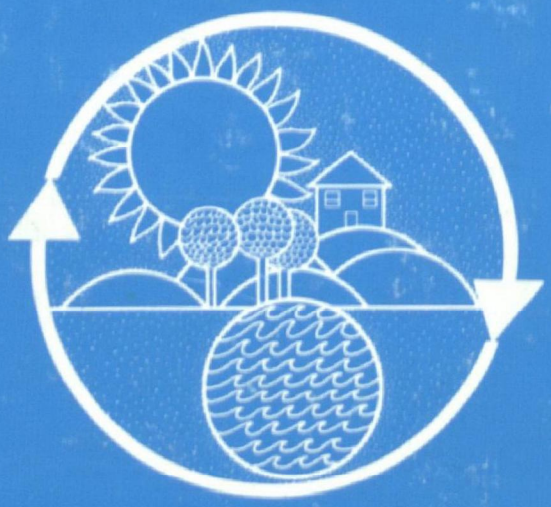
Proceedings of the 1983 Workshop on

UTILIZATION OF MUNICIPAL WASTEWATER AND SLUDGE ON LAND



edited by

A. L. Page, Thomas L. Gleason, III, James E. Smith, Jr., I. K. Iskandar, and L. E. Sommers

Sponsored by
U.S. Environmental Protection Agency
U.S. Army Corps of Engineers
U.S.D.A. Cooperative State Research Service
National Science Foundation
University of California-Kearney Foundation of Soil Science

1983
University of California
Riverside, CA 92521

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PREFACE

The proceedings consist of opening remarks, position papers, and workshop findings. Briefly, the position papers discuss institutional constraints, hydrologic and nutrient management, pathogens, metals, organics, engineering and economics, and an overview of public health effects. These papers set the stage for six separate subject matter sessions. Each workshop used the position papers as a basis for assessing the current state-of-the-art and identifying research needs for its respective topic area. The workshop sessions considered engineering systems, institutional constraints, management considerations in sludge and effluent use, and public health and risk assessment. Following the report of each workshop session appear questions and answers. These questions were raised both during the workshop session and when the moderators presented their findings in the concluding plenary session. In an appendix appear the names and affiliations of the individuals responsible for organizing the workshop and the contributors/participants to the workshop.

ACKNOWLEDGEMENTS

The workshop on Utilization of Municipal Wastewater and Sludge on Land was made possible by support from the U.S. Environmental Protection Agency, the U.S. Army Corps of Engineers, the U.S. Department of Agriculture—Cooperative State Research Service, the National Science Foundation, and the University of California-Kearney Foundation of Soil Science. The financial support from these agencies is gratefully acknowledged.

A number of various federal agencies, colleges universities, and municipalities participated in the organization and planning of the workshop. The idea for a second workshop to evaluate progress made in research and development on land application of municipal wastewater and sludges since the original 1973 workshop originated with Dr. A. L. Page, University of California, Riverside, Drs. Raymond Thacker, William Rosenkranz, Thomas Gleason, and James E. Smith, Jr., U.S. Environmental Protection Agency—Office of Research and Development, and Dr. Alex Iskandar, U.S. Army Corps of Engineers/Cold Regions Research and Engineering Laboratory. Dr. Lee E. Sommers, Purdue University, and A. L. Page made the necessary preliminary plans to establish the Organization and Planning Committee. The Planning Committee met twice (February 12 and June 9, 1982) and developed the format for the workshop. The Committee also selected topics and authors for the position papers as well as topics, moderators, and recording secretaries for the six separate workshop sessions. We acknowledge with appreciation all those who participated in the organization and planning of the workshops. Committee members, along with assignments, are listed in the Appendix.

Special thanks are due to the authors of the position papers, and the moderators and recording secretaries of each of the six workshop sessions. The moderators assumed the responsibility for the conduct of their respective workshop sessions, the presentation of the findings of the workshop session, and with the help of their recordings secretaries, the preparation of their workshop reports. Moderators and recording secretaries also served as reviewers for selected position papers. Sincere thanks are due the 209 invited workshop participants who were assigned to one of the six workshop sessions according to their interest and expertise. These individuals gave freely of their knowledge and experience in addressing the issues set forth in the workshop sessions. A listing of the workshop moderators, recording secretaries, and the invited individuals who participated in each workshop appears in the Appendix.

