monitoring.

Most wastewaters contain considerable SO_4-S. Sulfur deficiency in crops is relatively rare; except for air pollution, toxicity due to excess SO_4 has not been documented.

Micronutrients

Several elements, B, Mo, Mn, Fe, Cu and Zn, are essential to plants in small amounts, yet toxic in excess amounts. Deficiencies and toxicities are a rare occurrence in normal agricultural systems (9). Municipal wastewaters usually contain an ample amount of these elements to prevent deficiencies; toxicities due to municipal wastewater application are rare. Toxicity due to B is a distinct possibility if sensitive crops and a high-B wastewater is used (9,32). Copper and Zn toxicities could possibly develop after considerable time due to the tendency for these elements to be retained in the surface soil (9,35). Boron, on the other hand, readily leaches from most soils. Molybdenum could accumulate in plants in sufficient concentrations to be hazardous to grazing animals (35).

Other Elements

Other potentially hazardous elements present in trace concentrations in most municipal wastewaters include As, Hg, Se, Pb, Cr and Cd (29). With the exception of Cd, availability of these elements once they have reacted with soil constituents is low, and they are generally not considered a problem in land treatment systems (2). Cadmium, on the other hand, is of concern due to its potential for accumulation in the food chain (13). Hence crop, and perhaps soil, monitoring should be practiced to assure that Cd is not a problem. The extensive literature accumulating on Cd uptake by crops grown on sewage-sludge treated land indicates that yearly applications of 1 to 2 kg Cd/ha with a site lifetime maximum of 10 to 20 kg/ha can readily be tolerated (23,24). System monitoring for Cd should be required only if applications reach or exceed these limits.

Organics

Most wastewaters contain appreciable biodegradable organic matter. However, due to the ability of soils to rapidly degrade these materials, they are of little or no concern from a monitoring standpoint. While little information is available, wastewaters undoubtedly contain variable concentrations of persistent man-made organic chemicals. Their analysis is difficult and expensive. Further, they often are tightly bound in the soil, and thus their bioavailability would therefore be low. Wastewater monitoring will provide the first indication of the possibility of including persistent organics in a soil-plant monitoring program. In most cases, analysis likely will be confined to plant tissue since this will provide an indication of potential food chain magnification.

MONITORING PROGRAM

Philosophy of Soil and Plant Testing of Land Treatment Systems

Evaluating the effects of wastewater application on soils and plants is a necessary component of the total program. However, both soil and plant analysis have limitations with respect to proper interpretation. While much research is in progress to resolve these limitations, uncertainties will always exist.

Soils are characteristically nonuniform in chemical and physical properties. Vertical differentiation as a result of geologic incidents, soil formation, and of man's activities in the recent past, is readily recognized.

Often the lower horizons of a soil profile differ markedly from the upper one-half to one meter which is of importance to plant growth. Soils also vary widely in the horizontal plane, and this variability adds considerable uncertainty in soil testing procedures. The amount of time a soil has been exposed for weathering as well as differences in climate result in wide regional differences in soil characteristics such as the type and amount of clay, the pH, carbonate content, organic matter and buffering capacity of the soils. Within regions, nearly as wide variations can often be found depending on parent material, topography and drainage. Even within a given soil type in a field, microheterogeneities exist.

Since soil chemical properties markedly affect the reactions of critical nutrients in soils, it is not surprising that universal soil tests to evaluate bioavailability of these nutrients are not available. For example, several soil extractants to estimate available soil P are in use in soil testing laboratories in the U.S. (41). Soil tests for micronutrients and toxic metals are even less well researched. In fact, Melsted (29) considers the possibility of developing standard soil tests for these metals rather remote.

Nitrogen offers even more difficulties with respect to soil monitoring. Despite several decades of research effort, no valid, widely applicable test for estimating the amount of soil organic nitrogen which would become available over a growing season is available. Monitoring of inorganic N (especially NO_3-N) in the profile of soils receiving wastewater also presents formidable problems due to soil heterogeneity and the rapid rate of movement of NO_3-N under high water application rates. Intensive soil profile sampling is out of the question due to the expense involved while porous cup samplers(also known as suction lysimeters) generally give widely variable results which are difficult to interpret.

Plant analysis presently offers the most logical approach to monitoring the performance of land treatment systems with respect to many elements. Since the plant integrates the various soil and climatic factors involved in nutrient and heavy metal availability, analysis of the tissue for the concentration of these constituents will provide an indication of their availability. Further, methods for total elemental analysis of plant tissue are widely available and uniformly applicable. The average range in nutrient composition for numerous agronomic crops grown under a wide range of soils and climates, is available. However, these data usually are gathered under conditions ranging from deficient to sufficient in nutrients. What is much less well characterized is the upper tolerance levels, particularly of the nonessential elements (29). Progress is being made in defining these limits, particularly with respect to sludge application on land.

Soil Sampling Procedures

We propose that the soil sampling programs be developed for two separate objectives. One program should involve regular sampling at frequent intervals (at least yearly) of the surface soil horizon to the depth of tillage or the root zone. Additionally, sampling of the profile to a considerable depth should be conducted perhaps every third to fifth year to assess movement of P and metals.

It has long been accepted in soil testing that sampling variation is much greater than analytical variation and that the validity of a soil test is dependent upon the quality of the sample taken (37). A soil sampling program should provide not only an indication of the characteristic being monitored, but also a measure of its variability.

The first step is to define the sample unit. This must be done with respect to the soil type (the lowest unit in the natural system of soil classification) rather than a geographical area. For example, a 53 ha sprinkler irrigation system might be a geographical unit but two or even several soil types, with different chemical and physical properties, could be present in this area. Subunits of 2 to 8 ha often are arbitrarily suggested but the size of the subunit obviously varies with the degree of uniformity of the area (37).

Soil sampling generally involves mixing several borings to make up a composite sample, and obtaining one to four such samples per unit. We recommend that a minimum of two samples per unit be taken to obtain an indication of variability. There is no universally accepted number of borings needed for a sample since this also is a function of field variability. We recommend between six and twelve borings per sample. The samples should be thoroughly mixed, preferably in a laboratory, before subsampling. Sample handling and storage can markedly affect results of some tests, most notably those for inorganic N and exchangeable K. The most widely accepted approach is to air-dry the sample (for example, spread the sample thinly on a plastic sheet) as soon as possible after collection. High temperature (>40°C) drying should be avoided. The sample then can be ground to pass a 2-mm screen, remixed, and stored in an air-tight container. Great care throughout the sampling and sample preparation procedure must be taken to avoid contamination by heavy metals. Subsamples are then taken, as needed from the bulk sample. For analysis methods that use a small amount of material (e.g. the semi-micro Kjeldahl method), a relatively large subsample should be finely ground (<80 to <100 mesh) and again subsampled to obtain material for the analysis (8).

Tissue Sampling Procedures

There are generally two distinct objectives to tissue sampling programs: (a) Sample a representative portion of the entire harvested portion to determine either quality of the crop or total nutrient removal, or (b) sample a specific plant part (usually a recently-matured leaf) at a given growth stage to evaluate potential deficiencies or toxicities. The latter approach has been discussed by several authors (1,6,21,23,30).

The primary advantages of leaf sampling is that the sampling is relatively easy, it can often be done in time to correct the suspected deficiency, and considerable literature is now available to facilitate interpretation of the results, at least for most nutrient elements (1,30). However, there are several disadvantages. For one, elements are never distributed uniformly throughout the plant. For example, many crops have lower heavy metal concentrations in the grain than in the vegetative tissue. For another, nutrient composition changes, often markedly, during the growing season (30). Thus results of leaf tissue tests must be correlated with other indices such as crop yield or quality. In many cases, data to construct this type of relationship are not available (1).

While much more data on the relationship between diagnostic leaf tests and land treatment system performance is needed, we recommend that such tests be included as part of a routine sampling program, with sampling being conducted relatively infrequently (once every three to five years) and limited to the heavy metals. However, if poor plant growth is noted, specialists should be consulted. Plant tissue tests likely will provide valuable clues of system problems.

We also recommend that the plant monitoring program include determination of the yield and N and P concentration of the harvested crop. Heavy metals could also be analyzed on this material once every three to five years. If a forage or pasture crop is utilized in the land treatment system, we also recommend that the NO_3-N concentration be determined to evaluate potential NO_3 toxicity problems. Determination of other quality characteristics specific to a particular crop may be desirable (e.g., total digestible nutrients, protein, oil, storability).

In all cases, we recommend that a control site for each crop be maintained. This would be a site where normal management of the crop is practiced, and no wastewater is applied. Results of tissue tests from the control area will be valuable in interpreting the impacts of land treatment.

Once collected, plant tissue samples should be dried (65°C) as quickly as possible, ground and stored for analysis. Several wet digestion and dry ashing procedures are available for preparation of the sample for analysis (22).

Baseline and Infrequent Monitoring of Soils

Evaluation of system performance requires consideration of changes in various measurements over time. Unfortunately, preapplication data for the soils and vegetation are usually not obtained. While baseline vegetation data are useful only where the existing vegetation is used (e.g., irrigation of forests) the soils data are necessary. This is because the previously mentioned soil variability may result in samples from untreated areas being significantly different than those from the areas to be treated. Further, soil samples obtained before land treatment is initiated should be stored indefinitely. If methods change, or if analysis for a previously unsuspected contaminant should be desired, the initial soil samples then can be reanalyzed.

The baseline data recommended are given in Table 1. These measurements are largely those required in site characterization for system design. Additionally, we recommend a comprehensive profile analysis for metals and P so that the rate of leaching of these constituents can be determined from subsequent analyses.

The references listed in Table 1 are intended only to direct the reader to the appropriate literature. Further information on appropriate methods can be obtained from USDA, EPA, or University personnel.

Table 1. Recommended soil analyses to be taken before system initiation and once every three to five years.

Analysis	Profile depths(cm)	Reference
Total N (Kjeldahl)	0 to 5; 5 to 10; 10 to 15; 15 to 30; 30 to 60	8
Organic C	As for total N	4
Carbonate C	As for total N	5
Cation exchange capacity	0 to 30; 30 to 60	10
pH in H_2O and 0.01 $\underline{M}$ $CaCl_2$	As for total N plus 60 to 90; 90 to 120; 120 to 150	27
Lime requirement	0 to 15	27
"Available" phosphorus	As for pH	As recommended
"Total" Cu,Zn,Ni and Cd by hot HNO_3-H_2SO_4 or HCl[a]	As for pH	12
"Available" Cu,Zn, Ni and Cd	As for pH	As recommended

[a]Total analysis by more complete digestion procedures could also be used. The hot-acid method is recommended as it normally extracts >90% of the soil metals and is much easier to do.

Yearly Soil Monitoring Data

Yearly monitoring, preferably in the autumn of the year, should be conducted with the primary objective of evaluating the suitability of the land treatment system for sustaining optimum plant growth and to check for possible deleterious effects due to salinity and sodicity. In essence, this is a soil fertility testing program, and appropriate USDA and University Extension personnel should be consulted for advice on analyses to be conducted and their interpretation. Table 2 lists indices which may be included. The sampling should encompass the major portion of the root zone (e.g., 0 to 15 cm).

Table 2. Recommended yearly soil analyses.

Analysis	Reference	Comments
Soil pH in H_2O or 0.01 <u>M</u> $CaCl_2$	27	If acid (<6.0 in H_2O) a lime requirement test (27) should be conducted and corrective lime additions made.
Exchangeable Ca, Mg, K and Na	18,38,39	Exchangeable Ca, Mg and K should be adequate; Na not more than 10% of the cation exchange capacity.
"Available" P	33	As recommended.
Soluble salts	7	If >4 mmho/cm in saturation extract, corrective measures should be initiated.

Plant Monitoring

Suggested plant tissue tests are listed in Table 3. The character of the wastewater may dictate that additional analyses be conducted, as indicated in the table. Unfortunately, it is easier to obtain than to interpret many of these data. While it is tempting to present generalized normal composition and toxic level values for plants, the many variables that go into plant tissue composition of elements dictate that specific advice be sought should problems be suspected. For hazardous elements, particularly Cd, specific food levels have not yet been established. A suggested guide for assessment of potential metal toxicity problems is given in Table 4. The values are modified from those given by Melsted (29) and it must be emphasized that they remain largely judgement values.

It is imperative that "control" values obtained from the same crop, sampled at the same time on similar soils that have never received wastewater, be available. These control data will greatly aid in interpretation of the plant tissue monitoring results.

ASSESSMENT OF MONITORING PROGRAM

We have presented a wide array of potential soil and plant analyses. As was emphasized initially, the frequency and number of analyses required will be site-specific. It is impossible to attempt to establish firm recommendations. As we also have indicated, mere collection of data is not a monitoring program. Data interpretation and evaluation must be an on-going activity.

A major omission in our recommendation is the monitoring of NO_3-N in the soil. We feel this presents more difficulties than the effort warrants. Rather, we recommend that this be included as part of the ground water or underdrain monitoring program.

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Table 3. Suggested analyses for plant tissue.

Plant part	Analyses	Frequency	Comments
Harvested portions	Total N,P	Yearly	To establish removal efficiencies. Other quality tests may be desired.
Grasses, forage or silage	NO_3-N	As used	Minimize potential NO_3-N poisoning of livestock. Problems possible if >1,000 mg NO_3-N per kg of dry material (42).
Harvested portions or leaf tissue	Cu,Zn,Ni,Cd	Once in each 3 to 5 years	Establishes trends in metal uptake.
Harvested portions or leaf tissue	Other potentially hazardous substances in waste. Se,Hg,As,B,Cl, Pb,Cr, persistent organics		Frequency will depend upon potential hazard. Yearly analyses may be required until trends in uptake are established.

Table 4. Average composition range of selected elements in leaf tissue of common agronomic crops and suggested tolerance level to be used for monitoring purposes.

Element	Average composition range[a]	Suggested tolerance level[a]
	—— mg/kg dry plant material ——	
Arsenic	0.1 to 4	>5
Boron	30 to 75	75
Cadmium	0.05 to 0.5	2
Chromium	0.1 to 1.0	2
Cobalt	0.01 to 0.5	5
Copper	5 to 20	50
Lead	0.1 to 5	10
Manganese	15 to 150	500
Molybdenum	0.2 to 1.0	3
Nickel	0.1 to 1.0	50
Selenium	0.05 to 2.0	3
Zinc	15 to 200	400

[a]Taken from 2,3,21,23,24,29. Values are for recently matured leaf tissue.

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MONITORING REQUIREMENTS FOR LAND TREATMENT SYSTEMS

MONITORING OF MICROBIOLOGICAL AEROSOLS AT WASTEWATER SPRINKLER IRRIGATION SITES

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ABSTRACT

The application of wastewater to land by spray irrigation may generate aerosols containing pathogenic microorganisms which can migrate beyond the wetted zone. Aerosol monitoring may have usefulness in assessing the potential human exposure and the infectious disease risk relative to other sources of exposure. Microbiological aerosol sampling can be performed with a wide variety of samplers including inertial impactors and impingers, electrostatic precipitators and scrubbers. Simultaneous, continuous meteorological information is required to insure valid sampling and is also required for predictive mathematical aerosol modeling. A successful monitoring program should include a quality assurance program for sampling and microbiological assays. The objectives and protocols should be well planned and strictly adhered to. Field logistics problems such as sampler sterility, maintenance and repair, sample shipment and processing, and acquisition of flexible, dependable field personnel must receive careful attention. To provide statistically valid aerosol data for interpretation and modeling requirements appropriate types of samplers in adequate numbers must be used. Several predictive models are available to provide estimates of downwind aerosol migration. The models incorporate physical decay and dispersion. Some attempt to predict biological dieoff and the proportion of water aerosolized.

INTRODUCTION

The use of land application as a means of wastewater treatment is accompanied by a concern for potential infectious disease hazards to workers and surrounding populations. The human health hazard from aerosolization of treated wastewater is not well defined (1, 2). Spray aerosols near land application sites abound with a wide range of microorganisms, including enteric viruses, bacteria, fungi and protozoa. Downwind migration and dispersion of microbiological aerosols, as well as the associated dieoff (biological decay) and fallout (physical decay) have been investigated at several locations (3-8). However, information gaps remain. Exposure levels constituting an unacceptable increased probability of human infection have not yet been defined. In addition, the human dose response via aerosol dispersion has not been adequately determined. Unless future research indicates the absence of any health hazard, some form of aerosol monitoring or other risk assessment may be required.

The objective of this presentation is to provide researchers, site management and operations personnel, and health enforcement agencies some indications of (a) methods of aerosol monitoring at spray irrigation sites (b) technical and logistical approaches and problems, (c) sampling and assay techniques, (d) data evaluation methods and (e) mathematical modeling approaches and model validation.

Aerosol Formation During Spray Irrigation

Common types of spray equipment used in wastewater application include impact and fan sprinklers, rain guns and fixed-aperature rocker-arm sprayers. All of these produce aerosols that travel beyond the wetted zone of application (9). The smaller droplets produced in spray activities are quickly reduced in size by evaporation, yielding still smaller droplet nuclei which may remain aerosolized for prolonged periods. Most droplet nuclei produced by spray irrigation with wastewater are greater than 0.5 μm in diameter. The term aerosol is generally applied to airborne particles up to 50 μm in diameter.

Microbial aerosol concentrations during spraying are dependent on a number of factors. One is the concentration of microorganisms in the wastewater itself. Another is the aerosolization efficiency of the spray process, i.e., the proportion of water sprayed that enters the aerosol state. This, in turn, is influenced by a number of interrelated features (9) including nozzle size and pressure, angle of spray trajectory, angle of spray entry to the wind and other mechanical features such as the use of impact devices to propel the sprinklers.

Factors Influencing Microbiological Viability and Migration Downwind

Once aerosolized, microorganisms contained in droplets or droplet nuclei are subjected to a number of stresses which influence their survival. As the aerosol is formed, there is an immediate aerosol shock which may, within seconds, reduce the level of viable microorganisms by over an order of magnitude (7, 10). However, after the initial shock, the continued aerosol dieoff is relatively slow and is highly dependent on temperature and relative humidity. Important factors in microbial aerosol inactivation are desiccation, osmotic changes and ultraviolet or ionizing radiation. On the other hand, partial desiccation may concentrate materials within the droplet that protect the microorganisms. Moreover, for larger particle sizes there is an increased proportion of particles bearing more than one viable organism. This leads to greater apparent resistance, since the particle is rendered non-viable only upon inactivation of every contained organism.

Microorganisms in aerosol particles are also influenced by meteorological and geographical factors. The rate of physical aerosol decay (deposition to the ground) is influenced by wind velocity (higher winds keep particles aloft to greater distances), atmospheric stability (turbulent air hastens dispersion of aerosol plumes), and local topography.

Human Health Implications

Pathogen-containing aerosol particles of any size are considered potential health hazards after they have settled or impacted upon surfaces such as tools, equipment or clothing, which may be handled by humans with subsequent ingestion. This overview is most concerned with the inhalation of 0.5 μm - 5 μm aerosol particles, which are most likely to be desposited on the bronchial or alveolar passages of the respiratory tract (11).

The evaluation of infectious disease hazards in wastewater aerosols must emphasize the concentration of pathogens to which a person is exposed over a limited period of time. Thus the concentration of pathogenic aerosols at any downwind location is the primary criterion to be used in health risk assessment.

Aerosol Samplers for Microorganisms

A variety of aerosol samplers designed for or adaptable to environmental sampling of microorganisms is currently available. Generally these samplers can be put into 3 groups: inertial impingers/impactors, electrostatic precipitators and scrubbers. Table 2 describes some representative samplers and tneir capabilities.

Sampler Efficiency and Calibration

It is essential to know the collection efficiency of the sampling equipment used especially for aerosol particles in the size range of greatest human health concern. Also the collection efficiency of each sampler type should, when possible, be measured in laboratory quality assurance tests for each aerosol parameter to be studied in the field.

A majority of droplet nuclei encountered downwind of spray irrigation operations fall between 0.5 μm and 5 μm diameter, the range of maximal pulmonary deposition. In this size range, there are several widely used types of samplers that exhibit good microbiological collection efficiency as shown in Table 2. Collection efficiency decreases sharply at the lower end of the particle size range

Measurements of identical aerosols using the AGI-30 or Andersen sampler are often used as a standard against which to

Table 1. Characteristics of Representative Samplers

Sampler Type	Representatives	Sampling Capacities (l/min.)	Features: good	Features: bad
Impactors	Multislit impactors	12-2000	easy to use	low effic. for small particles
	Andersen multistage	28	particle size discrimination	sampler overcrowding
Impingers	AGI, Greenburg-Smith	12-24	inexpensive, std. sampler to rate others	low effic. for small particles, low volume
Electrostatic Precipitators	LEAP & Litton model M	1200 & 10,000	high volume & recovery	expensive, complex
Scrubbers	Venturi scrubbers	25 or 850	high volume	not commercially available

measure other samplers or collection techniques. Factors adversely affecting sampling efficiency fall into two major categories; mechanical losses and losses in viability.

Mechanical collection efficiency can be assessed by using an aerosol of known strength consisting of an aerosol-stable organism such as Bacillus subtilis niger spores and comparing actual to predicted recovery. A second approach to estimation of mechanical efficiency relies on measurement of the slippage of aerosol particles through the sampler. A second sampler of the same kind is used in tandem with the first for this determination; aerosol recovery in the second sampler indicates slippage. For the Andersen sampler; if recovery on the last stage is close to zero, very limited slippage is indicated for any aerosol particles above 0.5 µm.

Important factors that reduce mechanical collection efficiency in the field are: line voltage drops affecting airflow in samplers, partially clogged air intakes or critical orifices, improper volume of fluid in impingers or of collection agar in impactor samplers and incomplete wetting of precipitator sampler collection discs. Reduced electrostatic potential in electrostatic precipitator samplers may also cause a decline in aerosol sampling efficiency. Using the LEAP sampler, we observed the following collection efficiencies for fluorescein-methylene blue aerosols with median particle diameter 2.0 µm (assume efficiency at 13 kilovolts (kV) = 100%): 9 kV - 75%, 7 kV = 50%.

Use of disposable plastic petri dishes in Andersen samplers was reported by Andersen (1958) (14) to result in lowered sampling efficiency in relation to glass, but Dimmick (Dimmick and Akers 1969) (16) found no significant difference. In laboratory studies, using bacterial aerosols in the 2 to 5 µm particle size range, we have not detected reduced efficiency of the Andersen plastic dish system using the AGI-30 as a standard.

Viability losses may result from prolonged sampler operation. In an impactor sampler such as the Andersen, this is dependent on microbial species and humidity, but 30 to 40 minutes is probably a maximal sampling time in the absence of an evaporation retardant. In laboratory studies we observed a 30 percent loss in impacted Klebsiella and E. coli in 40 minutes in the Andersen sampler at 50-75% relative humidity. On the other hand, in the LEAP sampler, recirculation of aerosolized E. coli and S. marcescens for 40 minutes at 30-40 percent relative humidity and 12 kilovolts potential, did not cause measureable viability losses.

The collection efficiency was not affected by microbial aerosol concentration in tests performed in our laboratory.

General discussions of aerosol sampling and aerosol behavior have been provided by Fuchs (1964) (16), Dimmick and Akers (1969) (15) and Friedlander (1977) (17).

Table 2. Aerosol Collection Efficiencies of Selected Sampler Types in the Particle Diameter Range 1 to 5 μm

Air Sampler	Collection efficiency %	Data Source
Small-Volume Samplers		
AGI-30	90 to 99	Tyler and Shipe (1959) (12)
Multistage liquid impinger	80 to 90	Dimmick and Akers (1969)(15)
Midget impinger	83	Tyler and Shipe (1959)
Andersen viable	up to 99	Various sources
Large-Volume Samplers		
Multislit liquid impinger	15 to 60	Decker et al. (1969) (13)
LEAP electrostatic precipitator	90 to 95	Manufacturer's literature (Environmental Research Corp)
Litton Model M	75 to 95	Decker et al. (1969)
electrostatic precipitator	90 to 99	Manufacturers literature

Basic Tasks for Field Sampling

To properly assess microbiological aerosols several analytical operations should be performed. The wastewater, at the spray nozzle or lagoon, should be sampled and analyzed for microorganisms of concern during every aerosol test. If the water comes directly from a wastewater treatment plant without storage, the samples should be taken frequently enough to detect diurnal microbial fluctuations. Also, chemical tests should be performed on the wastewater to include such parameters as BOD, TOC, total and volatile solids (dissolved and suspended), pH, ammonia and Kjeldahl nitrogen and, if disinfected, the active residual disinfectant. The aerosolization efficiency of the specific spray machinery at normal operating line pressure should be determined by performing dye studies (dyes experience only physical and not biological decay) with aerosol measurement at close downwind points. The microbial aerosol monitoring program should be designed to obtain statistically valid sampling for an assessment of aerosol concentrations, particle sizes, dieoff and downwind dispersion. It is also important that meteorological data be gathered to determine the effective sampling periods; this is also a necessary input for aerosol dispersion equations and for mathematical modeling. Together, these test parameters can be used to reveal the extent of downwind microbial aerosol travel under a range of conditions.

Use of Air Samplers in the Field

All-glass impingers such as the AGI-30 or the Greenburg-Smith are useful for sampling both tracer dyes and microorganisms. Andersen six-stage impactors are commonly used because they differentiate the aerosol into its various particle size ranges. These or other small sampling equipment are best adapted to close-in sampling. For high volume air sampling, which is necessary for low concentration aerosols or distant downwind sampling, electrostatic precipitators (LEAP or Litton Model M) are the most efficient. Many samplers exhibit shortcomings in field operation because: some must be carefully leveled, none can sample air isokinetically at various wind velocities, orifices often can't be maintained into the wind, liquid impinger surface film or sample volume is difficult to maintain; dehydration of semisolid medium or over-sampling (crowding of sampled organisms) must be checked and can limit sampling time. Furthermore, the more complex samplers are prone to mechanical or electronic breakdown and also may present problems in the maintenance of sample sterility.

In general, it is necessary to use several sampler types in unison. In this way the need for several sampling capabilities, i.e., high sensitivity, particle size discrimination and precision of estimate, can be met, and the aerosol plume can be reasonably described.

Also multiple samplers help provide a statistically valid determination of dieoff, particle size distribution and concentration of organisms at any specific downwind point. It is of importance to set the samplers at a uniform elevation of approximately 1.5 to 2 meters to represent the height of aerosol inhalation by humans.

There are two principal approaches to aerosol sampling for determination of the concentration and nature of downwind microbial aerosols. The first approach is to set a number of samplers close together and close to the spray source, possibly 30-50 meters beyond the wetted zone. In this manner it is possible to obtain a statistically valid set of aerosol values for regions near the source. However, estimates of maximum downwind migration (i.e., distance required to resume ambient background aerosol levels) must then be obtained by extrapolation and must rely heavily on meteorological data and dispersion equations.

The second approach is to measure the aerosols at some distant point of interest, such as property lines, buffer zone boundaries or housing areas to determine if background levels are being exceeded. To perform such sampling, large volume samplers and/or long sampling times must be utilized, and generally a relatively large number of samplers would be required to provide statistical validity. To its credit, this sampling procedure provides the most accurate estimate of the extent of downwind survival. Sampling at distant locations requires extreme care to insure that the microorganisms recovered are from the spray source and not from residential, commercial, animal or other background sources. Background sampling (during periods between spray irrigation) should be performed to determine the nature and extent of these aerosol sources.

The positioning of samplers may be tailored to site-specific objectives. Often, especially in research studies, it is desired to have both close-in and distant aerosol measurements. One common approach is to determine beforehand the mean wind direction and then to set up samplers directly (or in an array) in line downwind. This method is quite successful when the wind direction is stable. However a shift in wind direction of even 20° to 30° before or during sampling may require termination of a run or realignment of samplers. Another method is to orient a sampler line crosswind or parallel to a line of sprinklers. A greater range of wind direction can then be accepted.

The choice of the sampling procedure to be used is finally dependent on the size and type of aerosol source (point, line or field source), number and type of samplers available (both small and high volume), and available personnel and facilities. One or two persons obviously could not perform a field test where a large number of samplers, some of which require almost constant attention, are used and where complex assay procedures required for bacterial and viral pathogens are desired.

Meteorological Data Requirements

On-site meteorological information is essential for determining appropriate conditions for initiation or continuance of sampling and for providing input to predictive dispersion equations. Continuous recording of wind direction and wind speed is necessary. Data on the solar intensity, angle of the sun and cloud cover may provide useful data related to both atmospheric stability and biological decay rates.

Logistical Problems

Because of the nature of field sampling, the logistics are formidable. Numerous equipment items, and in many cases a field laboratory facility, are required. Samplers, air pumps and most meteorological equipment require AC current. This necessitates a main electrical connection from a nearby source or field generators, hundreds of feet of electrical line, and circuit breakers. Another logistical problem is that of supplying adequate parts and repair capability for field equipment. This problem is typically acute with high volume air samplers, which, because of their many electronic and mechanical components are often out of service. Additional problems in the field are: maintaining sample sterility before and after sampling; providing communication in the field and from field to laboratory; providing transportation of sampling equipment to and from field stations and rapid return of samples to the laboratory

Field experience demonstrates that careful planning and appropriate alternative sampler deployment patterns are a necessity. It is also important to have a staff that is flexible to changes in working hours and that will persevere in accomplishing the sampling task.

Sampling of Aerosols and Wastewater

Methods used in field sample collection and laboratory analysis of samples have been discussed (6, 18).

For determination of microbial aerosols, both semi-solid medium (Andersen sampler, slit sampler) and broth or tissue culture fluid (LEAP, Litton samplers) are used. The semi-solid medium used may be a general purpose one, such as tryptose glucose-extract or standard plate count agars, which measure the overall aerobic bacterial population. Alternatively, differential or selective media may be employed for estimation of specific microbial groups, e.g., Endo agar for coliform bacteria. A procedure has been used in our laboratory to sample airborne coliphage directly, using Andersen samplers. The plates are overlaid with soft agar seeded with host E. coli. Overlay may be performed either before or after sampling, but incubation at the appropriate temperature must be initiated immediately after sampling.

Aerosols of fluorescent tracer dyes are sampled by liquid impingement in distilled water containing antifoam (AGI) or a wetting agent (LEAP, Litton Model M). Liquid collection media used for either microorganisms or dye should be sampled prior to exposure, as a control on sampler contamination.

Prior to sampler operation a number of procedures should be performed. Andersen sampler sieves should be washed, checked for any occluded apertures and disinfected by ethanol or isopropyl alcohol wiping between runs to eliminate sample contamination. The assembled sampler with critical orifice (orifice used to standardize sample volume) should be frequently checked for air leaks by stoppering and applying a vacuum. Agar volume per plate should be sufficient to allow 2.5 mm clearance between agar and sieve (14). Contact between the agar surface and the sieves should be avoided. AGI sampler orifices should be checked and should be held in an upright position by cylindrical cups or holders affixed to the sampler stands.

Studies in this laboratory have used continuously recirculating medium in the LEAP and Litton model M samplers in the field to concentrate the microorganisms sampled. Evaporative losses were restored by addition of sterile distilled water during sampling, as needed. These samplers should be leveled with care to facilitate even distribution of sampling fluid and complete wetting of the collection (impingement) disc. Sterilization of large volume samplers is also extremely important between runs, since contaminant organisms may multiply on interior sampler surfaces. We have used 0.1% HOCl, pH 6.5, to disinfect inlet and collection tubing. This is followed by sterile sodium thiosulfate solution and sterile water. The collection disc is cleaned with detergent solution and disinfected with 70% alcohol. Disinfection may also be achieved by running steam through the system.

Samples of the wastewater being sprayed should be obtained from spray heads or the irrigation system reservoir simultaneously with air sampling. For indigenous organisms, two or three grab samples taken from a well-mixed reservoir during runs will suffice, and these may be composited. If tracer seeded material (dyes or microorganisms) is used, a larger number of samples, distributed in time over the entire sampling period should be taken. These should not be pooled, as a check on fluctuation in tracer level at the spray head.

Analysis of Samples

All liquid samples containing microorganisms should be placed in ice promptly to await removal from field to laboratory. Otherwise, dieoff of some organisms, especially injured ones, may occur due to cumulative stresses, while others, in rich medium, may undergo multiplication. Analysis of samples should take place on the same day even if samples must be shipped to a distant laboratory.

Liquid samples are analyzed for standard plate count, indicator organisms or other bacteria or viruses according to any appropriate procedure. Standard Methods (19) should be followed when possible. Sensitivity may be maximized by concentration procedures, e.g., membrane filtration.

Exposed agar medium (Andersen sampler, slit sampler) is simply removed from the samplers for incubation. Each Andersen sampler plate count is subjected to a positive-hole statistical correction (14). From each Andersen sample, consisting of six plates, a median particle diameter for colony forming particles is calculated (14, 15).

In studies using fluorescent dye, fluorimetric determinations are made on all liquid samples including zero time samples from AGI's, electrostatic precipitator or other sampling equipment. Wastewater grab samples are also measured.

Since the degree of fluorescence is pH-dependent, care must be taken to avoid any significant difference in pH between wastewater and aerosol samples at the time of fluorimetric reading.

Field Data

Two field studies were performed at Ft. Huachuca, AZ, (4-6) in which the aerosol output of a single impact sprinkler, applying chlorinated or nonchlorinated secondary effluent, was measured. In Table 3 are presented mean observations for the aerobic bacterial aerosol (standard plate count) obtained by Andersen samplers. Nighttime runs have been averaged separately, and certain meteorological data are included. Individual standard plate count observations 42-49 m downwind from the sprayer ranged up to $10^4/m^3$, with mean values as shown. Bacterial levels at the greatest distances sampled, 120-152 m, were about one order of magnitude lower. Under nighttime conditions, characterized by lower wind speeds and increased atmospheric stability, aerosol levels were somewhat greater than under daytime conditions. Chlorination, ca 6 mg/l total residual, reduced bacterial levels in the wastewater by more than 3 orders of magnitude, and the bacterial concentration in aerosols to near-background values.

Field aerosol observations at Deer Creek Lake, Ohio, are discussed elsewhere in these proceedings (18). At this site total source strength in colony forming units (CFU) per second averaged 2- to 3-fold less than at Ft. Huachuca, while aerosol strengths measured downwind at 21-50 m averaged $5 \times 10^2/m^3$, within the range of corresponding values in Table 3. In 14 experimental runs, the net aerosol strength at 200 m averaged 8 percent of that at 30 m.

At the Arizona site, total presumptive coliform levels (using the membrane filter method with m-Endo broth) comprised 0.5 percent of standard plate count in the Ft. Huachuca wastewater and averaged 2.0 percent of standard plate count in the aerosols. Low source strength coupled with sensitivity to chlorination, seriously limits the utility of the coliform group as an indicator of the presence of microbial aerosols at land treatment sites. Approximately half of the presumptive coliforms were classified as Klebsiella, and LEAP sampler estimates of the viable Klebsiella aerosol 46 m downwind ranged from $1/m^3$ to $23/m^3$.

Bacteriophage (coliphage) f2 was used as a tracer (ca 10^5 PFU/ml final concentration) in a number of runs at Ft. Huachuca. The f2 aerosol was collected by both LEAP and Andersen sampler with post-exposure overlay using seeded soft agar. Prior laboratory experiments had shown an efficiency for this method of collection equal to ca 23%, using the AGI-30 as a standard. Field observations were multiplied by a factor to compensate for this. As a tracer in studying wastewater aerosols, f2 was shown to have several advantages; (1) It is much more resistant to chlorination than standard plate count or coliforms (2) like coliform

Table 3. Bacterial Aerosol Strength, Colony Forming Particles (CFP)/m^3, Observed Under Various Conditions, Without Effluent Chlorination, at Ft. Huachuca, AZ (mean values)

Year	Daytime or Night	No. of Runs	Rel. Hum. %	Wind Speed (mps)	Temp. (°F)	Source Strength* CFU/ml	Source Strength* CFU/sec	Aerosol Levels (Geom. means) Nearest Point Distance (m)	Nearest Point CFP/m^3	Farthest Point Distance (m)	Farthest Point CFP/m^3
1974	Day	9	55	5.1	64	3.3×10^5	2.1×10^9	49	4.3×10^2	139	8.6×10^1
	Night	4	63	3.2	52	3.7×10^5	2.4×10^9	45	5.6×10^2	122	1.7×10^2
1975	Day	9	20	3.8	74	2.1×10^5	1.6×10^9	46	1.4×10^3	120	1.3×10^2
	Night	3	10	2.6	72	3.9×10^5	2.7×10^9	46	6.3×10^3	152	4.1×10^2

*Total applied water

bacteria the corresponding ambient aerosol is essentially zero, and this absence of background facilitates detection of the aerosol at lower concentration than the general bacterial aerosol and (3) It is more aerosol-stable than coliforms at intermediate. though not at low (Table 4), relative humidity levels.

Fluorescent dye (sodium fluorescein) was used as a tracer, at 1 to 10 mg/l final concentration, in several runs at each field site studied. Model predictions of the resulting aerosols, which are free of a biological dieoff factor, were made using plume dispersion equations. Since the model assumes that 100 percent of the applied water is aerosolized, the ratio observed to predicted aerosol strength estimates the aerosolization efficiency. These estimates have ranged from 0.2 to 0.85 percent of the total wastewater sprayed.

Similar dispersion model estimates have been made for microbial aerosols. Such estimates assume zero dieoff. Because dieoff actually occurred, the resulting ratio: observed/predicted aerosol strength is lower than for fluorescein. From this difference in aerosolization efficiency, percent viability loss on aerosolization for each organism was calculated according to the formula: fractional viability loss =

$$1-\frac{\text{efficiency, microbe}}{\text{efficiency, dye}}$$

Combining data from the two sites, aerosolization efficiencies and percent viability loss are given in Table 4 (20).

Table 4. Mean Aerosol Efficiencies and Viability Losses for Ft. Huachuca and Deer Creek Lake Trials

Aerosol	Aerosolization Efficiency (Geom. mean)	Viability Loss (percent)
Fluorescein	8.48×10^{-3}	0
Standard Plate Count	2.10×10^{-3}	75
Presumptive Coliforms	2.35×10^{-3}	72
Coliphage f2	4.83×10^{-4}	94

A very substantial fraction of bacteria- and coliphage-bearing particles, 50 percent for the Ft. Huachuca observations and 75 percent for those at Deer Creek Lake, fell within the size range 1 to 5 µm.

STATISTICAL CONSIDERATIONS

Biological aerosol studies are subject to high levels of innate statistical variability. In the field, many sources of variation exist due to the lack of control over relevant source and environmental conditions. Numerous technical, logistical and economic factors affect the variability and statistical significance of the data. The degree to which these factors will introduce constraints will vary and be dependent on the objectives of the program (Table 5).

Table 5. Factors Influencing Statistical Variability During Aerosol Monitoring Programs

1. Wastewater concentration and variability (daily, seasonal, etc.) of specific microorganisms to be monitored.
2. Sample handling and microbiological assay procedures.
3. Efficiency, flow rate and calibration of aerosol samplers.
4. Type of irrigation machinery.
5. Local meteorological and topographical characteristics.
6. Duration of monitoring program and frequency of sampling.
7. Magnitude of quality assurance effort.

The initial step in designing a monitoring program should be definition of the desired level of statistical reliability required. These requirements should be consistent with the overall objectives of the monitoring program. Obviously, the more stringent the statistical requirements, the greater will be the cost and time investment.

Expertise in statistical experimental design will be needed to estimate the level of effort necessary to meet the desired objectives. Estimates of the expected variability of the data due to the factors listed in Table 5 will aid in determining such requirements as number of simultaneous samples (samples per trial), number of sampling trials, sampling volumes, etc. Unfortunately, due to variability in site characteristics, only limited data are available to facilitate such estimates. The most comprehensive sources of such, developed during studies at Pleasanton, CA and other sites are described by Johnson, et al., (7) Camann, et al., (21) and Bausum, et al., (8). A pilot monitoring program may be required to obtain better estimates of the expected statistical variability and the necessary level of effort.

It must then be determined if the objectives can be achieved with the resources allocated. If not, attempts should be made to obtain additional resources or to relax objectives, which is favored over attempting an ambitious effort with limited resources, as the later may fail to provide any useful information.

We wish to emphasize the overwhelming impact statistical reliability requirements have in generating constraints on an aerosol monitoring program. Each additional increment in sensitivity or statistical reliability desired in state-of-the-art aerosol monitoring methods is associated with a disproportionate increase in cost and time investment. A single example, that of the Deep Creek Lake study, will suffice to demonstrate this trend.

The principal objective of this study was to characterize the aerosol source strength from this field source aerosolization system, in which wastewater from a recreational area received prolonged holding in lagoons and chlorination. Due to limitations in resources, it was decided that the program would emphasize evaluation of one microbiological parameter, the standard plate count. Sensitivity was constrained by the non-availability of a sufficient number of high-volume samplers. Thus, Andersen samplers were used predominantly. Since the objective was to characterize aerosol source strength, the majority of the samplers were positioned close to the spray field. The Fort Huachuca study had indicated that statistical reliability for such a study could be improved by placing aerosol samplers side-by-side at each sampling distance rather than along a line running downwind from the source. This permits a determination of variability arising from sampling, handling and analytical processes (Table 5). Characterization of aerosols close to the source dictated a close-in sampler row. However, the prior studies had also shown that, if samplers were placed too close to the source, increased variability would result due to discrete conbutions of individual sprinkler heads. Thus, two rows, each with four Andersen samplers on 3 m centers were placed at approximately 25 and 45 meters downwind from the spray field. From twenty aerosol sampling trials, the average aerosol concentration levels at the 25 and 45 meter distances were 469 and 413 viable bacteria/m^3, respectively. The standard deviation within rows associated with these measurements was ascribed to differences in the measurements made by adjacent samplers. It was calculated that to obtain 90% or 95% confidence that average aerosol concentration measurements, collected within the aerosol plume, at each distance were within 10 percent of the true value, 19, or 30 Andersen samplers would have to be positioned side-by-side. It is interesting to note that only 7 or 9 samplers are required for ± 20 percent departure from true value. Unfortunately, information was not available prior to this study to predict this degree of variability. Indeed, prior laboratory determinations had indicated far less sampler variability. While the data remain useful, they have less statistical reliability for comparison purpose than had been hoped.

The following sampling strategies, which apply to most sites, may reduce statistical variability or insure that it remains relatively constant.

1. Collect a large number of aerosol samples at a limited number of downwind distances. Model determinations based on accurate aerosol concentration measurements at a limited number of sampling distances (minimum of two) have appeared more reliable than outputs based on highly variable concentration measurements from numerous downwind distances.

2. Within the monitoring program's resources, use the most sensitive sampling equipment available. The sensitivity of the sampling instruments must exceed the required overall sensitivity for the monitoring program.

3. Have a well defined and strictly managed protocol. All sampling, sample handling and microbiological assay procedures must be standardized to prevent additional sources of statistical variability in the data. Changing procedures during the course of the program should be undertaken only after assessing the impact these modifications will have on

the accuracy and precision of the entire program.

4. A continuing quality assurance program for aerosol sampling and microbial assay should be included in the monitoring plan. An investment of 10-20 percent of the budgetary and personnel commitment may be required to insure the reliability of the data being collected.

5. Statistical expertise must be available throughout the program, providing statistical input during the design phase and periodic evaluations of the sampling data and the results from the quality assurance program.