Local arrangements for the workshop were coordinated with Mr. Stanley Smith and Mr. Roger Dean from the Denver Regional Environmental Protection Agency office and Dr. Hunter Follett of Colorado State University. We acknowledge with thanks their help in handling the many details associated with local arrangements. The most capable assistance of Mr. Michael

Linville, U.S. Environmental Protection Agency—Cincinnati, in the organization of registration and meeting room facilities, and Mrs. Winnie Gillette, University of California, Riverside, who assisted in registration and the typing of visual aids and workshop reports is also gratefully acknowledged.

We acknowledge with deep appreciation the most capable assistance of Mr. Tony Ganje, University of California, Riverside, who reviewed and edited the proceedings from cover to cover.

Finally, we owe a most special debt of gratitude to Mrs. Leslie Webster, University of California, Riverside, who assisted the workshop committee in each and every phase of their responsibilities from the initial meeting to the final publication of the workshop proceedings.

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WORKSHOP HIGHLIGHTS

WORKSHOP HIGHLIGHTS

The significant developments in land treatment and future research needs identified in each workshop are briefly summarized.

POLITICAL AND INSTITUTIONAL CONSTRAINTS

Findings

- Research conducted in the past decade has provided information needed to base federal, state and local guidelines and regulations; identify and resolve design problems and operational constraints, and to identify and resolve general concerns.

- The need exists to provide land treatment information to the general public in an understandable form to allay health fears.

- Unqualified assurance of no risk cannot be provided. Assuming good management in a given situation, assurance can be given that land application of sludges and wastewaters can be accomplished with minimal risk.

- Federal regulations should contain minimum standards for key health parameters; require public notification of noncompliance as well as good management practices; require consideration of reasonable alternatives for management of wastewaters and sludges, and provide sufficient flexibility to permit states and local agencies to combine environmentally sound and socially acceptable management practices with their unique social, political, and physical factors.

- Public acceptance can be improved by involving of elected officials, opinion leaders in decision making, and continuing education and communication programs for all affected groups and individuals.

Research Needs and Recommendations

- Major research needs include: risk assessment data for pathogens and organic contaminants in wastewater and sludges; land treatment system case studies and follow-up studies; a continuing interagency sludge task force, and better communication between universities and municipalities.

ENGINEERING SYSTEMS

Findings

- Land application is a viable, environmentally sound and cost-effective technology for both wastewater and sludge disposal. More than a thousand wastewater land treatment systems are currently in operation; a similar number in various stages of planning are expected to be in operation within the next decade.

- Land treatment is not just a disposal process. This total systems approach allows rational engineering designs to include appropriate provision for pollutant removal within the soil-plant ecosystem. Wastewater treatment is the primary goal of a land treatment system.

- When properly designed and operated, the reliability and performance of land treatment of municipal wastewater systems are equal to or greater than other conventional practices. They are dependable in all climates and geographical areas of the USA.

- Information developed since 1973 permits the design of systems that meet existing water quality criteria.

Research Needs and Recommendations

- Major information gaps exist in the following areas: postconstruction performance evaluations and cost documentation of existing land treatment systems; design criteria for application of municipal wastewater and sludge to forest ecosystems; and heavy metal loading criteria in terms of regional considerations and long-term sludge loadings.

MANAGEMENT CONSIDERATIONS IN EFFLUENT USE

Findings

- Land treatment systems are now successfully operated throughout the world. Infiltration rates, nitrogen leaching and crop uptake of nitrogen, and runoff among other potential problems, can be controlled through proper engineering operation and management.

- Soils and climates have been characterized as to their suitabilities for various land treatment systems.

- Adequate information is available for slow rate systems to assess: crop selection, response, and management; erosion control; hydraulic loading rates; heavy metal, suspended solids and phosphorus retention; quality criteria for irrigation use; and decomposition of most of the organic materials in the wastewater (some synthetic organics are exceptions).

- Adequate information is available for rapid infiltration systems to determine retention of suspended solids, reduction in biochemical oxygen demand (BOD) and heavy metal loadings.

Research Needs and Recommendations

- There is a general need for more technology transfer and training of managers of land treatment systems.

- Mathematical models of slow rate and rapid infiltration systems have been developed but must be verified by field data so that they can be used to maximize the efficiency and reliability of wastewater treatment systems.

- A major research need is for risk assessment of potential human exposure to pathogens in land treatment systems, as opposed to those risks associated with conventional municipal wastewater treatment systems. Also required are long-term studies of ecosystem dynamics both on-site and off-site, of renovation efficiency in relation to site aging; and of ecosystem collapse and recovery in a system not properly operated.