Mathematical Modeling

Several modeling approaches have been undertaken to describe the generation of viable aerosols by wastewater sprinkler irrigation systems and to predict downwind aerosol concentration. The intent is to develop a model that is uniformly predictive for those factors influencing aerosol generation, dispersion and dieoff that are not site-dependent. Such a model would ideally require only the input of; (a) the wastewater microorganism concentrations, (b) the sprinkler irrigation machinery aerosolization efficiency and (c) local meteorological and topographical information. The ultimate goal would be to eliminate the necessity for long term aerosol monitoring programs at all wastewater land treatment sites.

Mathematical modeling has also been a necessity when the goal of the aerosol monitoring program has been to measure or predict aerosol concentrations at relatively distant, i.e. greater than 200 meter, downwind locations. Micrometeorological conditions often render aerosol plume travel complex. Therefore, atmospheric dispersion models are also employed to assist in the placement of downwind aerosol samplers. More importantly, limited sensitivity of samplers often prevents the collection of measureable concentrations attributable to the sprinkler irrigation system. Thus, modeling often becomes the only practical means of viable aerosol estimation at distant locations.

Modeling efforts to date have used atmospheric diffusion models from air pollution monitoring programs (22-24). These models are often satisfactory in predicting aerosol behavior and plume distribution in relation to meteorological conditions. The existing models however, fail to account for losses in aerosol strength due to biological dieoff. Since the microbiological aerosols must be viable to initiate infection and, thus to be of human health concern, a useful model must accommodate the factors that affect microorganism survival.

Camann, et al., describe a recent effort to develop such a model (21). The simplified form of this model estimate of the aerosol concentration (P) at a particular downwind location is:

$$P = D \times E \times I \times e^{(a - a_{50})}$$

where:

- D = physical diffision model estimate, of downwind aerosol concentration; assumes all wastewater sprayed becomes aerosolized and no biological dieoff.
- E = Aerosolization efficiency factor. That fraction of the sprayed wastewater that leaves the site (beyond 50 m.) as aerosol.
- I = Microbiological impact factor. Cumulative number representing several interactive factors that influence microorganism survival and aerosol concentration during the initial seconds (0-50 m downwind) of aerosolization (I>0).
- λ = Microorganism dieoff rate at distances greater than 50 m from source.
- $a - a_{50}$ = Aerosol age from 50 m downwind to the location where concentration estimate (P) is made.

The model's development still requires verification at operational sites, but it has been relatively successful in describing physical diffusion (D), aerosolization efficiency (E) and the microorganism dieoff rate (λ). A major limitation in verifying these model components has been associated with inadequacies in the state-of-the-art for obtaining reliable microorganism concentrations in effluent and aerosol samples.

Refinements in the model are probably necessary to separate the individual components presently included in the microbiological impact factor (I). This factor was initially postulated to account for the expected rapid microorganism dieoff rate (>λ) during the initial seconds (between 0-50 meters) of aerosolization. Thus, fractional values between 0 and 1 were expected. Quite often, however, values of I greater than 1 were calculated, indicating an increased number of viable particles over what was expected. The most logical explanation of this phenomenon offered to date is that the initial assay for microorganisms in the wastewater was an underestimate of the true number of viable microorganisms present. This could result from either inadequacies in assay procedures or from the tendency of microorganisms to occur in wastewater as aggregates. The forces active at the sprinkler head and during aerosolization could break apart these clumps and cause a net increase in the number of viable particles assayed. Median values of I above 1 would result for those groups of microorganisms generally considered resistant to the lethal effects of aerosolization (fecal streptococci, Clostridium perfringens, mycobacteria and Pseudomonas). Conversely, the less hardy groups of microorganisms (total coliform, fecal coliform, coliphage and standard bacterial plate count) all had I values less than 1; indicating a greater influence of biological dieoff in the overall calculation of the microbiological impact factor. A second limitation is the inadequacy of coliform bacteria, standard bacterial plate count or coliphage to serve as representative indicators of the pathogen content of wastewater aerosols. Camann, et al., (21), have shown all of these organisms to be relatively fragile in the aerosol state relative to a number of wastewater-borne pathogenic bacteria. Thus, use of these organisms as indicators will consistently under-estimate the pathogen content of the atmosphere in the vicinity of a sprinkler irrigation system.

CONCLUSION:

The forgoing has reviewed some of the requirements and limitations associated with microorganism aerosol monitoring programs at wastewater sprinkler irrigation systems. The incentive to conduct such studies has been a concern that the distribution of pathogenic microorganisms in the atmosphere may be associated with an increased probability of human infection. Of concern is the health status of employees at the wastewater land treatment site and their families and of individuals who work or reside in the vicinity of the site. Unfortunately, correlations have yet to be developed to define aerosolized pathogen concentrations that significantly increase the probability of infection compared to other environmental sources of the same disease organisms. Therefore, the individual considering the establishment of a microbiological aerosol monitoring program is forewarned that a yardstick does not yet exist to permit conclusions regarding the actual human health hazard that is associated with a measured aerosol concentration.

DISCLAIMER

The findings in this report are not to be construed as an official Department of the Army position unless so designated by other authorized documents.

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DESIGN CRITERIA

DETERMINATION OF APPLICATION RATES AND SCHEDULES IN LAND TREATMENT SYSTEMS

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ABSTRACT

Three methods can be used to determine application rates in land treatment systems: field measurements; comparison with rates from similar existing projects; and field measurements combined with previous experience and judgment. In view of the present state of the art in land treatment technology, the third method is recommended.

In irrigation systems, application rates are determined using the water balance, the nitrogen balance, and measured or estimated percolation rates. In rapid infiltration systems, field measurements are necessary, coupled with a detailed knowledge of the subsurface hydrology and comparisons with existing systems. In overland flow systems, application rates depend mostly on wastewater treatment needs and are currently determined by comparison with existing treatment systems.

INTRODUCTION

The determination of application rates in land treatment systems is one of the most crucial and complex problems in planning and design. In planning studies, the relationship between soil permeability and the application rate (for infiltration systems) presented in the EPA Process Design Manual can be used [1]. In design, the recommended practice is to make field measurements (especially for rapid infiltration) and to temper the results with experience from similar projects. Considerations of wastewater quality and treatment levels required also influence the selection of application rates. In this paper, application rates, application schedules, and comparisons with existing systems are discussed for irrigation, rapid infiltration, and overland flow systems.

IRRIGATION

Application Rate

Typical application rates for irrigation or slow rate systems range from about 1.5 to 10 cm/wk. The choice depends on the climate, soil permeability, crop type and management practices, and required quality of the treated water. The climate and soil permeability are evaluated using the water balance; the required quality of the treated water is usually evaluated using the nitrogen balance. When crop production is the overriding concern, the irrigation requirements of the crop may limit the application rate.

Hydraulic Loading Rate. In the water balance, the inputs are wastewater application and precipitation and the outputs are evapotranspiration and percolation. The precipitation and evapotranspiration values should be determined for a design year in which wetter-than-normal conditions occur. The percolation rate can be either estimated from the soil permeability or measured in the field. On the basis of experience with wastewater, the long-term percolation rate can safely range from 4 to 10% of the permeability of the most limiting layer in the soil profile. Recommended field measurement techniques are basin flooding or the sprinkler infiltrometer [1].

Nitrogen Loading Rate. Because of the ability of nitrate nitrogen to move with the percolating water, nitrogen is often the limiting water quality parameter in infiltration systems. In slow rate systems, the input nitrogen is balanced against crop uptake, denitrification, and the nitrogen that percolates through the root zone. The climate can also influence the nitrogen balance. For example, in a humid climate, the water from precipitation (in excess of evapotranspiration) can dilute the percolating nitrogen concentration, as shown in the following comparison (Table 1) for a percolate containing a maximum of 10 mg/L nitrogen. Thus, in a humid climate, 36% more nitrogen can be applied than in the arid climate.

Table 1. Comparison of Nitrogen Loadings in Humid and Arid Climates [1]

Parameter	Humid climate	Arid climate
Applied nitrogen, mg/L	25	25
Precipitation minus evapotranspiration, m/yr	0.5	-0.5
Crop uptake, kg/ha	336	336
Denitrification, % of applied nitrogen	20	20
Hydraulic loading, m/yr	4.0	2.9
Wastewater nitrogen loading, kg/ha	980	720

Application Schedule

The application schedule for irrigation depends on the soil permeability, type of crop, application technique, and the climate. Operator convenience should also be considered. For permeable soils, the schedule should be to irrigate once a week or more frequently. For heavier soils using surface irrigation, the schedule may be to irrigate 1 day and rest for 4 to 6 weeks.

At Penn State, the schedule for the solid-set sprinkler system was to irrigate for 8 hours and rest for 6 days [2]. When 24 h/d operation is used, it may be more convenient to either automate the system or operate for 12 hours per setting. Each application schedule requires dividing the total field area into plots or settings for rotation.

The scheduling of application does not seem to materially affect the percolate quality of slow rate systems. At Hanover, New Hampshire, for example, the scheduling was varied from 8 hours once a week, to 24 hours once a week, to 8 hours three times a week without noticeable differences in percolate quality [3].

Comparison With Existing Systems

Wallace compared the infiltration rates with the actual application rates at three slow rate systems, and none of the systems was hydraulically limited [1]. For example, on the basis of soil permeability, the system at Muskegon, Michigan, could have been designed hydraulically for moderately high rate infiltration. In many slow rate systems, the application rate depends more on crop, groundwater, and nitrogen considerations than on hydraulic loadings.

RAPID INFILTRATION

Application Rate

In rapid infiltration, evaporation and vegetation are relatively unimportant but the soil permeability is critical. It is therefore important to concentrate the planning and design on determining the optimal infiltration rate to ensure that the system will work hydraulically while providing the necessary wastewater treatment.

Hydraulic Loading Rate. Although the soil permeability can be related to the infiltration rate, as previously described, it is recommended that the soil profile be evaluated and that field measurements of infiltration rates be conducted. The preferred method of determining the infiltration rate depends on the nature of the soil profile. If the profile is generally homogeneous, a surface flooding basin 2 m or more in diameter can be used. The basin is filled with clean water until the soil is saturated, and then the rate of infiltration is measured. Clean water is generally used unless the actual wastewater is available.

When the hydraulically limiting layers are deep within the soil profile, the various methods proposed by Bouwer can be used [4]. The gradient intake or double tube methods can be used to measure the permeability at the bottom of an auger hole.

Treatment Performance. In rapid infiltration systems, the required treatment performance is of primary importance in determining the application rate. Lance has shown that decreasing the application rate from the hydraulic limit can result in increased removals of constituents, especially nitrogen [5]. Because the chief mechanism of nitrogen removal in rapid infiltration systems is denitrification, the requirements of biological denitrification must be met:
(1) adequate detention time,
(2) anaerobic conditions (or at least anaerobic micro-sites), and (3) adequate carbon to drive the reaction. The reduction in application rate increases detention time and increases the potential for denitrification.

At Hollister, California, it was observed that high BOD to nitrogen ratios (5.5 to 1) in the applied wastewater and a moderate infiltration rate (about 15 m/yr) produced a nitrogen removal 93% [6]. If nitrogen removal is the limiting criterion in treatment performance, the loading rate, ratio of BOD to nitrogen in the wastewater, and the application schedule deserve special attention.

Application Schedule

Treatment efficiency in rapid infiltration systems responds to variations in application cycle. Short, frequent application schedules such as 0.5 to 3 days on and 1 to 5 days off, maximize nitrification but minimize nitrogen removal. As the drying time during the cycle increases, the potential for nitrogen removal increases. For example, the application schedule at Hollister, California, is 1 to 2 days on and 14 to 21 days off [6]. At Phoenix, Arizona, maximum nitrogen removal occurred when 10 days on was followed by 10 to 20 days off.

Comparison With Existing Systems

Because so few rapid infiltration systems have been designed for wastewater treatment and only a few of the others have been monitored, it is necessary to compare any proposed design with a few existing systems. The existing systems include those at Hollister, California [6]; Phoenix, Arizona [7]; Lake George, New York [8]; and Boulder, Colorado (pilot study).

At Hollister, California, the long-term application rate after 30 years averaged 4.2 cm/d [6]. Tests were made of the infiltration rates with both clear water and primary effluent using cylinder infiltrometers. With clear water, the infiltration rate averaged 85 cm/d while with primary effluent the rate was 98 cm/d (no statistical significance to the difference in rates). The long-term operating rate is therefore about 4 to 5% of the measured infiltration rate. Other ratios of long-term operating rates to measured infiltration rates are presented in Table 2.

Table 2. Ratios of Operating Rates of Application to Measured Infiltration Rates for Rapid Infiltration Systems [1]

Soil texture class	Type of wastewater	Ratio, % operating rate to infiltration rate
Sand	Oily cooling water	19.0-31.0
Gravelly sand	Secondary kraft mill	0.5-1.0
Loamy sand	Secondary kraft mill	2.0-12.7
Sand gravel	Secondary municipal	4.6-6.2

OVERLAND FLOW

Application Rate

Overland flow is a form of a fixed-film biological reactor. The application rate should therefore be predictable on the basis of the kinetics of treatment and the required treatment levels. For example, high-quality effluent has been produced in research projects using these application rates: raw wastewater, 10 cm/wk; primary effluent, 15 to 20 cm/wk; and secondary effluent, 25 to 40 cm/wk [9]. With all three application rates, the runoff contained less than 10 mg/L BOD on the average. Slopes were 2 to 4% and 36 m long.

Smith is studying the kinetics of overland flow treatment in an attempt to predict the necessary detention time and application rate that will provide a specified level of treatment [10]. This type of research and full documentation of existing research and demonstration findings are needed for a more complete understanding of the overland flow process.

For the present state of the art, application rates are determined from comparisons with existing systems and research projects. For raw wastewater, 7.5 to 10 cm/wk should be considered. For primary effluent, 10 to 20 cm/wk could be used, depending on the level of overall treatment required. For secondary effluent, either from ponds or conventional secondary facilities, the need is either for polishing (further reductions for BOD and suspended solids) or for nutrient removal. For polishing, 20 to 40 cm/wk could be considered. For nutrient removal, only 10 to 20 cm/wk may be possible because adequate detention time is necessary for denitrification.

Application Schedule

Experience with existing systems has led to application schedules of 6 to 8 hours on and 16 to 18 hours off over 5 to 6 d/wk. At Melbourne, Australia, however, the application has been continuous rather than intermittent. The optimum cycle will depend on the climate and the BOD loading. Other considerations include the harvesting of the grass and the potential for propagation of insects.

Comparison With Existing Systems

Hydraulic loading rates for overland flow have been increased, in research projects, as more experience is gained with the process. For example, Thomas began treating raw, comminuted wastewater in 1971 at rates of 7.4, 8.6, and 9.8 cm/wk. Hydraulic loading rates for the more recent research projects are summarized in Table 3.

Full-scale industrial systems at Paris, Texas, and at Davis and Sebastopol, California, have been of value in determining successful application rates and cycles at new facilities. Research and demonstration at Ada, Oklahoma, has been most instrumental in determining loading rates for various types of municipal wastewater [11]. Research and demonstration projects at Utica, Mississippi [12], and Hanover, New Hampshire [13], have added climatic, wastewater, and management variations to the body of knowledge on overland flow technology.

Table 3. Selected Hydraulic Loading Rates for Overland Flow Research Projects [1, 12, 13]

	Type of effluent applied	Hydraulic loading rates, cm/wk	Degree of slope, %	Slope length, m
Ada, Oklahoma	Raw comminuted	10-20	2-4	36
Ada, Oklahoma	Trickling filter	25-40	2-4	36
Pauls Valley, Oklahoma	Oxidation pond	25	2-3	45
Utica, Mississippi	Oxidation pond	6-12	2-8	45
Hanover, New Hampshire	Primary and secondary	5	5	21

CONCLUSIONS

It should be possible to determine application rates for land treatment systems on a case-by-case basis using data on soil infiltration rates, climate, wastewater characteristics, and required treatment performance. Unfortunately, neither the relationship between the infiltration rate and loading rate nor the relationship between the loading rate and treatment performance have been developed to the point where they can be used in design without requiring comparisons with other land treatment systems. Research and development in this area is promising; but more data relating design loadings to field conditions and treatment performance are needed.

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DESIGN CRITERIA

UPTAKE OF NUTRIENTS BY PLANTS IRRIGATED WITH MUNICIPAL WASTEWATER EFFLUENT

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We present comparisons of plant nutrient uptake by corn and forage grasses when these crops were irrigated with secondary municipal wastewater effluent or treated with inorganic fertilizer. Characteristic analyses of effluent from various locations are given for the macro plant nutrients as well as for quality indicators. The importance of the presence of varying amounts of N, P, and K in effluent studies is discussed. Micro elements in effluent are considered for their use to meet nutrient requirements of these crops as well as for their potential for environmental contamination. Total dry matter produced when combined with the content of an element in the harvested part of the crop directly reflects the quantity of an element removed. We graphically present total seasonal N uptake by corn irrigated with effluent in Minnesota, including comparisons with data from Pennsylvania and Florida experiments. Similar N uptake curves are plotted for reed canarygrass and other selected forage grasses grown at four locations. Nitrogen uptake with time within seasons by reed canarygrass during the normal first cutting period is graphically illustrated. Grass from control and high effluent treatments accumulated N linearly over the entire period, but N uptake was curvilinear when we used a low effluent rate. Tables of data including crop yields, amount applied, and uptake for P, K, Ca, Mg, and Na are presented and discussed.

INTRODUCTION

The renovation of treated municipal wastewater (secondary effluent) by irrigation of plants growing in soil has been adequately demonstrated. The goal is for optimum agricultural benefits as well as providing most efficient renovation of effluent without adverse environmental effects. The major advantage of applying effluent to growing crops is the possibility of recycling part of the plant nutrients. Secondary effluent contains, on the average, about 20 mg/l nitrogen, 10 mg/l phosphorus, and 15 mg/l potassium (Pound and Crites, 1973). To equal or surpass normal plant requirements, an application of 250 ha-cm/yr of effluent would provide 500 kg N/ha, 250 kg P/ha, and 375 kg K/ha.

A conference proceedings edited by Sopper and Kardos (1973a) covers land application of wastewater to agronomic and forest crops through 1972. They emphasize the research at Pennsylvania State University which involved studies of plant nutrient uptake and balances, in addition to soil responses and water quality changes. Bouwer and Chaney (1974) reviewed treatment of wastewater on land including fate of wastewater constituents in soil, crop response and system designs. The Cornell Agricultural Waste Management Conference Proceedings of 1976 and 1977 (Loehr, 1977a and 1977b) present several papers concerned with plant uptake of nutrients contained in wastewater effluent.

Our objective in this paper is to provide a comparison of total uptake of nutrients by corn and perennial forages irrigated with secondary municipal wastewater effluent.

EFFLUENT COMPOSITION

Secondary municipal wastewater effluent characteristics (Table 1) used in experiments compared herein show a similarity in most constituents. Where variations exist for a given location from the 'typical' values of Pound and Crites (1973), some unusual situation of treatment plant modification is usually given as explanation. Only the macro nutrients, N, P, K, Ca, and Mg are discussed in detail in this paper. Sulphur data is usually not reported in the literature probably because of analytical difficulty. Sodium and calculated sodium absorption ratio (SAR) values are shown since some effluents have high Na contents which may lead to problems in soil structure and permeability or in competition with K for exchange and uptake. Electrical conductivity (EC) and pH values when available are also compared as indicators of effluent quality. The micro nutrients including Fe, B, Mn, Mo, Cl, Zn, and Cu are present in effluent at very low concentrations and usually do not show signifi-

Table 1. Characteristic analyses of secondary municipal wastewater effluent from various locations.

	Source				
Constituent	Minn*	Penn†	NH‡	Mass§	Alberta¶
	mg/l				
Nitrogen, total	20.8	23.7	25.2	17.9	---
NH_4-N	16.5	12.9	18.0	7.5	13.3
NO_3-N	1.5	5.0	5.2	9.2	1.4
Phosphorus, total	8.4	---	6.3	8.5	3.9
PO_4-P	7.6	6.0	5.8	7.7	---
Potassium	12	16	11	9	34
Calcium	71	25	13	8	138¶
Magnesium	24	13	3	4	---
Sodium	274	28	45	38	147
SAR	7.0	1.1	3.0	2.8	3.1
	µmhos/cm				
EC	1940	---	400	---	---
pH	8.2	7.8	7.2	---	---

*Apple Valley, MN; USDA-SEA-FR, and University of Minnesota, St. Paul, 1974-77. (Larson, 1974-77).

†State College, PA: Pennsylvania State University, 1967-71. (Sopper and Kardos, 1973b; Hook and Kardos, 1977).

‡CRREL, Hanover, NH; U.S. Army Corps of Engineers, 1974-76. (Palazzo and McKim, 1978).

§Otis Air Force Base, MA: Woods Hole Oceanographic Institution, 1975-76. (Deese et al., 1977).

¶Taber, Alberta, Canada; Agriculture Canada, Lethbridge, 1972-75. (Bole and Bell, 1978). Ca concentration = Ca + Mg.

cant increases in plant uptake when compared with fertilized control treatments.

Nitrogen

Emphasis in effluent studies is placed on N because its transformations in the soil-plant system encompass the whole N cycle. The main consideration from the standpoint of N content of effluent is the relationship between crop uptake and quantity applied. Forms of N are important, with the effluents compared here (Table 1) giving ratios of NH_4-N:NO_3-N:organic N ranging from 79:7:14 (Minnesota) to 42:51:7 (Massachusetts). A high NH_4-N to NO_3-N ratio appears to be desirable from both the soil storage and water quality aspects. The NH_4-N form will allow exchangeable NH_4^+ to be stored on clay and organic matter sites as well as requiring microbial transformation to NO_3-N for either N uptake or leaching into ground water. The Pennsylvania project experienced problems with yield reductions for corn and high NO_3-N leaching when effluent-N changed to higher NO_3-N levels in 1971. A few attempts have been made to account for N in the soil-water-crop system by measuring characteristic N components in a field experiment (Sopper and Kardos, 1973; Clapp et al., 1977; Deese et al., 1977). Excessive N may have adverse effects such as reducing yield and quality of fruit crops, delaying maturity of cotton, lowering sugar content of sugar beets and starch content of potatoes, and causing lodging of grain crops (Bouwer and Chaney, 1974). However, feed quality of forage grasses and corn fodder was not reduced when these crops were grown on land irrigated with effluent supplying large amounts of N (Sopper and Kardos, 1973; Marten et al., 1978).

Phosphorus

In nutritional importance to crops, P ranks second only to N. Some soils require additions of P fertilizer to sustain plant growth. Effluents, when applied in sufficient amounts to satisfy N needs for plants, will often provide P in excess of crop uptake (Table 1). Most soils have the capability of sorbing large quantities of P and thus prevent serious losses of this key eutrophication nutrient into surface waters. The long-term effects of P application are usually associated with determining when the soil sorption capacity will be exceeded and thus shorten the life of a land treatment site.

Potassium

Potassium is present in significant concentrations in effluents (Table 1). Effluent application experiments have not been designed to study K interactions and uptake because it is not a serious water contaminant. Usually fertilizer K is applied in ample amounts, in addition to K in effluent, so as not to limit plant growth.

Calcium, Magnesium, Sodium

The nutrients Ca and Mg usually vary in concentration in effluents with respect to water source. Waters from high-lime soil areas contain higher concentrations of Ca and Mg (Minnesota and Alberta, Table 1), whereas, those from acid soil areas usually contain lesser amounts (New Hampshire and Massachusetts). Levels of Ca and Mg also may be affected by food processing or sludge treatments. Sodium, while not a nutrient, is contained in effluent in varying amounts, but appears to be highest where NaCl is used in water softeners to reduce water hardness due to Ca and Mg (Minnesota) or where soils are normally high in Na (western U.S.).

Micro Nutrients

Micro elements in effluent as nutrients and as potential environmental contaminants were studied by Sopper and Kardos (1973b) -- Mn, Fe, Cu, Zn, B, and Cl; Sidle et al. (1976) -- Zn and Cu; Deese et al. (1977) -- Mn, Fe, Zn, and Cu; Bole and Bell (1978) -- Mn, Fe, Zn, and Cu; and Dowdy et al. (1978) -- Mn, Fe, Zn, Cu, and B. The major concern has been to determine whether plant uptake of the metals would occur at phytotoxic levels or at levels harmful to the food chain. Sidle et al. (1976) concluded that uptake of heavy metals by reed canarygrass and corn silage from effluent application is not a hazard to the food chain except for Cu on reed canarygrass. These crops, however, received sewage sludge as well as effluent which increased the metals content considerably over effluent alone. The results of Dowdy et al. (1978) indicated that in spite of high effluent loading rates, trace metals content of corn and grasses, in general, did not change appreciably from fertilized control treatments. However, Cu concentrations in corn and grasses decreased almost to plant deficiency levels. Effluent additions also caused reduced Fe and Mn concentrations, but increased B levels. From limited studies of uptake of micro nutrients from effluent, one may conclude that removal of these elements is less than 10% of the annual application. These quantities should not pose a hazard to crop growth or to the food chain.

CROP YIELDS

When municipal effluents are applied to agricultural land, the primary objective becomes their use as a source of plant nutrients for maximum crop production. Similarly, the most effective method of removing elements is to harvest the plants that have taken up the elements, and remove them from the area. Crop yields as an indicator of effectiveness of an effluent irrigation system are influenced by many additional factors other than those directly related to the effluent, e.g., climate, soil physical and chemical properties, drainage, crop species, crop and soil management, and irrigation rates and schedule. Total dry matter produced (crop yield) when combined with the content of an element in the harvested part of the crop directly reflects the quantity of an element removed by a crop irrigation system.

PLANT COMPOSITION AND UPTAKE

For the macro nutrients, N, P, and K, elemental concentrations in harvested crop tissues have wide ranges. Means, minimum, and maximum levels for corn fodder (mature whole plants) are 1.4% N, 0.8 to 2.4; 0.3% P, 0.2 to 0.5; and 0.9% K, 0.8 to 1.1 (Miller, 1958). Perennial cool season grasses fall within the ranges of 0.5 to 4.5% N, 0.1 to 0.9% P, and 0.5 to 4.1% K. Other elemental concentrations for macro and micro nutrients for selected crops are cited by Miller (1958), Rickard (1972), and Kardos et al. (1977). Many of the mean nutrient values included in Miller's report are too low for adequately fertilized crops. It is misleading to use these data in calculating potential nutrient uptake based on average crop yields and effluent composition. Our Minnesota effluent experiments indicate more realistic concentration values of N, P, and K, respectively, for corn fodder of 1.2% N, 0.3% P, and 1.0% K; for reed canarygrass of 3.0% N, 0.4% P, and 2.7% K (Larson, 1974-77).

Total quantities of N removed by crop uptake under fertilizer systems may range from as little as 60 kg N/ha/yr for small grains to as much as 400 kg/ha/yr for reed canarygrass. The same respective crops will remove from 9 to 62 kg P/ha/yr and from 16 to 380 kg K/ha/yr (Fried and Broeshart, 1967). For other macro nutrients, Ca uptake can range from 10 kg/ha/yr for grasses to 170 kg/ha/yr for legumes and Mg uptake can vary from 8 kg/ha/yr for small grains to 25 kg/ha/yr for corn. Kardos et al. (1977) suggested that Na uptake is relatively low (< 5 kg/ha/yr) for forage crops fertilized with inorganic elements or with effluents containing low Na concentrations. However, Larson (1974-77) reported much higher Na uptake values (35 kg/ha/yr for corn and 60 kg/ha/yr for tall fescue) by plants irrigated with effluents having high Na concentrations. The micro nutrients Mn, Fe, Cu, Zn, and B are usually in microgram per gram quantities, giving crop uptake values of from 40 mg Cu/ha/yr to 400 g Mn/ha/yr (Allaway, 1968).

Nitrogen

Figure 1 describes total N uptake by corn irrigated with effluent for a 4-year period in Minnesota. Two effluent rates (∿ 5 and 10 cm/wk) with varying N concentrations produced N applications ranging from 150 to 630 kg/ha. Uptake of N varied from 110 to 145 kg/ha at the lower N applied levels (67% uptake) then appeared to reach a constant value of 180 kg/ha (30% uptake) as applications increased. Control blocks fertilized with NH_4NO_3 showed a mean of 300 kg N/ha applied with 175 kg N/ha uptake (58%) for 3 of the 4 years. In 1975, a combination of favorable conditions produced high yields and N concentrations to give an uptake of 75%. Calculated points from the Pennsylvania (Sopper and Kardos, 1973b; Hook and Kardos, 1977) and Florida (Overman and Nguy, 1975) studies are plotted for comparison. The Pennsylvania data were quite variable over the 1968-73 period, and included N applied and N uptake ranging from 259 and 46 kg/ha in 1968 to 176 and 249 kg/ha in 1969, respectively. By 1973, 233 kg N/ha applied gave only 67 kg N/ha uptake, as corn fodder yields decreased to 8.7 mt/ha. The authors suggested several possible reasons for these decreases, including weed competition, applied N not meeting peak demands, and N mineralization rates too slow to supply adequate N. We suggest the additional factor of effluent N changing from predominantly NH_4-N to NO_3-N in 1972, resulting in leaching of NO_3-N below the plant root zone. The Florida data were taken from a study designed to relate N uptake to N applied at several stages of corn growth. The means for two application rates, 5 and 20 cm/wk, were calculated from the authors' N uptake curves.

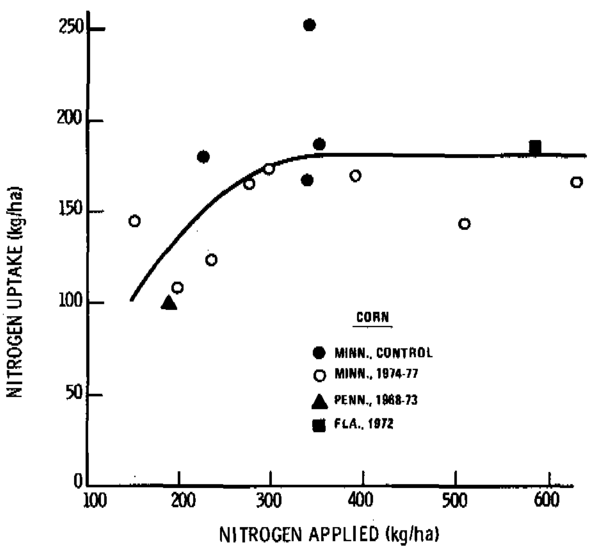

Figure 1. Total seasonal nitrogen uptake by corn irrigated with secondary municipal wastewater effluent.

Figure 2 compares total N uptake by reed canarygrass at four locations as a function of N applied. The Minnesota data cover a range in N applied from 235 to 800 kg N/ha and N uptake from 235 kg N/ha (100% uptake) to 420 kg N/ha (52% uptake). Control blocks fertilized with NH_4NO_3 have a mean of 308 kg N/ha applied with 296 kg N/ha uptake for the 4-year period (96%). The Pennsylvania values (Hook and Kardos, 1977) are grouped about a 4-year mean of 630 kg N/ha applied and 390 kg N/ha uptake (62%). The 4-year data from Alberta (Bole and Bell, 1978) are derived mainly from lower N applications when the N content in the effluent decreased by 50% between the first and second years. Applied and uptake N ranged from 52 and 59 kg/ha (113% uptake) to 410 and 237 kg/ha (58% uptake). The Massachusetts (Deese et al., 1977) data were derived from a figure of accumulative N applied plotted against accumulative N uptake. For three application rates during 1976, values for N applied and N uptake ranged from 300 and 190 kg/ha to 150 and 85 kg/ha with a constant uptake of 46%. Insufficient data were provided for further comparisons.

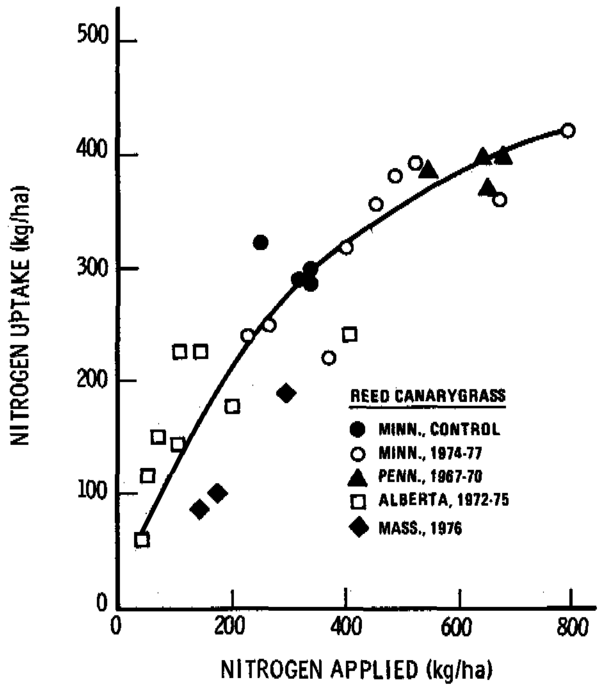

Figure 2. Total seasonal nitrogen uptake by reed canarygrass irrigated with secondary municipal wastewater effluent.

Figure 3 represents N uptake with time for reed canarygrass irrigated with effluent at 6 and 11 cm/wk during 1976 compared with fertilized controls. Samples were taken from the first cutting of each of the 4-time, 3-time, and 2-time cutting schedules (Larson, 1974-77). Sharply increasing yields combined with decreasing N contents give N uptake curves which are still increasing for the control and high effluent treatments, but leveling off for the low treatment. These data may provide valuable input information for testing computer models of effluent water quality.

Total N uptake by selected forage grasses other than reed canarygrass at three locations is compared with N applied (Figure 4). The uptake curve is similar to the one for reed canarygrass (Figure 2). The forage mixture in the New Hampshire study (Palazzo and McKim, 1978) received up to 1120 kg N/ha with an uptake of 450 kg/ha (40%).

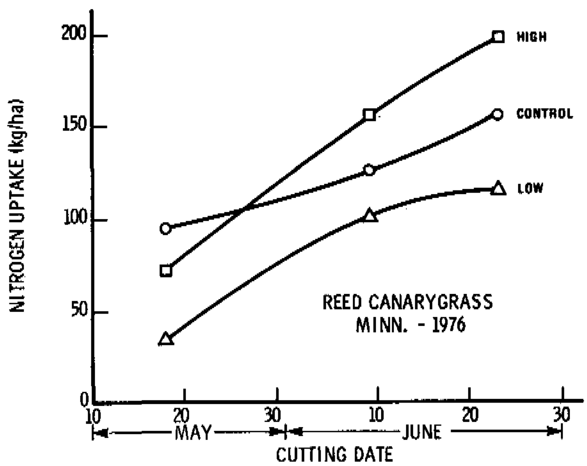

Figure 3. Nitrogen uptake with time in the first crop of reed canarygrass irrigated with secondary municipal wastewater effluent.

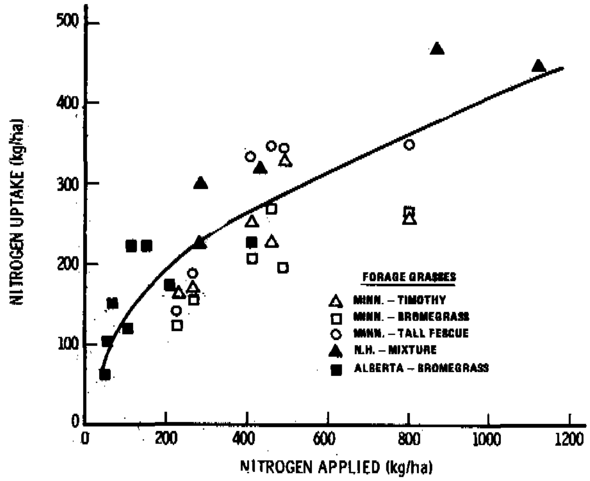

Figure 4. Total seasonal nitrogen uptake by selected forage grasses irrigated with secondary municipal wastewater effluent.

Phosphorus

Yields and P uptake for several crops at various locations as a function of effluent application rate are compared in Table 2. Data were obtained from the same experiments as for the N comparisons presented in Figures 1 to 4. Results for fertilized control treatments are reported whenever available (0.0 cm/wk). For most crops and locations, yields increased with increasing effluent application rates. Yields from controls were higher than or equal to those from the high rates of effluent. Uptake of P remained quite constant within species, however percentage uptake always decreased with increasing amounts of P.

Potassium

Uptake of K for several crops at various locations (Table 3) ranged from 108 kg/ha (68% uptake) on reed canarygrass at Massachusetts to 345 kg/ha (133% uptake) on the mixed forages at New Hampshire. Where lower amounts of K were applied (Pennsylvania and New Hampshire), K uptake exceeded 100%.

Calcium, Magnesium, Sodium

Uptake data summarized for Ca and Mg (Table 4) show values within the ranges reported by Fried and Broeshart (1967). These elements are difficult to study in an effluent irrigation experiment because liming and fertilizer treatments often confound separation of the nutrient source. Uptake of Na has been included because of the high concentrations in the effluents applied at Minnesota. The Na uptake by corn, reed canarygrass, and tall fescue (34, 38, and 58 kg/ha, respectively) are the highest values reported for municipal wastewater effluent experiments.

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Table 2. Crop dry matter yields and uptake of phosphorus for several crops at various locations as a function of application rate of secondary municipal wastewater effluent.

Crop	Location	Years	Rate	Yield	Applied	Uptake	Uptake
			cm/wk	mt/ha/yr	---kg/ha/yr----		%
Corn*	Penn	1970	0.0	10.7	--	24	--
	Penn	1969-70	2.5	13.4	22	34	154
	Penn	1968-71	5.0	9.3	46	29	63
	Penn	1972-73	7.5	7.7	64	34	53
	Minn	1974-77	0.0	16.0	76	39	51
	Minn	1974-77	5.1	12.7	104	35	34
	Minn	1974-77	9.4	13.4	190	37	19
Reed Canarygrass†	Penn	1967-70	5.0	12.4	162	56	35
	Minn	1974-77	0.0	11.1	76	38	50
	Minn	1974-77	7.0	9.7	138	42	31
	Minn	1974-77	12.6	12.2	248	54	22
	Alberta	1972-75	3.0	5.8	27	26	96
	Alberta	1972-75	6.0	9.4	54	36	67
	Mass	1976	5.0	6.0	150	24	16
Smooth Bromegrass‡	Alberta	1972-75	3.0	6.0	27	23	85
	Alberta	1972-75	6.0	10.0	54	35	65
	Minn	1974-75	0.0	9.6	152	44	29
	Minn	1974-75	6.5	7.4	158	34	22
	Minn	1974-75	11.4	7.9	277	43	16
Timothy§	Minn	1974-75	0.0	8.8	152	34	22
	Minn	1974-75	6.5	7.6	158	32	20
	Minn	1974-75	11.4	8.6	277	40	14
Tall Fescue¶	Minn	1974-76	0.0	11.6	101	40	40
	Minn	1974-76	6.3	10.9	151	43	28
	Minn	1974-76	11.3	12.0	268	49	18
Mixed Forages**	NH	1974-75	5.0	10.2	94	32	34
	NH	1974-75	15.0	13.3	262	44	17

*<u>Zea mays</u> L. (Varieties: Penn - 'Pa.890-S' and 'Pa.602-A'; Minn - 'Minhybrid 4201', 'Pioneer 3780', 'Northrup King PX-488' and 'PX-476').

†<u>Phalaris arundinacea</u> L. (Varieties: Penn - not given; Minn - 'Rise', and "NCR-C1'; Mass - not given).

‡<u>Bromus inermis</u> Leyss. (Varities: Alberta - not given; Minn - 'Fox').

§<u>Phleum pratense</u> L. (Variety: Minn - 'Climax').

¶ <u>Festuca arundinacea</u> Schreb. (Variety: Minn - 'Kentucky 31').

**<u>Phalaris arundinacea</u> L., <u>Phleum pratense</u> L. var. 'Climax', <u>Bromus inermis</u> Leyss. var 'Lincoln'.

Table 3. Crop dry matter yields and uptake of potassium for several crops at various locations as a function of application rate of secondary municipal wastewater effluent.

Crop*	Location	Years	Rate	Yield	Applied	Uptake	Uptake
			cm/wk	mt/ha/yr	---kg/ha/yr---		%
Corn	Penn	1970	0.0	10.7	—	110	--
	Penn	1969-70	2.5	13.4	57	144	251
	Penn	1969-70	5.0	14.2	111	122	110
	Minn	1974-77	0.0	16.0	462	162	35
	Minn	1974-77	5.1	12.7	459	119	26
	Minn	1974-77	9.4	13.4	532	131	25
Reed Canarygrass	Penn	1967-70	5.0	12.4	304	292	96
	Minn	1974-77	0.0	11.1	462	306	66
	Minn	1974-77	7.0	9.7	516	260	50
	Minn	1974-77	12.6	12.2	628	335	53
	Mass	1976	5.0	6.0	159	108	68
Smooth Bromegrass	Minn	1974-75	0.0	9.6	421	284	67
	Minn	1974-75	6.5	7.4	412	214	52
	Minn	1974-75	11.4	7.9	443	230	52
Timothy	Minn	1974-75	0.0	8.8	421	257	61
	Minn	1974-75	6.5	7.6	412	225	55
	Minn	1974-75	11.4	8.6	443	250	56
Tall Fescue	Minn	1974-76	0.0	11.6	505	318	63
	Minn	1974-76	6.3	10.9	485	269	55
	Minn	1974-76	11.3	12.0	557	306	55
Mixed Forages	NH	1974-75	5.0	10.2	127	236	186
	NH	1974-75	15.0	13.3	260	345	133

*See footnotes to Table 2.

Table 4. Crop dry matter yields and uptake of calcium, magnesium, and sodium for several crops at various locations as a function of application rate of secondary municipal wastewater effluent.

Crop*	Location	Years	Rate	Yield	Uptake Ca	Mg	Na
			cm/wk	mt/ha/yr	-----kg/ha/yr-----		
Corn	Penn	1970	0.0	10.7	47	17	0.1
	Penn	1969-70	2.5	13.4	34	26	2
	Penn	1969-70	5.0	14.2	24	23	2
	Minn	1974-76	0.0	16.0	37	37	9
	Minn	1974-76	5.0	13.1	25	28	25
	Minn	1974-76	9.0	14.0	26	24	34
Reed Canarygrass	Penn	1967-70	5.0	12.4	55	40	4
	Minn	1974-77	0.0	11.1	38	40	6
	Minn	1974-77	7.0	9.7	37	32	24
	Minn	1974-77	12.6	12.2	51	38	38
	Mass	1976	5.0	6.0	10	14	2
Smooth Bromegrass	Minn	1974-75	0.0	9.6	34	26	5
	Minn	1974-75	6.5	7.4	32	18	18
	Minn	1974-75	11.4	7.9	35	22	20
Timothy	Minn	1974-75	0.0	8.8	35	24	5
	Minn	1974-75	6.5	7.6	32	18	16
	Minn	1974-75	11.4	8.6	41	22	24
Tall Fescue	Minn	1974-76	0.0	11.6	51	48	15
	Minn	1974-76	6.3	10.9	43	32	37
	Minn	1974-76	11.3	12.0	50	34	58

*See footnotes to Table 2.