MANAGEMENT CONSIDERATIONS IN SLUDGE USE

Findings

- Guidelines have been developed to enable the environmentally safe use of sewage sludge containing median concentrations of metals and organics when the sludge is applied at agronomic rates based on nitrogen or phosphorus utilization by crops. Sludge can be successfully used as a substitute for conventional fertilizer materials, especially phosphorus and to a lesser extent, nitrogen.

- Current guidelines for the cumulative application of copper, zinc and nickel are safe but should be refined to reflect regional soil differences. Soil properties should be re-assessed to account for regional differences in soil characteristics.

- Concentrations of synthetic organics in sludges are generally low but high concentrations may exist in some sludges. Most synthetic organics are decomposed in soil. Current federal regulations for PCB's in sludges are adequate to protect animal health.

- Groundwater monitoring for nitrate-nitrogen is not needed where sludge nitrogen additions do not exceed fertilizer nitrogen recommendations for the crop grown.

- Utilization of sludge for reclamation of disturbed land at rates higher than those for agricultural land, when properly implemented and managed, improve the quality of soils, groundwater or vegetation.

Research Needs

- Major information needs for the agricultural use of sludge include: determination of the availability of metals to plants following termination of sludge applications; development of models to predict and further refine nitrogen behavior in sludge-amended soils with emphasis on ammonia volatilization, organic nitrogen mineralization and denitrification; and determination of the factors affecting organic chemicals' decomposition, mobility, fate and potential accumulation in plants and animals.

- Major information needs for the application of sludge to forest land include: the quantitative determination of nitrogen cycling, and development of data required to establish criteria for site selection, metal limits, and runoff and leachate quality. Improved application and management techniques are also needed.

- For land reclamation, methods are needed to maintain vegetative cover, and to identify the plant species that are most responsive to relatively high loading rates of sludge as well as tolerant of adverse conditions of disturbed land.

PUBLIC HEALTH AND RISK ASSESSMENT: PATHOGENS

Findings

- With proper management and safety allowances based on research data, land application is a safe, beneficial and acceptable alternative for treatment of municipal wastewater and sludge.

- At the time of the workshop, in the U.S. there are no known outbreaks of infectious disease attributable to land application of wastewater or sludges.

- Major improvements in monitoring methods, especially in the virus area, have been achieved. Data generated during the decade since 1973 provided the scientific bases for criteria at the federal level.

- In terms of current detection capability, federal sludge disposal criteria are adequate to protect public health from pathogenic microorganisms. Data currently being generated will provide guidance to any consideration for relaxation of the criteria. Criteria and management guidance based upon geographical considerations are warranted.

Research Needs and Recommendations

- Although some major research needs identified in 1973 are still pending, research priorties have changed. For example, epidemiological studies no longer remain a high priority research need.

- For bacteriology, information needs include: survival and growth of pathogens, improvement of detection methodology and virulence assessment, effectiveness and mechanisms of treatment processes, relevance of current indicators and their movement through soils.

- For parasitology, the needs include: pathological effects of repeated low level ascarid infections; virulence of swine vs. human ascaris; potential pathological significance of naegleria and similar organisms in waste-treated soils, and survival of ascaris ova in sludge-treated soils.

- For virology, the needs include: site monitoring studies of survival and movement of viruses under a wide range of soils, climatic, hydrologic, and waste conditions; effectiveness of conventional sewage treatment processes along with land treatment, and methodology to detect hepatitis A, Norwalk-agent and rotaviruses.

PUBLIC HEALTH AND RISK ASSESSMENT: ORGANICS AND INORGANICS

Findings

- No longer are there serious gaps in knowledge of the impact on human health of organic and inorganic contaminants in municipal wastewater and sludge when applied to land.

- At annual sludge application rates less than 15 mt/ha, where a reasonable drainage and cyclic establishment of sustained aerobic soil conditions occur, and groundwater remains deeper than 0.3 to 0.7 meters from the soil surface, leaching of metals and organics should pose little or no threat to groundwater processes. However, adjustments may be necessary because of site specific conditions.

- Wastewaters should not be chlorinated prior to high rate application to land since precursors to halogenated organics may form.

- Except of cadmium, heavy metals are not expected to create a human health problem in sludge-amended soils.

Research Needs and Recommendations

- In view of the substantially better data base today, an international panel should re-evaluate the 1972 recommendations of the WHO/FAO limits for cadmium in the diet.