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DESIGN CRITERIA

STORAGE CAPACITY AND LOADING RATES FOR NITROGEN AND PHOSPHORUS

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ABSTRACT

Removal of N and P in harvested crops and capacities for storage of these elements in the soil are discussed as important factors in determining loading rates for wastewater on lands. Storage mechanisms for N are irreversible trapment of NH_4^+ in clays, accumulation of NH_4^+ as an adsorbed but exchangeable ion and accumulation in the organic form. Loading rates for N should be adjusted to storage capacities, removal in harvested crops, and the amount of N lost by volatilization as NH_3 and as N_2O and N_2 from denitrification. Storage of P is by reaction with inorganic constituents in the soil. Sorption of P on surfaces is used as the dominant soil reaction and sorption concepts are used to develop an empirical model for predicting the capacity of soils to store the P added in excess of removal in crops for a low rate system.

INTRODUCTION

Of the 16 elements recognized as essential for the growth of plants, nitrogen (N) and phosphorus (P) are the two that are most frequently deficient in soils and are the dominant elements in inorganic commercial fertilizers. These two elements are also of significance in water quality problems. Both are essential for biological productivity in waters and are thus involved in problems of unwanted or excess biological growths.

When wastewaters are used for irrigation the N and P they contain can meet the needs of crops or greatly exceed these needs depending on concentrations of N and P, amounts of wastewater used, crop needs, the nature of the soil and the management system. When crop needs and the capacity of the soil system to store these elements in unavailable forms are exceeded, their escape into surface and groundwaters create undesirable environmental effects. Thus, removal of N and P in harvested crops and/or their inactivation in or losses from the soil should be considered in establishing loading rates.

Nitrogen and P behaviors in soils are so distinctly different that each must be treated separately. Detailed discussions of soil reactions and removal in harvested plants of each element relative to the use of wastewaters on lands were presented in recent reviews by Broadbent (1977) and Pratt (1977).

NITROGEN

Storage

In a steady-state situation over a period of years or decades where additions to and removals of N from the soil system are equal, the storage capacity of the soil is not important even though residence time in the soil may be quite long. In actual wastewater application practice, particularly during the first few years of use of the wastewater with slow rate systems, storage can be important because steady-state or equilibrium conditions are not quickly attained. The main storage mechanisms are fixation of NH_4^+ by clay minerals and by organic matter, retention of NH_4^+ as an exchangeable cation and incorporation into soil organic matter through plant and microbial utilization.

Ammonium fixation. Soil clays, particularly those of the vermiculitic group have a capacity to trap NH_4^+ within the crystal lattice. The NH_4^+ ions fixed by this mechanism do not exchange readily with other cations and are not accessible to nitrifying bacteria (Nommik, 1965). And the quantities of N thus fixed will remain fixed indefinitely. The NH_4^+ added in most wastewaters will keep the fixation capacity by this mechanism saturated.

Quantities of NH_4^+ fixed by three different soils that received five consecutive applications of a solution containing NH_4^+-N at 100 mg/l, without intervening drying periods are shown in Figure 1. The Aiken soil containing mainly kaolinite clay fixed no NH_4^+. The Columbia fine sandy loam fixed about 22 mg/kg (308 kg/ha-m of soil). This soil and the Sacramento clay contain vermiculites and montmorillonites capable of NH_4^+ fixation.

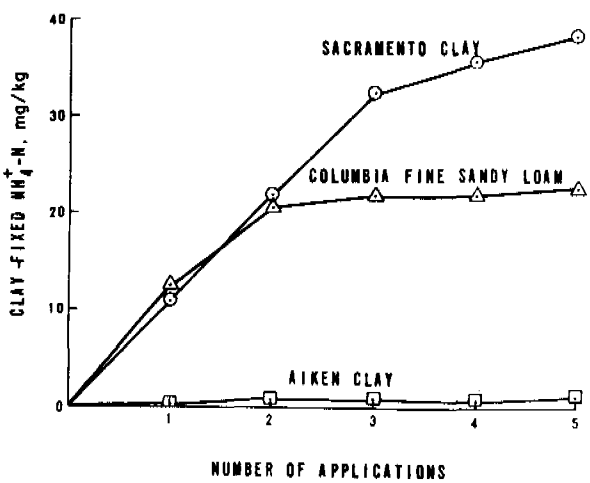

Fig. 1. Clay-fixed NH_4^+ in three soils resulting from five applications of a solution containing NH_4^+-N at 100 mg/l, without intervening drying (from F. E. Broadbent, Unpublished Data, University of California, Davis).

Another mechanism of N storage involves the reaction of NH_4^+ with soil organic matter to form stable organic matter complexes. But, because the amounts fixed depend strongly on pH and on the organic matter present (Burge and Broadbent, 1961), this mechanism is not likely to be important in soils of low organic matter contents and at the NH_4^+ and organic matter concentrations found in most wastewaters and at the pH of these waters.

Exchangeable ammonium. Like other cations in wastewaters, NH_4^+ can be adsorbed by the negatively charged clay and organic colloids in soils. Lance (1975) calculated the NH_4^+ adsorption from the NH_4^+ adsorption ratio (AAR) and the cation-exchange capacity (CEC) of the soil. The AAR is the ratio of NH_4^+ concentration to the square root of the concentration of Ca^{+2} plus Mg^{+2} when concentrations are in mmoles/l. In slow rate systems the NH_4^+ adsorption capacity is usually sufficient to retain the applied NH_4^+ in the surface of the soil. Continuous flooding in rapid infiltration systems will in time saturate the adsorption capacity and the NH_4^+ will move downward with the percolating water. Storage in the exchangeable form is temporary in any case because the NH_4^+ is nitrified as soon as oxygen (O_2) becomes available. But, exchangeable NH_4^+ plays an important role in the nitrification-denitrification sequence in high rate systems by holding N in the soil until the environment becomes oxic during drying.

Incorporation into organic matter. The amount of organic N stored in the organic matter tends to reach some equilibrium amount under a given set of conditions. This equilibrium level is attained when the rate of production of organic N equals the rate of destruction (mineralization). To store more organic N the rate of production must increase relative to the rate of mineralization. If the rate of production is increased the amount of organic N stored will increase rapidly for a few years and then asymtotically approach a new equilibrium as the rate of mineralization increases to balance the new rate of production.

Net immobilization of N by soil microorganisms requires the presence of decomposable organic matter having a N content less than about 1.25%. Except for cannery and certain industrial wastewaters this condition is not met for land treatment systems. Crop residues on land receiving wastewaters may result in a new microbial immobilization of a small amount of N, though probably not more than 45 to 65 kg/ha-yr.

The storage of N through plant uptake of NH_4^+ or NO_3^- and subsequent conversion of roots and other plant residues into soil humus can be large when the original soils have low organic matter contents. This storage is illustrated in Figure 2, using data from a cropped area near Bakersfield, California, where wastewater had been used for 36 years. Total N down to a depth of 1.5 m increased by 8,290 kg/ha when compared with an adjacent untreated area. This represents an average storage of 230 kg/ha-yr, although most of this storage would have occured during the first 10 to 15 years with only small annual amounts stored during the remaining time. Lesser quantities of N would be stored in soils of high organic matter contents and in some cases the application of wastewater can change the environmental conditions to favor net mineralization of organic N (Sopper and Kardos, 1973). Thus, the storage of N by incorporation into organic matter is highly dependent on the wastewater and the environmental conditions in the soil-water system.

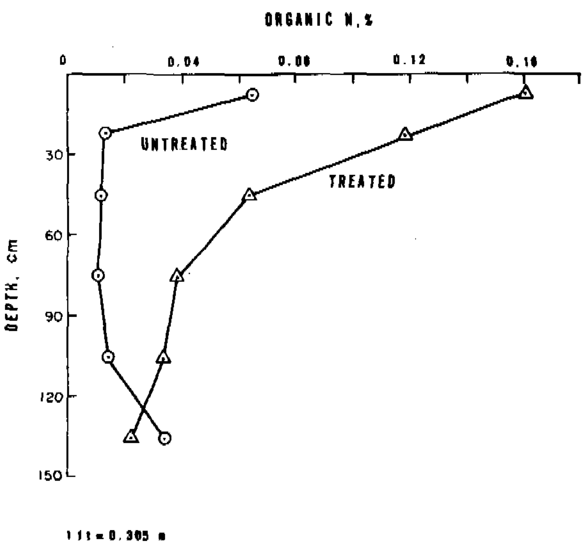

Fig. 2. Effect of 36 years of wastewater application on organic N in a soil at Bakersfield, California (F. E. Broadbent, Unpublished Data, University of California, Davis).

Loading Rates

The effectiveness of land treatment in removing N as the wastewater moves through the soil-plant system is determined by the annual loading rate minus the sum of annual 1) storage in soil, 2) removal in harvested crops, and 3) volatilization as NH_3 and as the gaseous products, N_2O and N_2, of denitrification. The objective is to minimize the difference between input and storage plus outputs other than leaching of NO_3^-.

Storage in the soil. An estimate of the quantity of N that can be stored by NH_4^+ fixation by clays can be obtained from laboratory studies using appropriate soil samples from the land area to be used. The number of samples and the depth of sampling will depend on the local conditions relative to soil variability and the length of the pathway for water moving through the soil and on financial resources.

Laboratory analyses can also be used to determine the capacity for temporary NH_4^+ storage in the CEC of the soil. The appropriate data are CEC in meq/100 g of soil, the AAR of the wastewater and the bulk density of the soil of the field. Of course the relationship between AAR and the percent saturation of the CEC must be known. Incorporation into organic matter must be estimated from experience and from knowledge of local conditions.

In slow rate systems storage can be important particularly in the first few years of a project. But, with time the storage capacity will be saturated. In rapid infiltration systems the temporary storage as adsorbed NH_4^+ is perhaps the only storage mechanism of importance.

After a few years of a given project, storage in the soil can probably be ignored and the steady-state model, in which annual inputs are equal to removal in crops, plus volatilization losses plus leaching losses, can be used.

Removal in harvested crops. Removal in harvested crops varies from a few kg/ha-yr for some fruit crops to hundreds of kg/ha-yr for some forage crops. Tucker and Hauck (1978) reported annual removals as low as 9 kg/ha for olives and as high as 361 kg/ha for corn silage and 438 kg/ha for hay (nonlegumes). Pratt et al (1976) reported a maximum removal of 468 kg/ha-yr from double cropping with sudangrass and barley. Nitrogen removals by bermudagrass as calculated from data of Fisher and Caldwell (1958) by Tucker and Hauck (1978) are presented in Figure 3. The maximum removal in kg/ha was at an input of about 1500 kg/ha, but the maximum percent removal was at an input of about 200 kg/ha. However, Fried et al (1976) reported that with corn the maximum yield was obtained at 224 kg N/ha and the maximum removal and maximum percent removal were at inputs of 336 and 224 kg/ha, respectively.

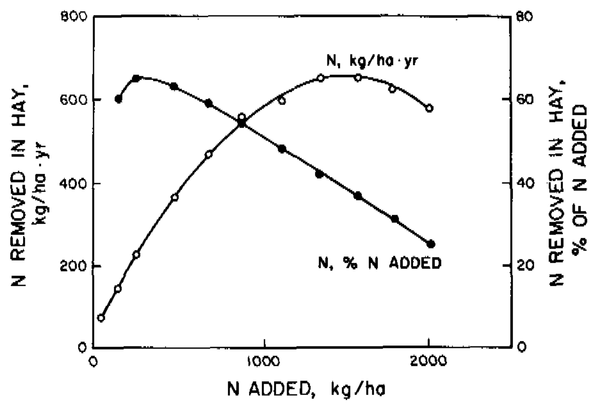

Fig. 3. Relationship of N removed in bermudagrass hay and the N added (Data of Fisher and Caldwell (1958) as presented by Tucker and Hauck (1978)).

Obviously, management decisions relative to removal in harvested crops must consider not only the capability of a crop to remove N but its suitability for the climate, the soil system being designed, and the potential financial returns. In slow rate systems the crop can be an important sink for N, but in rapid infiltration systems these removals can be so small compared to total inputs that crop removals can be ignored.

Volatilization. The conditions that favor volatilization of NH_3 and denitrification are sufficiently well known that projects can be planned to provide these conditions. The volatilization of NH_3 is favored by a high pH, soils of low buffering capacity, warm temperatures and high concentrations of NH_4^+-N at the surface of the soil. These effects are illustrated in Figure 4. The main effects are soil pH, the texture of the soil and the depth of placement of the $(NH_4)_2SO_4$ in the soil. Another factor is the water content, with wet soils favoring volatilization. Besides the selection of sandy soils, management can add agricultural limestone to make the soil calcareous to give a pH of 7.5 or higher.

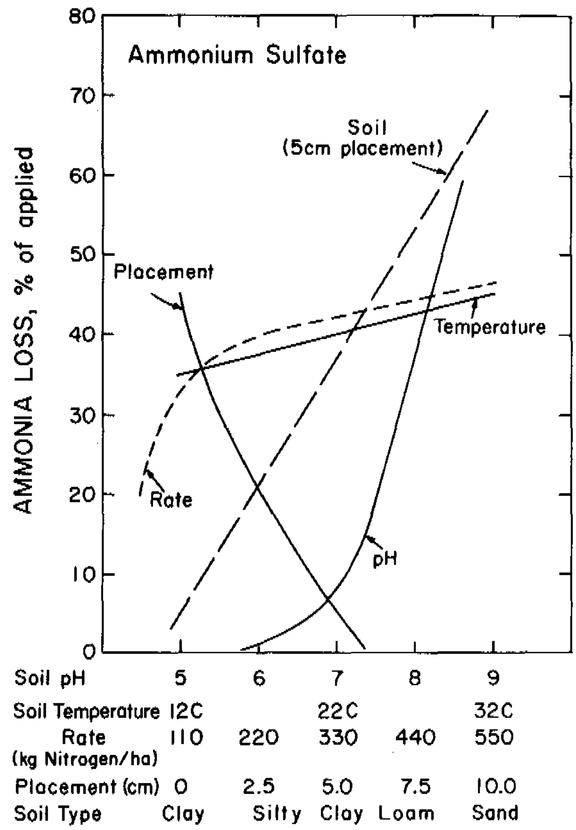

Fig. 4. Approximate relationships of NH_3 loss to soil pH, soil temperature, NH_4^+ application rate, depth of placement, and soil type for ammonium sulfate. The relationships are for NH_3 loss from a clay soil (excluding soil type curve), surface applications (excluding placement curve, 550 kg N/ha NH_4^+ application rate (excluding NH_4^+ rate curve), 30C (excluding temperature curve), and pH 7.6 (excluding pH curve). Taken from Rolston, 1978).

Denitrification is favored by anoxic conditions, a supply of readily available organic material, a pH between 6 and 8 and relatively high temperatures. Development of anoxic conditions are favored by saturation of the soil with water so that O_2 diffusion through the soil is restricted, by rapid consumption of O_2 by soil microorganisms and plant roots. Anoxic conditions develop not only in saturated soils but also in micropores of unsaturated but wet soils in which O_2 consumption is more rapid than O_2 supply. Growth of plants and application of organic materials such as manures, crop residues and other wastes favor denitrification by providing energy materials for microorganisms which in turn increases O_2 consumption. The nature of the soil has a large effect on denitrification with soils of low water transmissivity favoring development of anoxic conditions thus favoring denitrification.

Management can select soils, manage water and add organic materials to favor denitrification. Wetting and drying cycles have been developed to alternately provide nitrification-denitrification cycles to maximize N losses as N_2O and N_2. Examples of the manipulation of conditions for the removal of N from wastewaters using denitrification as the main process are the Flushing Meadows project at Phoenix, Arizona (Lance and Whisler, 1972; Bouwer et al, 1975; and Lance, 1975), the Santee project in California (Merrell et al, 1965), the Detroit Lakes project in Minnesota (Larson, 1960) and the Fort Devens project in Massachusetts (Satterwhite and Stewart, 1974).

PHOSPHORUS

Storage

The P that enters a soil is either removed physically, stored in the soil as part of the solid phase or is leached on through the soil in soluble forms in the drainage water. It can be physically removed in harvested crops or lost in wind or water erosion. However, we will assume that lands used for receiving wastewaters will be well protected from erosion, leaving removal in Harvested crops as the only physical removal process.

Concentrations of P in wastewaters usually range from 1.0 to 40 mg/l, but most concentrations are less than 20 mg/l. Thomas (1973) used 10 mg/l as a typical concentration. In contrast, the concentration of dissolved P in the soil solution is usually between 3.0 and 0.03 mg/l with typical concentrations of a few tenths mg/l. These differences in concentrations are reflective of the reactions which remove P from wastewaters when they are added to soils. The extent of P removal from solution depends on the intensity of these reactions, the capacity of the soil to maintain them and the time allowed for them to proceed.

The dominant reactions of P in soils involve inorganic orthophosphate ions. As organic forms of P in wastewaters are generally readily converted to inorganic forms, organic forms of P are expected to be relatively unimportant in P storage in the soils used for wastewater treatment, and will not be discussed.

The removal of P from solution can be described loosely as retention, a term that implies no distinct reaction mechanisms. The term sorption is used to describe the retention of P at surfaces as contracted with precipitation with inorganic species directly from solution. Explanations of the nature of P retention have oscillated from those implying the dominance of sorption reactions to those relying heavily on solubility and precipitation principles. An overview of the literature suggests that P retention involves mainly sorption at surfaces and that sorption provides the more meaningful approach to predicting the capacity of soils to store P. Direct precipitation of phosphates in soils appears to be fairly well limited to high P concentrations in the vicinity of fertilizer bands. Relative to the retention of P from wastewaters, precipitation of Ca phosphates can be expected in soils and/or wastewaters high in Ca and secondary precipitation of other phosphates might occur at solution P concentrations in the higher range of concentrations found in wastewaters. Under most circumstances, however, conditions favoring sorption reactions are expected for wastewater treatment systems.

Sorption surfaces. Many studies have pointed to the importance of secondary Fe and Al oxides and hydrous oxides, particularly those of amorphous character (Syers et al, 1971; Williams et al, 1958; Saunders, 1965). Data tabulated by Syers and Williams (1976) show that crystalline Fe and Al oxides sorb five to ten times more P than crystalline aluminosilicate clays or calcium carbonates and that amorphous oxide gels sorb five to 100 times more than their crystalline counterparts and approaching 1000 times more than crystalline aluminosilicates and calcium carbonates. Even at alkaline pH values characteristic of calcareous soils, the presence of amorphous hydrous oxides, particularly if present as coatings, may reduce to relatively minor importance the sorption of P by calcium carbonate as demonstrated by Shukla et al (1971) and Holford and Mattingly (1975).

Sorption equations. The sorption equation most frequently used for P in soils is that of Langmuir. A plot of sorption data using one of the linear forms of this equation permits the determination of a sorption maximum and a sorption energy. Although Langmuir (1918) originally developed this equation to describe the sorption of gases by solids, its extension to sorption from aqueous solutions has been justified in a theoretical discussion by Giles (1970).

Recent work by Ryden and Syers (1975a, 1977ab) and Ryden et al (1977 abc) and McLaughlin et al (1977) has shown that three Langmuir equations described P sorption up to a concentration of 20 mg/l. Each equation accounted for at least 80% of the sorption occurring over its operational solution concentration range. Chemisorption of P at hydrous oxide surfaces was indicated for solution P concentrations below about 0.2 mg/l. In contrast, above this concentration an increasing proportion of the sorbed P was described by the third equation, the parameters of which in conjunction with additional data, indicated a more-physical sorption reaction which was reversible when the P concentration in solution was decreased.

Conformity of P retention data to the Langmuir equation does not constitute irrefutable proof that a sorption reaction is involved (Brunauer et al, 1966; Veith and Sposito, 1977). However, additional data relating to charge, pH and ionic strength relationships and the ease of desorption and isotopic exchange support the concept that mainly sorption reactions are involved in P retention by soils in the solution P concentration range found in most wastewaters.

One of the consequences of the sorption process in which the quantity of P retained is a function of the solution P concentration, is that as the wastewater is added to the soil the P moves down in a diffuse boundary as shown in Figure 5. In contrast, if a compound of definite solubility were being formed a very much more abrupt boundary is expected. It should be noted that a diffuse boundary between enriched and unenriched soil horizons is generally observed in field studies (Pratt et al, 1956; Kardos and Hook, 1976) indicating the importance of sorption as a removal mechanism. The bottom of the diffuse boundary represents near zero saturation of the sorption capacity and the top of the diffuse layer represents near saturation at the top, with an infinite number of saturations in between.

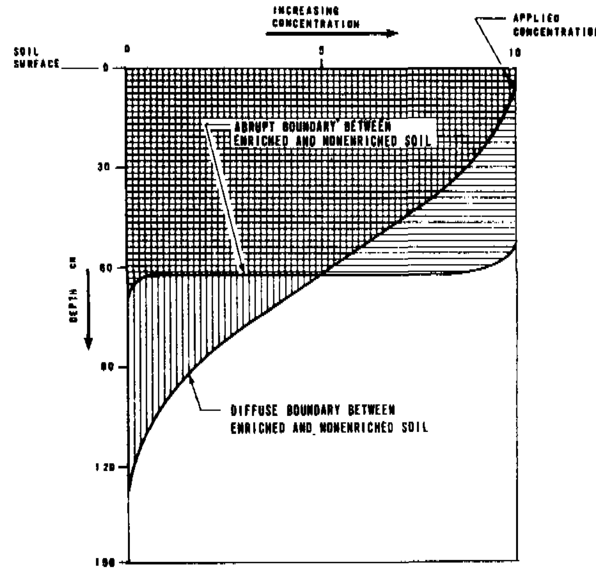

Fig. 5. Relationship between P concentration and depth for abrupt and diffuse boundaries between nonenriched and enriched soil.