- More research is needed on the factors affecting migration of organics through various soils, and their decomposition in groundwater.

INTRODUCTION AND BACKGROUND

INTRODUCTION AND BACKGROUND

On February 23-25, in Denver, Colorado, a workshop on *Utilization of Municipal Wastewater and Sludge on Land* was convened. A total of 203 invited researchers and practitioners of land treatment from all areas of the United States and abroad assembled to evaluate advances made in the state of knowledge during the last decade, that is, since 1973 when a similar conference was held in Champaign, Illinois. Thirty-seven percent of the participants in the Denver conferences were from colleges, universities, and agricultural experiment stations; thirty-one percent were from federal government agencies; sixteen percent each were from state and local governments and consulting firms. More than thirty people participated in each of the six separate workshop sessions. Significant developments in land treatment since 1973, and critical future research needs and recommendations identified by each workshop session, are discussed in the "Highlights" and "Workshop Findings" sections of these proceedings.

This workshop was co-sponsored by the U.S. Environmental Protection Agency, USDA-Cooperative State Research Service (CSRS), the University of California-Kearney Foundation of Soil Science, the U.S. Army Corps of Engineers, and the National Science Foundation.

The 1973 Conference in Champaign was cosponsored by the U.S. Environmental Protection Agency (USEPA), U.S. Department of Agriculture (USDA) and the National Association of State Universities and Land Grant Colleges (NASULGC). In 1973, there was a need to identify research needs associated with the recycling of municipal sludges and effluents on land. The major approach then was presentation and discussion of state-of-the-art papers, and development of an agenda for critical research. This earlier conference provided the stimulus for an extensive program of research conducted by federal, state, municipal and private agencies.

The data base established from that ongoing research enabled the promulgation of federal and state regulations and guidance concerning municipal sludge and effluent use or disposal on land. In addition to regulations formulated on the federal level, more than 20 state governments established regulations or guidelines for land application of municipal sludges and effluents. Over the past ten years, continuing research has led to the use of land application for treatment of municipal wastewater and sludge at more than 1,000 sites nationwide.

Although substantial progress has been made in land application of sludges and effluents, there was a need to assess and evaluate progress over the past decade. In 1979, USDA-CSRS Technical Committee for "Optimum Utilization of Sewage Sludge on Agricultural Land" contacted the chairman of the EPA-USDA-NASLUGC Subcommittee on "Recycling of Municipal Effluents

and Sludges on Land" to explore the feasibility of convening such a workshop. Subsequently, *ad hoc* meetings were held to discuss in greater depth the possible need to hold a second conference along the lines of the 1973 EPA-USDA-Universities workshop. In 1981, an EPA sludge policy committee was formulated between Agency offices, thereby clearly signaling a need to assess information available and to define future needs for research on land application practices. All of these processes, happening more or less simultaneously, culminated in a planning meeting held in Washington, D.C., on June 9, 1982, where the decision was made to convene a second workshop with the following objectives: (1) reassess the research needs enumerated in 1973 in light of present knowledge on land application of municipal sludges and effluents, (2) identify still existing critical research needs, and (3) formulate recommendations for future research and development.

This publication includes state-of-the-art papers and the findings of the workshop. It also delineates specific research needs and recommendations and thereby provides agencies with the information essential to establish priorities for research. The information contained in the publication will provide guidance for all those involved in research and development associated with the land application of municipal wastewater and sludge.

OPENING REMARKS

A DECADE OF PROGRESS

Courtney Riordan

Acting Assistant Administrator
Research and Development
U.S. Environmental Protection Agency
Washington, D.C.

INTRODUCTION

I am pleased to participate in this second Environmental Protection Agency (EPA), Office of Research and Development (ORD) *Workshop on Utilization of Municipal Wastewater and Sludge on Land.* The co-sponsorship of the other agencies is greatly appreciated. I have an abiding interest in the use of the land as a medium for managing treatment, use, and disposal of sludges. We are revisiting an environmental problem area that is as important today as it was at the time of the first workshop.

This afternoon I would like to review briefly some of the progress made since the Champaign, Illinois Workshop in 1973. A great deal of work has been completed since 1973 in both the municipal wastewater and sludge areas. I expect that the workshop participants will recognize the accomplishments during those 10 years, but I also realized that there is still work to be done. The science, public perception, and socio-economic problems and issues are extremely complex, having defied fully definitive answers for many years.

The ORD within the EPA has long been actively involved in fostering the development of applicable technology and providing scientific data for the Agency in the areas of land treatment of municipal wastewater and sludge treatment, utilization, and disposal. The 1973 workshop accomplished the objective of identifying the state-of-knowledge in these two important technology areas. It was also successful in identifying the critical categories of research and development where the views of national experts were sought concerning additional scientific knowledge and technology needed for successful utilization of land application techniques. Consideration of the economic, engineering, health and aesthetic aspects were included in this request for guidance. The workshop proceedings, *Recycling Municipal Sludges and Effluents on Land,* identified research needed in ten major areas:

1. Educational and Informational Needs
2. Public Health Aspects
3. Dimensions of the Problem
4. Land Resources - Sludge
5. Land Resources - Effluents
6. Plant Characteristics and Response - Soil - Nutrient Relationships, Crop Selection, and Management
7. Plant Characteristics and Response - Toxic Chemicals
8. Options, Problems, and Economics - Engineering Systems
9. Options, Problems, and Economics - Agricultural Management
10. Political and Institutional Constraints

The science and engineering communities utilized the guidance emanating from the workshop to direct their research efforts. The progress toward resolving the issues is substantial and I'd like to take a few minutes to give you my perspective relative to the two areas to be addressed during these three days.

LAND TREATMENT OF MUNICIPAL WASTEWATER

The 1972 amendments to the Federal Water Pollution Control Act (PL 92-500) called for the EPA Administrator to encourage recycling of potential sewage pollutants through the production of agriculture, silviculture, or aquaculture products, reclamation of wastewater and the ultimate disposal of sludge in a manner that will not result in environmental hazards. This paraphrases a key section of the Act, but when placed in the context of a newly established national goal of eliminating the discharge of pollutants, it became clear that use of the land in managing wastewater and sludges was going to be necessary. In 1973 the technology for this concept was in its infancy and we really didn't know how to design land treatment systems with assurance that they would perform satisfactorily.

Several things happened for research on land application of wastewater in a relatively short time in the early 70's. The Muskegon Land Treatment Demonstration Project, largest in the nation, got underway along with several smaller projects and began producing data. Those land treatment aspects which could have impact on public health such as aerosols and toxic materials such as heavy metals in the food chain were intensively studied, along with nutrient management (primarily phosphorus and nitrogen) and engineering design.

The result of these efforts by the university community, state, federal, and local governments was the development and publication in 1977 of the *Process Design Manual for Land Treatment of Municipal*

Wastewater. Continued study of land treatment systems resulted in the 1981 revision of the manual, incorporating the latest scientific and engineering information. This manual has been a real milestone in wastewater management, especially since it incorporated the joint efforts of the four federal agencies primarily involved in the development and evaluation of the technology: the Environmental Protection Agency, the Army Corps of Engineers, Department of Agriculture, and the Department of the Interior.

The 1977 and 1981 amendments to the Clean Water Act recognize that land treatmet technology has advanced beyond the R & D stage and has been designated as an alternative system(s) within the Innovative/Alternative Technology portion of the EPA Construction Grant Program. Most state governments have established regulations or guidance for land application of effluents and municipal sludges. We clearly have come a long way in this area during the past 10 years.

LAND APPLICATION OF SLUDGES

While the use of land for treating wastewater faced unresolved design and management questions as well as socio-economic problems and political issues, resolution of the problems and issues associated with application of sludges to the land has been even more difficult. In 1973 when PL 92-500 came into full play, a major portion of the municipal sludge generated in the United States was applied to the land in some manner. Some was even placed in open dumps and a large amount was stored in lagoons or by other means because of the difficulties associated with disposal.

Soon after the passage of the 1972 amendments to the Federal Water Pollution Control Act it became clear that a great deal more sludge would be generated because of the national requirement of secondary treatment. It also became quickly apparent that the then existing practice of disposal was inadequate for protection of the environment.

The 1973 workshop recognized these facts and identified many of the technical problems and the research and development necessary to resolve them. EPA recognized the need to come to grips with sludge management problems and formed an Interagency Task Force in 1974 to review the technical and socio-economic issues that existed and to develop guidance that could be applied at the state and local level to begin a nationwide sludge management effort. Management would include treatment technology, beneficial uses, and disposal.

The principal result of the efforts of the Task Force came about in 1977 when a technical bulletin *Municipal Sludge Management: Environmental Factors* was issued. Some of you here helped put that document together.