Reactions rates. The reaction of P with soils is characterized by an initially fast reaction, followed by a slow reaction, which continues for months or years. If short periods of time are used for observations, an apparent equilibrium is attained in a few days, but, if longer periods are used, the soil appears to approach an equilibrium in 1 to 4 years depending mainly on the temperature.

This slow reaction might be explained as the result of decreased solubility of precipitated P compounds as they crystallize and stabilize. One possible explanation for alkaline conditions could be the conversion of Ca phosphates to less soluble compounds, i.e., the conversion of dicalcium to octacalcium to hydroxyapatate phosphates. The formation of discrete compounds from saturated surfaces could expose more reactive surfaces to continue the sorption process.

However, changes in the properties of sorbed P, such as decreases in the ease of desorption and isotopic exchange, are associated with this slow reaction. Ryden et al (1977b) and McLaughlin et al (1976) have shown that a shift from more-physically bound to *chemisorbed forms* and the regeneration of sorptive capacity are characteristic of the slow reaction. It was also demonstrated that the shift to chemisorbed forms was accompanied by a decrease in 0.1M NaOH-extractable P. The latter finding strongly suggested the absorption of initially adsorbed P. This concept, which involves the entry of surface sorbed P into the "structurally-porous" amorphous hydrous oxides (an absorption process) allows not only for the slow increase in sorption sites, but also for the decrease in the ease of isotopic exchange and desorption.

Barrow and Shaw (1974 and references contained therein) proposed a "three-compartment" (A-B-C) model based on the kinetics of P sorption. Compartment A represented the soil solution, compartment B a weakly sorbed form and C a strongly sorbed form. With increasing time, P was transferred from compartment B to C. The properties of compartments B and C enable them to be identified, respectively, with the more-physically sorbed P and chemisorbed P defined by Ryden et al (1977a).

For a wide range of south Australian soils the transfer of P from compartment B to C was described by equation (1)

$$(1 - \alpha) = (kt + 1)^{-b} \qquad (1)$$

where α is the fraction of sorbed P in compartment C, $(1 - \alpha)$ is the fraction of the sorbed P remaining in compartment B, t is the time of contact, b is a constant and k is a temperature coefficient as defined in equation (2)

$$k = A\exp^{-E/RT} \qquad (2)$$

where A is a constant, R is the gas constant, T is the absolute temperature and E is the activation energy.

Barrow and Shaw (1974), who used the term "relative effectiveness" to describe $1 - \alpha$, found that it was proportional to the added P which remained effective for plant growth and which remained desorbable or isotopically exchangeable. Data obtained by Ryden et al (1978) indicated that $1 - \alpha$ was also related to the shift of P to chemisorbed forms, and by inference the regeneration of sorption sites.

Equation (1) describing the relationship between "relative effectiveness" and time, indicates that as the period of contact between soil and added P increased the rate of transfer from compartment B to C decreased. An increase in temperature, however, considerably increases the rate of transfer. In contrast, the effect of water content was negligible except at low water contents (<10%). The fact that equation (1) was applicable to such a wide range of soils, indicated that the model provides a generalized description of the kinetics of P sorption. This independence of soil type, which was also observed in a study reported by Ryden et al (1978), is the most important feature of the model and suggests a basis for a widely applicable model for P sorption kinetics in wastewater treatment systems.

Loading rates

Loading rates depend on the rate of removal of P in harvested crops and the capacity of the soil to store P. The application of P in amounts greater than the sum of crop removal plus the capacity of a given depth of soil to retain P will result in P moving below that specified depth.

Removal in harvested crops. Amounts of P removed vary from about 10 kg/ha-yr for such crops as sugar beets and peanuts to about 70 to 80 kg/ha-yr or more for high yielding forage crops. Typical data are summarized in Table 1. These quantities of P are important in slow rate systems where annual rates of P application are the same order of magnitude as used in agronomic practice. For a solution concentration of 10 mg P/1, wastewater at a rate of 1 m-ha/ha-yr equals a P rate of 100 kg/ha-yr. Thus wastewater at rates of 1 to 2 m would give P rates somewhat comparable to agronomic practice. At these rates removal of 60 to 70 kg P/ha-yr would be significant whereas at 20 to 30 m-ha/ha-yr a P removal at these rates would be relatively insignificant.

Removal of P in harvested crops can be predicted or estimated from yields and P concentrations in the materials harvested. These types of data are easily available for most crops grown in a local area.

Storage in the soil. A satisfactory predictive model for P storage should consider mainly the capacities and kinetics of the P sorption process in soils and sediments from the land surface to the point of discharge into surface or groundwaters. Because this pathway is not well known a given depth into the soil profile and underlying strata is considered.

and temporal variability in the hydraulic conductivity are known to occur and these variabilities bring up the questions of how many and what kinds of samples or measurements are needed. After the data are obtained, problems arise in averaging and interpreting such large variations. Also, there have been no studies of the numbers of samples needed to characterize the P reaction characteristics of a field soil to a given depth.

Table 1. Removal of P annually by the usually harvest portion of selected crops.

Crop	Yield metric tons/ha	Phosphorus kg/ha
Corn (1)	12	35
Cotton (1)		
Lint and seed	1.7	19
Wheat (1)	5.0	22
Rice (1)	3.2	22
Soybeans (1)	4.0	25
Grapes (1)	11	11
Tomatoes (1)	36	34
Cabbage (1)	32	18
Oranges (1)	25	11
Small grain-corn-hay rotation (2)	-	33
Reed canary grass (2)	-	45
Corn silage (2)	-	31-41
Poplar trees (3)	-	26-70
Barley-sudangrass rotation for forages (4)	-	85-91
Johnson grass (1)	11	95
Guinea grass (1)	11	51
Tall fescue (1)	3.2	33

(1) Agricultural Waste Management Field Manual (1975); (2) Kardos and Hook (1976); (3) Kutera (1975); (4) Unpublished data of Pratt and Davis, University of California, and USDA-ARS, Riverside.

All models are based on a materials balance. That is, the P that goes into a given soil volume is either retained in the solid phase, removed by plants or moved through the soil volume in percolating water. This means that all models have a water flow and a P reaction component and of course if the system has a crop, plant removal is a third component for both water and P. Models can consist of simple bookkeeping or of mathematical equations of various degrees of sophistication.

A number of researchers, using various approaches have reported limited successes in modeling the reactions of P with soils (Enfield et al, 1976; Enfield and Shew, 1975; Novak et al, 1975; Harter and Foster, 1976).

Limitation of models. There are a number of problems that need to be solved before any predictive models can be used with confidence. Large spatial

The concentrations of Fe, Al, Ca and perhaps other constituents in the wastewater may influence P reactions. Waters that acidify or alkalinize the soil can also have large effects.

Perhaps the most serious limitation is that to date the reaction of P with the soil cannot be predicted from simple soil properties that can be mapped in the field or measured quickly in the laboratory (Enfield et al, 1976; Pratt et al, 1969).

Proposed model. In this simple model the P added minus that removed in harvested crops is assumed to react progressively with successive depth increments in the soil profile. The first depth becomes "saturated" before the P moves to the next. The boundary between the P-enriched and the non-enriched soil is assumed to be abrupt as in the theoretical model of Novak et al (1975). The term "saturated" is defined as the condition in which the soil is sufficiently enriched that the P movement with percolating water is significantly above background for the original soil material. This model also assumes water movement to be so unimportant compared to P reactions that it can be disregarded. Consequently, there will be sufficient time for the slow P reactions to have an impact. The P sorption capacities for the depth increments will include the slow reactions.

This model expressed mathematically is

$$T = \frac{Sp}{Ip - Hp} \tag{3}$$

where T is time in years for the P front to reach a given depth, Sp is the sorption capacity of the volume of soil above that depth expressed in kg/ha, Ip is the P input in kg/ha-yr and Hp is that removed in harvested crops in kg/ha-yr. The values for Ip and Hp can be determined with reasonable accuracy by well established techniques. The measurement of Sp is less well established. However, recent work (Barrow and Shaw, 1975; Holford and Mattingly, 1975; Ryden and Syers, 1977b; Ryden et al, 1977a), has suggested a relatively simple approach to determining Sp.

The term Sp in equation (3) is not the overall sorption maximum of the soil, but the sorption capacity at the average solution P concentration found in the wastewater to be applied. This sorption capacity is difficult to obtain directly but can be indirectly estimated. The overall sorption maximum of a soil can be obtained from the sorption isotherms suggested by Ryden and Syers (1977b) and from this value an estimate of Sp for any P solution concentration can be obtained.

By plotting the fractional saturation of the equilibrium P sorption maximum against solution P concentration, as is done in Figure 6 for four contrasting soils (Ryden et al, 1976) a generalized sorption curve is obtained. From this generalized curve the sorption at any solution P concentration can be calculated by multiplying the corresponding fractional sorption times the P sorption maximum which is the only parameter that varies among soils.

The overall equilibrium P sorption maximum can be determined based on the method of Ryden and Syers (1975b). In this method a quantity of soil is shaken with a solution containing P at a relatively wide solution to soil ratio which is valid for equilibrium studies (Hope and Syers, 1976). The solution P concentration is determined at several times (t) greater than 70 hours and graphed against 1/t. The resulting linear relationship is extrapolated to 1/t = 0, i.e. infinite time. The solution P concentration obtained at 1/t = 0 is used as an estimate of the equilibrium solution concentration for the amount of P added. From these two values the amount of P sorbed at 1/t = 0 can be calculated. This quantity of sorbed P must be adjusted by adding to it the amount of sorbed P initially present in the soil, i.e., the amount of P extracted by a 0.1M NaOH solution (Ryden et al, 1977b). The equilibrium sorption maximum (b) can then be calculated from

$$b = \frac{X}{\theta} \tag{4}$$

in which X is the experimentally determined sorbed P (adjusted for the sorbed P initially present) for the single P application used and θ is the fractional sorption saturation obtained from Figure 6 for the experimentally determined solution P concentration for 1/t = 0. From the calculated b value the amount of P sorbed at any equilibrium solution concentration can be estimated by using the appropriate θ value obtained from Figure 6.

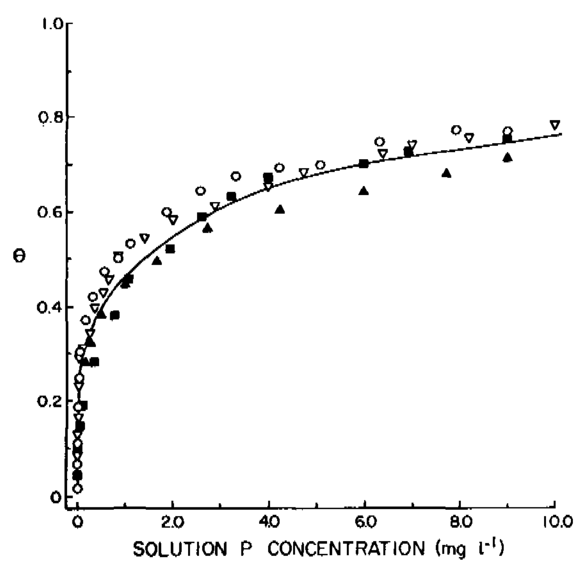

Fig. 6. Generalized equilibrium isotherm for P sorption by four

contrasting soils. $\theta = X/b$, where X is the amount of P sorbed at any given solution P concentration and b is the equilibrium sorption maximum. ● = Okaihau soil: ▼ = Egmont soil: ■ = Waikakahi soil: ▲ = Porirua soil. Data from Ryden et al (1977a).

The utility of the generalized P sorption isotherm (Figure 6) is illustrated by the interpretation of data presented in Table 2 for a Ramona soil, a soil not used in the original development of the generalized isotherm. The sorption maximum (b) calculated from equation (4) using θ values from the generalized sorption isotherm in Figure 6 was independent of the amount of P added. The fact that b was a constant suggests that P sorption by the Ramona soil was described by the same isotherm as the four soils used in developing the generalized isotherm.

Table 2. Equilibrium P sorption data for Ramona soil and related data calculated from the generalized P sorption isotherm in Figure 6.

Padded	Equilibrium P Concentration	Equilibrium P Sorption (x)[+]	θ[#]	Calculated sorption maximum (b)[&]
µg/g	mg/l	µg/g		µg/g
5	0.033	32.0	0.18	178
10	0.080	35.0	0.21	167
20	0.157	41.2	0.26	159
40	0.285	54.7	0.32	171
80	0.655	76.3	0.44	173
120	1.33	82.5	0.49	168
500	8.00	129	0.75	172
Average				170

+ Included 29 µg P/g native sorbed P.
# Obtained from the generalized isotherm (Figure 6).
& Calculated from equation (4).

If the Ramona soil were used to receive a wastewater having 5 mg P/l, a θ value of 0.68 will apply (Figure 6). From this value and the average sorption maximum (Table 2), the equilibrium sorption level at this solution P concentration can be calculated as 116 µg/g/0.68: 170 µg/g). However, the amount of native sorbed P must be subtracted to give an effective sorption of 87 µg/g. The data in Table 2 show that the sorption of P at only one addition of P plus one determination of the initially sorbed P would have been sufficient to calculate a net P sorption (Sp in equation (3)) for any given wastewater P concentration.

The uncertainties of this P balance-reaction model are the assumed abrupt boundary between enriched and non-enriched soil layers and the assumption that water movement is slow enough to allow complete sorption equilibrium. Furthermore, all of the general limitations of modeling also apply to this model.

There is ample evidence from soils that have received large amounts of P as fertilizers or wastes that soluble P decreases gradually as a function of depth with a diffuse front (Pratt et al, 1956; Spencer, 1957; Taylor and Kunishi, 1974; Kardos and Hook, 1976). Water movement in waste treatment systems will vary depending on water application rate, infiltration and transmissivity of the soil and evapotranspiration. Nevertheless, the model presented here might be useful in providing estimates of the storage of P in soils of wastewater treatment systems.

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DESIGN CRITERIA

MUSKEGON COUNTY, MICHIGAN'S OWN LAND WASTEWATER TREATMENT SYSTEM

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The Muskegon County Wastewater Management System is a lagoon-impoundment, spray irrigation treatment facility which services 13 municipalities and five major industries. The system consists of a 4,455 hectare site (11,000 acre) site which contains three aeration ponds, two storage lagoons of 344 hectares (850 acres) each and a total storage capacity of 19.3 million cubic meters (5.1 billion gallons), and 2,200 hectares (5,500 acres) of land irrigated by center-pivot irrigation rigs. The system is provided with a network of subsurface drains, open interception ditches and shallow wells to make possible the monitoring and control of the quality of water throughout the treatment process.

With an average daily flow of 102 thousand cubic meters (27 million gallons) from its first fully operational year in 1975 to the present, the system provided discharge water of a quality consistently above state NPDES specifications. Studies on water quality and soil-crop-nutrient balance revealed that by balancing the nutrients in wastewater with crop needs, more effective overall nutrient removal was achieved, simultaneously enhancing crop production and wastewater renovation. Revenues from crop sales are returned to the system to ameliorate treatment costs which amounted to between 6 and 7 cents per cubic meter (23-25 cents per 1000 gallons) from inception of full operation in 1975 through 1977.

Aspects of the system discussed include a description of components, their effectiveness in renovating wastewater, the impact on ground and surface water and costs.

DESIGN FACTORS

Geographic Location

Muskegon County is located on the eastern shore of Lake Michigan about 185 km (115 mi) north of its southern tip. The county is traversed by many streams which flow predominately westward and converge eventually into one of three inland lakes: Bear Lake, Mona Lake, and the largest, Muskegon Lake. The Wastewater Management Site is 24 km (15 miles) east of the Lake Michigan coastline and is drained by two stream systems, the Black Creek to the south and the Mosquito Creek to the north. Treated wastewater which is channeled into these creeks flows to Mona and Muskegon Lakes and finally to Lake Michigan.

Climate

The climate of the area is tempered by the position of the county relative to Lake Michigan. Precipitation, including snow, averages 76.2 cm/yr (30 in/yr) and is considered moderate; of the 218 cm/yr (86 in/yr) of snow, most falls during December and January. The mean average annual temperature is 8.4° C (47.2° F) with the average maximum in July at 26.8° C (80.3° F) and the average minimum in January at -8.0° C (17.6° F). The average frost-free period ranges from 160 to 170 days.

Soils

Many of the soils in the area are relatively infertile, unproductive, sandy, highly permeable and droughty.

Population

The population in 1970 was approximately 140,000 and the 1990 projected population is 180,000 with an estimated service area of 65,000 hectares (150,000 acres).

SYSTEM DESCRIPTION

The Muskegon County Wastewater Management System is comprised of two separate lagoon-impoundment spray irrigation systems of similar design. One system is large (159 TCMD [42 MGD] wastewater treatment design capacity) and one is small (5.3 TCMD [1.4 MGD]). The small separate system was built as an economical alternative to a long expensive sewer connection. This report discusses only the large system.

The system consists of collection, transmission, aeration, storage, irrigation, soil, crop, and drainage components (Figure 1). The system treated 27 MGD of wastewater in 1975, 60% of which was industrial flow, leaving a reserve capacity of 15 MGD for serving additional residential and industrial development. (Acreage requirements of various portions of the system are shown in Table 1).

Wastewater Collection and Transmission

Wastewater is collected via connector sewers and ten lift stations which deliver wastewater to a central pumping station. The wastewater is pumped from the central lift station 18 km (11 miles) through a 1.7 meter (66 inch) diameter pipe line to the land treatment site. The listing of those municipalities and industries connected to the collection system with their respective discharge volumes for 1975 is in Table 2.

The main pumping station consumed 7.8 million KWH of electricity in 1975 and the feeder stations consumed a total of 2.8 Million KWH the same year. Energy consumed in following years was similar.

Aeration and Storage

After reaching the treatment site, wastewater can be aerated in each of three 8-acre 42-million-gallon capacity aeration cells. There are six 50-horsepower mixers and twelve 60-horsepower aerators in each cell. Research and operating experience justified reducing the amount of aeration needed and cut electrical energy used drastically. The aeration mode most often used at this time is treatment with 8 aerators in cell 1 followed by treatment with 4 aerators in cell 2. With the current 27 MGD flow, the average retention time is about 1.5 days in each cell. After aeration in winter, the water flows into the storage lagoons. During the summer, the aerated water may either be sent into storage or retained briefly in an 8-acre solids settling cell before application on land through the irrigation system.

There are two storage lagoons, each 850 acres in size, with a combined storage capacity of 5 billion gallons. These large lagoons serve several functions: they provide a large reservoir of water for the irrigation season; they constitute a holding area in which suspended solids in wastewater may settle out; and, depending upon the duration of retention of the water, they facilitate the breakdown of BOD, toxic materials and pathogenic microorganisms.

Surrounding the storage lagoons are interception ditches which are designed to catch leachate from the lagoons as well as to create a small hydrologic gradient in the groundwaters toward the lagoons. This leachate--or seepage water--is monitored for water quality on a daily basis, and as it meets the standards of the Michigan Department of Natural Resources, it is pumped directly into the nearby receiving streams. Otherwise, it is pumped back into the lagoons.

Biological treatment, like the collection and transmission pumping, is a major consumer of electrical energy. The breakdown by year of power consumption by the biological treatment cells is in Table 3, and it shows that though the system was operating only wight months in 1973, the KWH requirements were about 40 percent higher than for the full year of 1975.

This reduction in electrical energy for biological treatment was achieved without appreciable sacrifice in efficiency of BOD removal. In fact, removal of BOD per KWH of electricity increased as shown in Figure 2. The additional BOD loading into the storage lagoons has caused no problems.

Final Disinfection

Before water is irrigated onto the land, it enters a discharge cell. Prior to entering irrigation ditches that supply the water to pumping stations, the water is chlorinated as necessary to meet health standards.

Irrigation and Farming

The pretreated wastewater is distributed to center pivot irrigation rigs via buried asbestos cement pipes. The operating presure is from 30 to 70 psi depending upon location in the system. The rigs were especially designed for spraying wastewater with downward pointing low pressure nozzles (Figure 3). There are 54 center pivot irrigation rigs (Figure 4) located in circular fields of 35 to 140 acres. The soils are mostly sandy.

During the 1975 season, 4,500 of the 5,400 irrigated acres were planted with corn and irrigated with up to 4 inches per week of wastewater. The other 900 acres were fallow or in rye grass. Total wastewater applied to the 5,400 acres varied from none to over 100 inches per field during 1975. Irrigation was performed from mid-April to mid-November with time out for cultivating, planting, and harvesting the corn crop. Irrigation pumping required about 5.2 million KWH of electricity in 1977.

Thus far corn has been the main crop. Corn planted in 1975 yielded an average 60 bushels per acre (31 to 90 bushels per acre, Table 4), which was nearly equal to the 65 bushel per acre average corn grain yield in Muskegon County on operating farmland. The average corn grain yield on 4,700 acres in 1974 (the initial year of operation) was 28 bushels/acre. The corn has been marketed through normal channels.

The 1974 and 1975 average grain yields of 28 and 60 bushels per acre are indeed remarkable considering the following: (1) the Muskegon system was new and untried; (2) the primary purpose of the system is wastewater renovation; (3) incomplete installation of irrigation equipment and many operation difficulties with the new irrigation system have caused interruptions particularly in 1974; and (4) most soils at the site are very poor, will not yield more than a few bushels per acre of corn grain without irrigation and nutrients, and normally only support scrub oak and other forest species.

During the 1976 and 77 seasons all fields were planted to corn and irrigated with wastewater. The yields averaged about 80 bushels/acre each year indicating a leveling off in corn production balanced against the primary goal of wastewater renovation. Some additional increase in average corn yield is expected as additional drainage problems in some of the fields are overcome and additional improvements are made in liquid nitrogen fertilization and weed control practices.

Drainage

Before construction, the groundwater table was very close to the soil surface in many of the fields. Tile drainage or drainage wells were installed and the water table lowered. The drainage network now collects the sprayed renovated wastewater after it has percolated through the crop soil filter and discharges it into the receiving stream. The drainage network (105 km long), along with interception ditches around the storage lagoon, is designed to protect the quality of the groundwater. Another interesting aspect of the design of the site drainage system was the bypassing of all drainage from lands upstream, and construction of berms to prevent storm water run-off from the site. Quantities of water drained to the receiving streams are shown in Table 5.

Operations, Management, and Reserach and Development

The entire system is being operated by 40 full-time persons and an additional part-time labor force of up to 10 workers. Some of their job activity is associated with the Muskegon EPA Research and Development Grant. It is essential for large operations of this nature to have laboratory and development study capabilities.

The success of this operation has depended and will continue to depend heavily on expert management which in turn is based on sound business, farming, engineering, and scientific skills.

Managment has benefited from the creation of a Farm Advisory Board made up of Agricultural Experts from Michigan State University and from a Research Advisory Board made up of experts within EPA. As a direct result of good management, directly assisted by research and development efforts, progressive improvements have been achieved and operational problems (inherent not only in any system, but also in a large previously untried system of this nature) have been overcome at very modest cost.

Examples of these coopertive management-research efforts are: (1) Studies and steps undertaken to eliminate problems with underground electrical cable and irrigation mains to irrigation rigs: (2) Economic stepwise modifications of the system to reduce problems with irrigation rig nozzle plugging and to overcome occasional odor problems from industrially discharged flows at the site; (3) Studies and steps undertaken to improve drainage in a number of inadequately drained fields; (4) Significant reduction in energy consumption based on studies of aeration cell operation; and (5) Studies to optimize population density of corn, nitrogen fertigation and weed control. Other examples of improvements, resulting from good management, are: Reduction in the full-time labor force from over 60 to 40; acquistion and updating of equipment for more efficient farming and self-maintenance of nearly every phase of the system; and hedging to insure and improve cash revenues from the corn crop.

Treatment Performance

Wastewater is being renovated to the degree shown in Table 6. Examination of this information shows that the goal of providing clean, high quality renovated water is being met by the Muskegon County Wastewater System.

The Irrigation-soil-cropping phase of the wastewater treatment system is providing not only what is often called tertiary or advanced wastewater treatment (AWT), but also utilized nutrients in the wastewater for growing a corn crop. Over $100,000 worth of nitrogen, phosphorus, and potassium from the wastewater was utilized in 1975 to improve the soil and grow food.

Calculations and experience has shown that if, for example, 3 inches of wastewater were applied each week over a 6-month season, an adequate amount of phosphorus and potassium is available for the corn crop (Table 7). However, the level of nitrogen would not be adequate because of the low nitrogen level present in the wastewater and because soils do not retain much nitrogen. In addition the nitrogen is utilized by corn primarily only during 2 months of the 6-month irrigation period. The nitrogen fertilizer, therefore, was injected (fertigated) into the wastewater daily during the active part of the corn growing season to increase corn growth and yield and to stimulate increased removal of phosphorus, potassium, and other wastewater nutrients. From 0-89 pounds per acre of nitrogen fertilizer was added to the different irrigated fields, depending upon the amount of wastewater applied and corn crop's needs (Table 8).

All Circles

The phosphorus (P) concentration in the land sprayed wastewater has been just over 1 ppm. At the indicated level of wastewater application the P applied is about equal to crop needs and therefore will not accumulate in the soil. If the total P level is 3 ppm, a majority of soil at the wastewater site should be able to remove the excess P and prevent leaching with the renovated wastewater for at least 50 years. This assumes that wastewater is sprayed uniformly on the soil under the conditions shown.

The quantities of potassium (K), sodium (Na), and calcium (Ca), in excess of crop needs are not causing problems on the sandy soils at Muskegon. Soil monitoring studies by Michigan State University indicate that these wastewater elements are reaching an equilibrium with the soil sorptive surface and will be passing through into the renovated wastewater with little change in concentration but at acceptable levels. Studies on other inorganic and organic constituents in the wastewater do not indicate any anticipated problems under the current mode of operation and with the wastewater as it now exists.

IMPACT ON GROUND AND SURFACE WATER

Monitoring

There is an extensive monitoring system. Monitoring results are used to determine the efficiency of treatment and to assure that the operation avoids the degradation of ground and surface water. Samples are taken for chemical and biological analyses once or twice daily at each step of the treatment process. Groundwater is sampled monthly to twice yearly from the over 300 wells for analysis. Finally, the surface water quality is examined in lakes and streams at some distance from the treatment site, but still under the influence of the wastewater treatment system.

Groundwater

There were 233 monitoring wells around the storage lagoons. Water quality results were summarized by averaging as given in Table 9. Preoperation nitrate nitrogen averaged 0.18 mg/l and doubled to 0.39 mg/l after start-up; in the same time frame, chloride went from 4.8 to 13 mg/l, and conductivity was slightly raised from 2.0 to 2.7 umhos/cm [x100]. Phosphate was stable. With the exception of chloride, the order of magnitude of these changes was not considered significant. A convincing argument may be made that the large change in chloride concentration is due to factors other than the wastewater treatment operations, namely past oil drilling operations.

There were also 56 site perimeter wells. The averaged data for preoperation and the 1974-75 post operation periods show essentially no change, and therefore are not given.

Surface Water

The quality of water in the receiving streams has remained about the same since diversion of wastewater to the treatment site in 1973, even though there has been a slight degradation of stream water quality below the treated effluent discharge points. The water quality in the lakes, however, has improved (Figure 5). The level of improvement is expected to increase further as more industries and communities within Muskegon County join the Wastewater Treatment System. The level of improvement is not adequate to eliminate eutrophication.

DEVELOPMENT COSTS AND CONSTRUCTION

Judged by any standard, the time frame in which all major construction components of the project were completed is remarkable: a period of 20 months. Start up was May 8, 1973. Total development costs are given in Table 10.

Excluding land acquisition the construction of the collection and transmission network comprised 35 percent of the construction costs. Installation of the force mains and the pumping stations was done between October, 1971, and June, 1973. With extremely minor exceptions, the collection system was on line to receive wastewater in late April, 1973.

Cost associated with excavations amounted to about 35 percent of the total. In the formation and installation of the aeration and storage lagoons, the amount of soil removed was 2.29 MCM (3 millioncubic yards), over an area of 8,645 ha (3,500 a). The digging of the drainage ditches involved over 16 km (10 mi) of excavations. All excavation work was completed within a period of 8 to 12 months; aeration lagoons were ready by January, 1973, storage lagoons by June 1973, and drainage ditches by February, 1973.

The distribution segment, including irrigation pressure pipelines, electrical lines and irrigation rigs, represented 11 percent of the total consturction costs. Sequencing required that the pipelines and underground cables be installed after the completion of the underground drainage system, so full-scale construction of these elements did not begin until September, 1973, and most of the work was not completed until late winter. The pipeline-electrical work lasted from September, 1973, until April, 1974. Installation of the irrigation rigs spanned the eleven month period of August, 1973, to June, 1974, and was likewise sequenced after the underdrainage system.

Underdrainage costs amounted to 9 percent of the total construction eosts. The installation of 109 km (68 mi) of plastic drain laterals was accomplished in five months (June-October, 1973) by the use of laser devices mounted on excavating equipment; this new technique lowered manpower requirements, minimized excavation excesses and dramatically lowered costs. To assay the importance of the laser technique in lowering construction costs, refer to the original bids: of the $50 million total original bid, the cost of installation of underdrainage by conventional means was $12.5 million. Actual cost was $758,000.

Muskegon County paid $16 million of the development costs, the State of Michigan $8.4 million, and U.S. EPA $20.1 million. The county issued $16 million worth of bonds to cover its needed capital outlay. The 1975 bond repayment was $1.2 million ($0.3 million capital and $0.9 million interest). Final repayment is due in 1997. Land acquisition costs were not eligible for federal funding at the time the grant was awarded, however, relocation allowances were. Approximately 190 families and 4 businesses were relocated.

OPERATION AND MAINTENANCE COSTS

The 1975 total cost for treatment, including collection, transmission, aeration, land treatment, depreciation, amortization, and debt retirement, was 24¢/1000 gallons of wastewater (Table 11). This cost is charged to users via a 17¢/1000 gallon operational and a 4.5¢/1000 gallon debt retirement fee (22¢/1000 gallon user charge) and acreage charges.

The costs for operation and maintenance compare very favorably with those for operating systems elsewhere within the United States which have comparable or lesser levels of treatment (Table 12).

OUTLOOK AND LIFE EXPECTANCY

Any wastewater treatment system has limitations and the Muskegon County Wastewater System is no exception. Operated, however, in it's present mode (with adequately pretreated wastewater of similar composition and irrigated with similar quantities and rates) most of the cropped soils at Muskegon are expected to adequately remove wastewater contaminants like phosphorus for at least 50 years. If after 50 years, the land were saturated with phosphorus and would no longer be able to provide adequate phosphorus removal, many other uses for the land will be possible. Additional research and development activity should more clearly be able to predict the life expectancy of all parts of the system for handling and treating all wastewater consituents, not only under the current mode of operation but also under different modes of operation using wastewater of different characteristics.

REFERENCE MATERIAL

Data in this report has been drawn from the following reports which have been published or are expected in print in 1978.

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TABLES AND FIGURES

Table 1. ACREAGE ALLOCATIONS AT WMS

Purpose	Allocation Acres	Allocation Hectares	Percent of total
Irrigated with wastewater	5350	2167	49
Wastewater storage lagoons	1700	689	16
Solid waste landfill & municipal & industrial development	1500	608	14
Ditches & roads	1000	405	9
Dry land farmed borders	1000	405	9
Aeration, settling, outlet lagoons, chlorination & other buildings	300	122	3
Total	10850	4395	100

Table 2. USER WASTEWATER QUANTITIES

User	Start-up Date	Daily Volume TCMD	Annual Volume TCM
City of Muskegon	5/10/73	27.7	10,086
City of Norton Shores	5/10/73	2.60	967
City of Roosevelt Park	5/10/73	2.60	967
City of Muskegon Heights	5/30/73	4.50	1,658
City of North Muskegon	6/9/73	1.10	415
S.D. Warren Company (pulp mill)	6/4/73	55.3	20,172
Story Chemical Corp.	4/18/74	3.40	1,244
Total		97.1	35,509

Table 3. ELECTRIC POWER CONSUMPTION IN BIOLOGICAL TREATMENT CELLS 1973-1975

Cell number/Apparatus	1973, KWH[c]	1974, KWH	1975, KWH
Cell number 1			
Aerators[a]	3,152,736	3,885,521	2,959,471
Mixers[b]	1,328,400	2,001,648	899,157
Cell number 2			
Aerators	3,046,464	3,853,868	1,863,983
Mixers	1,328,400	1,963,874	708,248
Cell number 3			
Aerators	1,700,352	443,004	731,899
Mixers	649,440	527,723	218,606
Total	11,205,792	12,675,638	7,381,364

[a]44.7 KW
[b]37.2 KW
[c]Cells in operation eight months

Table 4. INCREASED AGRICULTURAL PRODUCTIVITY BY RENOVATION/REUSE OF WASTEWATER IN MUSKEGON

	Corn Yield and Income 1974	1975	1976
		bu/acre	
Wastewater Site	28	60	81
County Average	55	65	45-50
		millions of dollars	
Gross Crop Revenue	0.35	0.7	1.0 (est.)

Table 5. DRAINAGE SYSTEM DISCHARGE VOLUMES, 1973-1975

Receiving waters & drainage source	Year, TCM 1973	1974	1975
Mosquito Creek			
Drain tiles[a]	16,580	23,027	25,207
Wells[b]	0	0	(est.) 0
Lagoon Interception Ditch	7,631	12,079	9.797
Total	24,211	35,106	35,193
Black Creek			
Drain tiles	5,186	7,378	6,447
Wells (circles 36,37,39, & 40)	0	(est.)378	378
Lagoon Interception Ditch	5,697	5,364	6,560
Total	10,883	13,120	13,385

[a]Outfall discharge minus wells and Ensley Pump Station
[b]No Discharge due to mechanical-electrical problems

Table 6. TREATMENT PERFORMANCE, 1975

Parameter	Influent	After Aeration	After Storage	Discharge* After Crop-Soil Filter	NPDES 30 Day Limit	System Design
		mg/liter				
BOD_5	205	81	13	3	4	4
COD	545	375	118	28	-	-
Suspended Solids	249	144	20	7	10	4
Total P	2.4	2.4	1.4	0.05	0.5	0.5
Ammonia-N	6.1	4.1	2.4	1.4	0.6	0.5
Nitrate-N	Trace	0.1	1.1	1.9	-	5
Zinc**	0.57	0.41	0.11	0.07	-	-
Fecal Coliform	$>10^6$/100 ml	$>10^6$/100 ml	10^3/100 ml	$<10^2$/100 ml	200/100 ml	0/100 ml

*Mosquito Creek
**Representative of heavy metal contents

Table 7. WASTEWATER NUTRIENTS ADDED TO SOILS

Element	Sprayed Wastewater Conc.	Nutirents in 75-inch (season) Effluent Sprayed	Nutirents for 100 bu/acre Corn Grain
	ppm	lbs/acre	lbs/acre
N	4-7	68-119	85
P	1-3	17-51	15
K	6-11	102-187	20
Na	140	2400	
Ca	60	1000	

Table 8. REPRESENTATIVE YIELDS OF CORN GRAIN IN 1975 BY IRRIGATED FIELD AT MUSKEGON

Circular Field Number	Soil Type	Wastewater Applied	Supplemental Nitrogen Fertilizer	Drainage Status	Yield
		inches	lbs/acre		bu/acre
5	Rubicon Sand	106	63	Excellent	83
8	Rubicon Sand	93	44	Excellent	53
17	Au Gres Sand	59	70	Good	71
37	Au Gres Sand	14	10	Poor	36
32	Granby Loamy Sand	14	27	Poor	61
18	Roscommon Sand	57	65	Good	90
35	Roscommon Sand	69	40	Poor	69
36	Roscommon Sand	14	0	Poor	31
ALL CIRCLES		54	44		60

Table 9. COMPARISONS OF SELECTED WATER PARAMETERS ON ALL LAGOON SEEPAGE WELLS BEFORE AND AFTER WASTEWATER OPERATIONS

Parameter	Year	Average	Standard Deviation
NO_3^--N, mg/l	1972-1973	0.18	0.91
	1974-1975	0.39	1.39
Chloride, mg/l	1972-1973	4.80	16.8
	1974-1975	13.0	38.9
PO_4-P, mg/l	1972-1973	0.03	0.05
	1974-1975	0.02	0.06
Conductivity, umhos/cm[x100]	1972-1973	2.00	1.00
	1974-1975	2.70	2.00

Table 10. DEVELOPMENT COSTS FOR MUSKEGON WASTEWATER SYSTEM

Component	Millions of dollars
Collection	$ 5.2
Transmission	6.8
Pre-Application Treatment-Aeration	3.1
Storage (5 billion gallons)	5.2
Land & Relocation	
Purchase	5.4
Relocation	1.2
Clearing	1.9
Distribution-Irrigation	4.1
Recovery-Drainage	3.7
Interest & Engineering	3.8
Other	2.3
TOTAL MUSKEGON SITE	42.7
TOTAL WHITEHALL SITE	0.8
TOTAL CAPITAL COSTS	43.5
NON-CAPITALIZED COSTS	1.0
TOTAL SYSTEM DEVELOPMENT COSTS	$44.5

Table 11. NET OPERATING COSTS FOR MUSKEGON WASTEWATER SYSTEM, 1975

Gross Operating

By Component

Collection & Transmission	$ 431,000
Aeration & Storage	191,000
Irrigation & Drainage	475,000
Farming	474,000
Laboratory & Monitoring	236,000
Other	77,000
Total Muskegon	$1,886,000
Total Whitehall	$ 62,000
TOTAL GROSS OPERATING	$1,946,000

Bond Repayment $1,200,00

Revenues

acres x bu x $/bu

Crop:	Corn	4500 x 60 x $2.58	$689,000
	Wheat	270 x 10 x $3.10	8,000
Laboratory services			8,000
TOTAL REVENUE			$714,000

Summary

Costs, millions of dollars

Operating	$ 1.9
Bond Repayment	1.2
TOTAL GROSS	$ 3.1
Revenues, Corn	$ 0.7
TOTAL NET	$ 2.4

NET COST TO TREAT WASTEWATER IN 1975
(27 MGD FLOW)
$0.24/1000 GALLONS

Table 12. WASTEWATER OPERATION AND MAINTENANCE COSTS, 1977

Wastewater Treatment Plants	O & M Costs $/1000 G	Water Treated MGD	Level of Treatment
Laurel, MD	0.65-0.80	5(7.5)	P + S + CP
Lake Tahoe, CA	2.70	3.5(7.5)	P + S + CP + N, etc.
East Lansing, MI	0.40-0.45	12(19)	P + S + CP + SF
Muskegon Co., MI	0.25	27(43)	Aeration + SI

( ) = Capacity available but not yet utilized
P - Primary
S - Secondary
CP - Chemical precipitation of phosphate
SF - Sand Filter
N - Nitrogen treatment
SI - Spray Irrigation

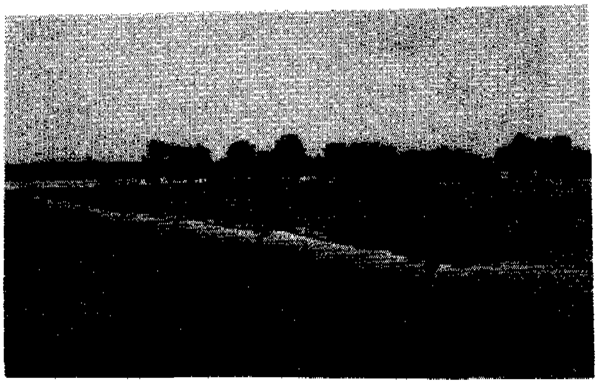

FIGURE 1 Aeral view of the Muskegon County Land Wastewater Treatment System. Shown are some of the circular irrigated fields, the storage lagoons and aeration cells.

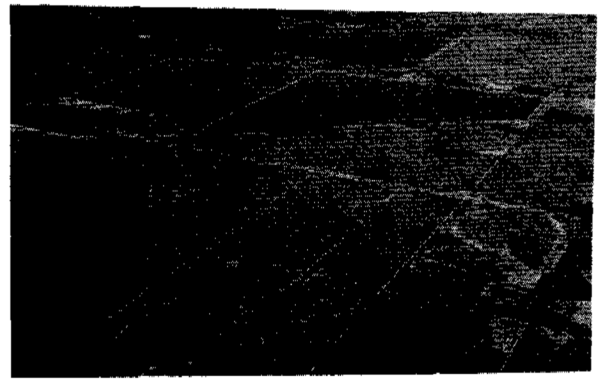

FIGURE 3. Center-pivot irrigation rig with downward pointing nozzles spraying corn at Muskegon
(Photo by the Muskegon Chronicle)

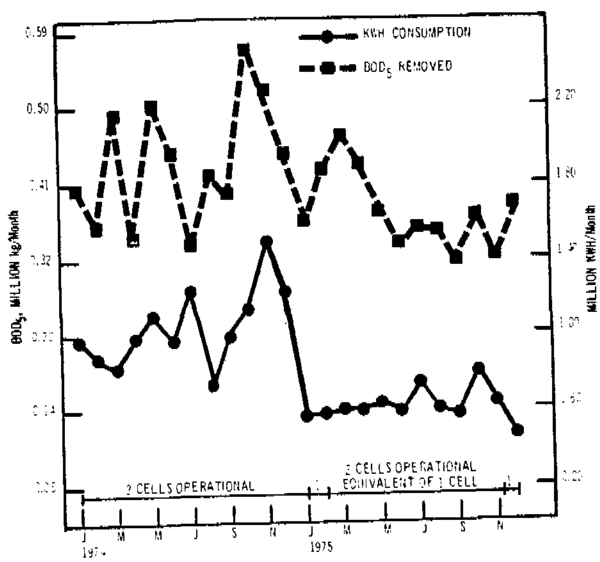

FIGURE 2. Five-day biochemical oxygen demand removal and KWH consumption in the biological treatment cells. 1974 & 1975.

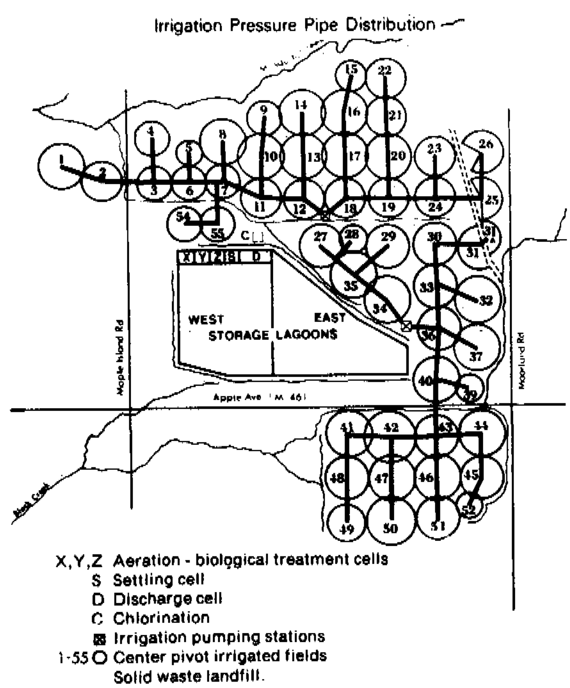

FIGURE 4, Muskegon Wastewater Treatment Site,

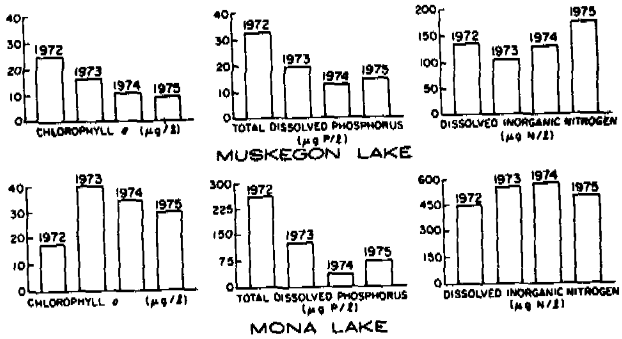

FIGURE 5. Annual average surface water concentrations of selected water quality variables in the Muskegon Lakes.

LAND TREATMENT OF WASTEWATER IN ISRAEL

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Abstract - Israel suffers from a severe shortage of water resources with over 95% of the total available water supplies being currently utilized. The utilization of wastewater has been included as a major national water resource in Israel's water planning for over 20 years and by 1978 about 20% of the available wastewater in the country was being utilized. Major regional projects are underway to quadruple the amount of reuse during the next ten years.

The health aspects of the land application of wastewater have been studied extensively in Israel. The 1970 cholera outbreak in Jerusalem demonstrated the risk associated with uncontrolled irrigation of salad crops with wastewater. Studies on the dispersion of aerosolized microorganisms from wastewater sprinkler irrigation have detected coliforms at 450 m. Salmonella sp at 60 m and enteroviruses at 100 m down-wind of fields sprinkler irrigated with partially treated non-disinfected wastewater. A retrospective epidemiological study in 77 Kibbutzim (co-operative agricultural settlements) practicing sprinkler irrigation with partially treated, undisinfected wastewater indicated that there appears to be a significant increase in enteric disease rates (salmonella, typhoid, Schigella and Infectious Hepatitis) in Kibbutzim practicing wastewater sprinkler irrigation, as compared to settlements not utilizing wastewater in any form. Sewage treatment leading to effective pathogen reduction in the effluent is recommended prior to land application in the vicinity of residential areas.

Studies on the use of the drip irrigation method with wastewater effluent indicate that there may be certain hygienic advantages using this agronomic technique developed in Israel.

Israel with its high level of research capacity can provide an excellent full scale laboratory for the study of many of the technological, agronomic and health aspects of the land application of wastewater since some 250 land application projects are in operation. Other major projects utilizing a wide variety of techniques are under development.

WATER RESOURCES REQUIREMENTS

Israel is a quasi-arid country having an annual average rainfall of 550 mm/year (22 in/year) in the central coastal plain which drops to 25 mm/yr (1 in/yr) at the soutern tip of the country on the shores of the Red Sea. The rainfall is seasonal with 90% distributed during the months of November to April. With a population of some three and one half million persons in 1978, Israel's population has more than quadrupled since the country achieved independence in 1948, while the area under irrigation increased more than 5 fold during the same period.

Large scale water supply development projects have tapped most of the country's surface and underground water resources and it has been estimated that in 1978 current water supplies for agricultural, industrial, and municipal purposes are utilizing some 95% of the country's economically feasible water sources. While present water supplies for all purposes have reached about 1600 million cubic meters per year (MCM) (4×10^5 MG/yr) it is estimated that by the year 1990 the country's population will reach some 4½ million persons and that there will be a deficit in water supply of some 200-400 MCM/year ($5\text{-}10\ 10^4$ MG/yr).

While sea water desalination may eventually have to be developed to supply additional required water for industrial and municipal use, the maximum reuse of wastewater has become a matter of national water resources planning policy for well over 20 years. Present estimates indicate that the cost of desalinization of seawater will most likely preclude its use for agricultural purposes except in the case of a few select high priced crops grown under special conditions for export. Thus the severe future water resources constraints facing the country have made wastewater reuse a priority program in Israel.

DEVELOPMENT OF LAND APPLICATION PROGRAMS

In the early 1950's projects to study the agronomic and health aspects of wastewater reuse in agriculture were initiated (Shuval, 1951; Bergner-Rabinovitz, 1956) which led to the development of wastewater irrigation regulations by the Ministry of Health. These regulations were aimed at limiting land application of wastewater to crops not eaten raw or to agricultural practices avoiding direct contact between edible crops or fruit and wastewater such as by ridge and furrow irrigation. However, the irrigation of crops such as potatoes normally used for human consumption after cooking but grown in direct contact with sewage was allowed. The early policy developed by the Ministries of Health and Agriculture favored minimum treatment of wastewater in anaerobic and aerobic-facultative oxidation ponds with a total detention period of about 10 days with serious restrictions as mentioned above on the types of crops irrigated. Raw wastewater is generally introduced into two parallel anerobic ponds with a depth of about 2.5 m and a detention time of about one day, with a load of 1500-2000 kg B.O.D/hectare/day. The primary ponds can be cleaned of accumulated sludge alternatively. The secondary aerobic ponds which follow generally have a depth of 1.2-1.5 m and a detention time of 7-10 days with a loading of about 150 kg B.O.D./hectare/day. (Noy and Feinmesser, 1977). Experience in Israel has indicated that such secondary ponds generally remain aerobic during most of the winter period. However, on occasion serious odor problems have been associated with ponds in Israel, generally due to overloading during winter months.

The possibility of much higher levels of wastewater treatment including disinfection so as to allow for unrestricted crop irrigation was ruled out on economic and policy grounds during the earlier phases of the wastewater reuse program in Israel but is now under consideration. (Shuval, 1977). As land values have increased in the vicinity of urban areas aerated ponds have been introduced to allow for a reduction in the pond area required.

In a survey of wastewater utilization for agricultural purposes carried out in 1975 (Feinmesser and Wikinski, 1975) it was reported that in the urban sector encompassing some 2.75 million persons 87% were connected to central sewerage systems. Some 30 MCM/year of the sewage was utilized for agriculture representing 15% of the total urban sewage. In the rural sector surveyed, representing a population of 110,000, 7.5 MCM/year was utilized, which is 33% of the total rural sewage available. On a countrywide basis some 17% of the total sewage flow was utilized, 16.5% directed to the land for agricultural purposes and 0.5% for industrial use. The total land area under wastewater irrigation in 1975 reached 5,200 hectares (13000 acres), while 4.2 MCM/year where utilized in fish ponds. At that time there were 49 urban and 153 rural waste water reuse projects in operation with additional major projects in the planning stage. Some 60% of the urban wastewater was used to irrigate field crops, mainly cotton and sugar beets. About 14% was used to irrigate pasture and forage crops with 21% going to citrus and other fruit trees. Of the remaining 5%, 3% went to fish ponds and 2% to various other crops. By 1978 some 20% of the available wastewater flow from sewered areas was being utilized.

LARGE SCALE REUSE PROJECTS

The first major regional wastewater

project includes the city of Tel Aviv, Israel's largest metropolitan center,and six neighboring municipalities which have joined together to form the Dan Region Sewerage Association. Two additional towns will also join the group.

The region's main mode of wastewater disposal has been through outfall sewers to the sea leading to serious pollution of nearby Tel Aviv bathing beaches.

The Dan Region Wastewater Reclamation Project was initiated in the early 1960's by Tahal-Water Planning for Israel with the aim of solving the problem of beach pollution and of recycling the large volumes of wastewater to the national water system (Amramy, 1965) (5). In 1978 the total population included in the region was about 1,000,000 persons. The estimated population for 1985 is 1,300,000 reaching about 1,700,000 in the year 2000. Total mean water consumption of the region in 1978 was .3MCM/day (75 MGD). Estimated water consumption for 1985 is .41 MCM/day (103 MGD) reaching .55 MCM/day (137 MGD) by the year 2000 (Amir and Ideliovitch, 1974).

Both due to the recent drastic inincreases in the costs of energy which has made seawater desalination an impractical alternative water source at this time, and the urgent need to reduce the pollution of the Tel Aviv beaches, the Dan Region Project has been given the highest priority in the national water resources development program. The project is now in an advanced stage of design and construction.

The Dan Region Project includes the construction of major intercepting sewers to convey the wastewater from all the towns in the region to the treatment site in the sand dune area along the coast south of Tel Aviv. There the wastewater will be processed by biological and chemical treatment processes prior to reuse. One reuse plan calls for pumping the effluent to recharge basins located on the sand dunes. The water will be pumped from the aquifer by a series of recovery wells located several hundred meters away from the recharge basis. The recovery wells have been designed to provide for a residence time of some 400 days in the aquifer and for a degree of dilution, with the ground water. The reclaimed water extracted from the aquifer will then receive final polishing treatment and further dilution. The reclaimed water will be conveyed mainly to the Negev area south of the Dan Region for agricultural use. An alternative reuse plan calls for direct land application of the treated effluent in the southern Negev area, thus avoiding ground water recharge. Surface storage of the effluent will be required and may involve a number of sanitary problems not yet fully solved.

First Stage

The first stage now in operation has a design capacity of .041 MCM/day (10.3MGD) and serves the southern suburbs of Tel Aviv as well as three neighboring towns. The treatment as now developed includes: biological purification in recirculated facultative oxidation ponds having an area of 120 hectares; chemical treatment by excess lime precipitation; ammonia stripping in open ponds; effluent stabilization in ponds and lime sludge disposal. The biological treatment ponds have been operated since 1970 while the chemical treatment and recharge of effluent was initiated in the early part of 1975. A series of observation wells have been placed at varying distances between the ponds, recharge basins, and recovery wells so that the effects of percolation from the ponds and recharge basins on the quality of the water in the aquifer can be determined at various stages prior to the arrival of the recharged water at the recovery wells.

Second Stage

The second stage will have a capacity of .27 MCM/day (68.5 MGD) and will serve the whole area of the Dan Region. This stage is now in an advanced stage of planning. The treatment includes: advanced biological purification by a modified low-rate activated sludge process which will provide for nitrification and denitrification, (the treatment in oxidation ponds will be abandoned in the second stage due to the large land area required); in-plant chemical precipitation; post aeration and disinfection of the effluent. Excess sludge disposal by means of a long sea outfall which will assure suitable dilution, dispersion and bacterial decay before reaching the shore lines is being considered as one possible solution. Existing ponds from the first stage will be integrated into the operation of the second stage treatment plant in order to reduce peak loads reaching the main

plant, thus allowing for savings in capital and operating costs (Idelovitch, 1977).
The additional area of irrigated land provided for by this project is about 10,000 hectares (25,000 acres)

The Kishon Reuse Project

A large scale regional reuse project for the wastewater of the city of Haifa and its neighbors, with a population of some 500,000, is in the advanced planning stages. The chlorinated effluent from the region's activated sludge plant will be conveyed to a large storage reservoir which also collects certain amounts of flood water. The diluted effluent will be withdrawn from the Kishon Reuse Project storage reservoir for agricultural use by agricultural settlements in the Jezriel Valley, enabling the irrigation of some 5000 hectares (12,500 acres). It is anticipated that the large scale Dan and Kishon regional projects, in addition to numerous other small projects will lead to a major increase in reuse of the country's water resources, with some 75% of the country's total available wastewater being utilized. Many of the newer reuse projects will involve the construction of reservoirs to store sewage over the winter for use during the 5-6 months irrigation season. The environmental implications and ecological problems involved need to be studied intensively so as to establish design criteria for such long term sewage reservoirs.

HEALTH PROBLEMS IN LAND APPLICATION

The fact that improper land application of wastewater can lead to serious health problems was dramatically demonstrated during the course of a cholera outbreak in Jerusalem in 1970 (Cohen et al,1971). It quickly became apparent that the outbreak was not associated with contaminated drinking water despite the fact that the 250 cases detected appeared in various regions of the city with no clear picture of transmission by direct personal contact. Our investigations clearly showed that after the first few clinical cases of cholera were reported, it was possible to detect Cholera vibrio (El Tor) in the municipal sewage which flowed untreated in a dry river bed past a number of villages practicing illegal irrigation of vegetables including salad crops. Soil samples from sewage irrigated plots and harvested vegetables were found positive for Cholera vibrio as well. It was concluded that the first few individual cholera cases in Jerusalem were introduced by tourists from neighboring Middle Eastern countries where a cholera epidemic had been progressing for a number of weeks. However, once the city's sewage became heavily contaminated with Cholera vibrio excreted by carriers or clinical cases, the main mode of dissemination of the disease to the public at large was through the sewage irrigated salad crops which were sold throughout the city on the same morning as harvested. This outbreak, although associated with illegal sewage irrigation practices led to a tightening up of the sewage irrigation standards and to further intensive studies of the health problems associated with the land application of wastewater (Shuval, 1977).

Dissemination of Microorganisms by Sprinkler Irrigation

Most irrigation in Israel is by sprinkler irrigation systems. The Hebrew University of Jerusalem studies have shown that enteric bacteria and viruses can be detected in the air at considerable distances downwind of wastewater sprinkler irrigation fields (Katznelson et al, 1977). Coliforms have been detected at 450 m., Salmonella sp. at 60 m., while echo, coxsackie, and polio virus have been detected up to 100 m. down wind. In most cases, the effluent for irrigation was partially treated in oxidation ponds with 3-5 days detention time having coliform counts of about 10^6/100 ml.

Further studies have been made in an attempt to determine the role of various meteriological factors in the survival of aerosolized enteric microorganisms. Preliminary findings indicate that relative humidity is positively correlated with bacterial survival, while ultraviolet radiation is negatively correlated. When nighttime samples of airborne bacteria were compared with daytime samples under equivalent wind speeds, relative humidity and distances, ten times more bacteria could be detected at night. It is important to note that in Israel most irrigation (including with wastewater) is carried out at night to reduce evaporative losses and due to more favorable wind conditions.

A significant portion of the land application of wastewater for agricultural purposes is carried out by cooperative agricultural settlements (kibbutzim) where the irrigated fields are mainly 300-1000 m. from residential areas.

To evaluate the possible epidemiological significance of wastewater irrigation, a retrospective study was carried out in kibbutzim practicing various forms of irrigation (Katznelson et al, 1976).

The incidence of enteric communicable diseases in 77 kibbutzim practicing wastewater spray irrigation with partially treated non-disinfected oxidation pond effluent was compared with that in 130 kibbutzim practicing no form of wastewater use. The incidence of shigellosis, salmonellosis, typhoid fever, and infectious hepatitis was two to four times higher in communities practicing wastewater irrigation. No significant differences were found for the incidence of streptococcal infections, tuberculosis, and laboratory-confirmed cases of influenza. Nor were differences found for enteric disease rates during the winter irrigation season (see Fig.1)

This study does not provide definitive proof that the primary mode of the pathogen transmission was airborne aerosolized pathogens resulting from sprinkler irrigation with wastewater. Other routes of infection, such as direct contact between irrigation workers and other residents in the communities is equally possible. However, these studies do indicate that a potential health risk does exist when only partially purified non-disinfected wastewater is used for land application in the close proximity of residential areas. It was concluded that wastewater treatment processes leading to a significant reduction of enteric bacteria and viruses prior to sprinkler irrigation with wastewater in the vicinity of residential areas is a minimum precaution required to provide reasonable protection of the public health.

Since the most widely practiced form of wastewater treatment prior to land application is oxidation ponds, a study was carried out to determine the efficiency of actual operating ponds in the removal of coliforms and enteroviruses. Our studies indicated that while a 90-99% reduction in coliforms

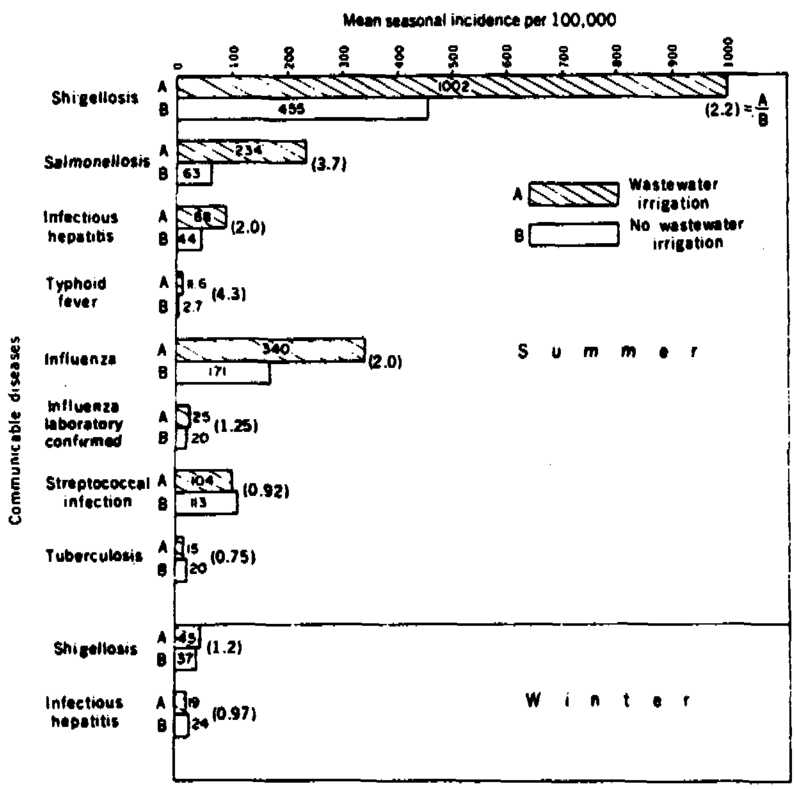


Fig. 1. Mean seasonal incidence of communicable diseases in kibbutzim with and without wastewater spray irrigation.

can be expected from a typical oxidation pond system having a 5-7 day detention period. The removal of enteroviruses as detected on B.G.M. tissue culture cell lines varies from 30-60%. It is the author's opinion that much higher levels of bacterial and virus reduction are desirable prior to land application, particularly if sprinkler irrigation is practiced in the vicinity of residential areas or public parks and roads.

Effective disinfection of a biologically treated effluent would provide the higher degree of safety that appears warranted in light of research findings to date.

Drip Irrigation with Wastewater

One possible method to reduce the risk of aerosol formation during wastewater application to the land involves the use of the drip irrigation technique developed in Israel. This method distributes the water continuously directly to the root zone of the irrigated plant through a small plastic pipe with tiny outlets for each plant.

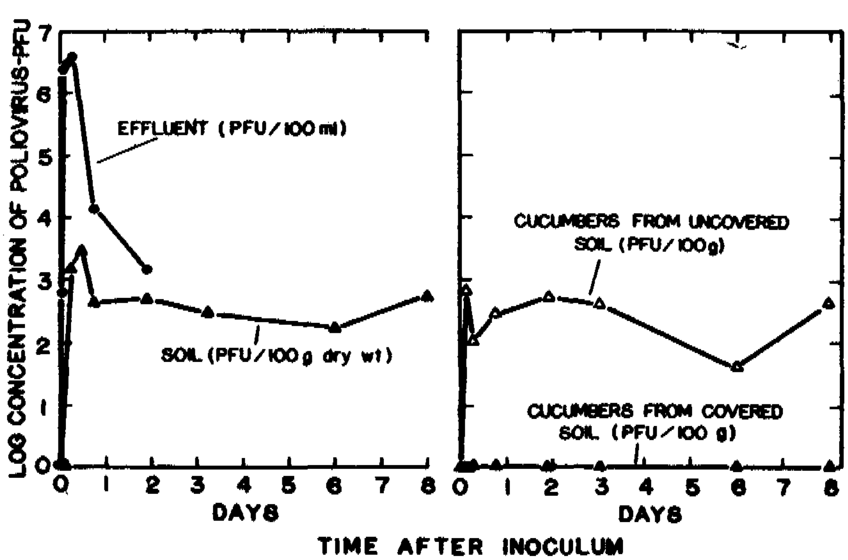
There are numerous agronomic advantages to the method, particularly for arid areas, but certain hygienic advantages may also be obtained when used for wastewater irrigation.

In our studies (Sadovsky et al,1976) we have been able to demonstrate that drip irrigation with wastewater does not prevent the contamination of vegetables which grow on the ground, such as cucumbers, and which are in direct contact with the wastewater wetted soil.

A technique using plastic sheets to cover the drip irrigation lines and the wetted soil area has developed in Israel as a kind of membrane "mulch" to reduce evaporation and cut down on the growth of weeds. Our studies indicate that with the use of such plastic sheets even crops growing on the soil do not in practice become contaminated by sewage.

In Fig. 2 it can be seen that while the poliovirus inoculated into the effluent survived more than a week in the soil and on cucumbers in direct contact with drip irrigated soil, cucumbers drip irrigated on plots covered by plastic sheets were free of poliovirus. The use of drip irrigation techniques for land application of wastewater may hold promise in overcoming some of the health problems associated with land application of wastewater and is worthy of further study.

Fig. 2. Virus contamination level on soil and on cucumbers irrigated with effluent seeded with Poliovirus. Comparison between plot covered with plastic sheets and uncovered plot. (Date of experiment 4.11.74)



Transport of Viruses to Crops Through The Roots

An investigation of the possibility that viruses from wastewater could be transported through the root system to the leaves was carried out (Mills,1977).

Subsoil was inoculated with poliovirus and the leaves of plants grown in the soil were tested for the virus. No viruses were detected in leaves of plants with undisturbed roots, however when the roots were damanged or cut, it was possible to detect viruses in the leaves within a few hours. Whether this mode of virus transport through damaged roots is a real public health risk in cases of drip irrigation with waste water is not clear at this early stage of the investigation.

CONCLUSION

Israel is at this time one of the leading practitioners of land application of waste water due to its severe shortage of water resources and national reuse policy. Some 250 land application projects are in operation with 20% of the country's available wastewater currently being recycled to the land. With the completion of major regional reuse projects now in the construction phase, it is anticipated that about 75% of the country's wastewater will be utilized by 1990.

Due to Israel's high level of engineering, agricultural and environmental health research institutions in both the governmental and academic sectors, the country can serve as an excellent full scale laboratory to study many of the yet unresolved questions associated with the land application of wastewater.

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