Several other positive things happened during the past decade: the Council for Agricultural Science and Technology (CAST) issued a report in 1976 and another recently (1981) dealing with sludge technology. The 1981 report addresses the problems of cadmium and zinc in sludges and their impact on crops.

A *Process Design Manual for Sludge Treatment and Disposal* was issued by EPA in September 1979. This manual was published to provide the engineering comunity and related industry with a new source of information to be used in the planning, design, and operation of present and future wastewater pollution facilities. This manual describes the latest treatment methods and techniques for more effectively managing municipal sludges.

A design manual on *Utilization of Municipal Sewage Sludge* will be available in October 1983. This manual will give detailed information for (1) agricultural utilization, (2) application to forest land, (3) application to disturbed land, and (4) dedicated land disposal.

Potential health impacts associated with applying sludge to the land have been studied extensively since 1973, addressing the heavy metals problems, nutrient issues, and the microbiological aspects. The parasite aspects have been particularly important since very little sound scientific knowledge was available at that time. In fact, analytical techniques to identify the presence and viability of parasitic organisms were not even available.

No single entity could conduct the R & D needed to fill the technology voids. However, a large amount of progress has been achieved through interagency or other cooperative efforts. A few brief examples will serve to illustrate how this has been done with EPA as one of the partners.

Composting

Cooperative research with the Department of Agriculture (USDA) has developed the static pile process from the concept to full-scale application. Major scientific issues have been resolved and adoption of the process by municipalities has been extensive. Within vessel techniques

applicable to more confined situations are now under study. The USDA work resulted in a manual for composting sewage sludge by the Beltsville Aerated-Pile method, which was published by EPA.

Disinfection

The Department of Energy with some EPA participation, developed a method for disinfecting sewage sludge by means of gamma ray irradiation. Development work was carried out at the Sandia Laboratories utilizing Cesium 137. The method destroys pathogenic organisms, making the product more widely acceptable for use as fertilizer or animal feed. The work has proceeded through pilot-scale and is ready for full-scale evaluation.

The Massachusetts Institute of Technology developed a method for sludge disinfection using high energy electrons. The National Science Foundation supported the work through pilot-scale and at least one city is planning to utilize the method at full-scale.

Land Application

EPA sponsored a field evaluation of a highly coordinated approach to land application of sludge. The project is conducted by the Ohio Farm Bureau Development Corporation in conjunction with Ohio State University. This is a multidisciplinary study considering the socio-political aspects in addition to microbiological and epidemiological features of land application. The five year study has been completed and the report is expected to be completed this fall. The project appears to have produced a successful model approach for use of sludge on private lands for meeting the needs of small communities.

Sludge-Amended Pasture Land

EPA and the Food and Drug Administration (FDA) cooperated with the Metroplitan Denver Sewage Disposal District #1 to evaluate the impact of cattle grazing on pasture land which had received controlled applications of municipal sludge. Pasture land which had never received any sludge was used as a control area. Some direct feeding of sludge was also included in the study, which showed that the effects on cattle from exposure to sewage sludge are essentially benign, especially when the sludge is relatively low in contaminants of concern and tilled into the soil. The direct ingestion studies showed increases of lead in bone tissues and PCB's, DDE, dieldrin and oxychlordane in fat tissues. With regard to cadmium, there was uptake into livers and kidneys in the feeding study,

but not in the pasture experiments. These increases apparently did not affect the health of the cattle. Muscle, the cattle part most consumed by humans, is protected from uptake of metal sludge contaminants.

We in EPA, plan to continue our research and development activities in the municipal sludge area over the next few years. Our efforts will be concentrated on the technology gaps that interfere with or constrain use or disposal.

Improved methods of treating or processing sludge to prepare it for environmentally sound use or disposal will be tested and evaluated. Digestion offers a great deal of potential for improvement in this regard and as a means of conserving energy at the same time. Other thermal processing techniques such as wet-oxidation have potential beyond current use and new processes to make a useful product, such as inert aggregate for highway construction or a useful oil substitute are being examined.

The research associated with the health aspects of sludge management is currently reaching a point where specific concerns, such as the importance of parasites, may soon be placed in proper perspective.

I again must emphasize that EPA resources are limited. We need the continued support and cooperation of the universities, federal, state and municipal agencies, equipment manufacturers and others. This is the only way that enough talent and other resources can be brought to bear on this difficult environmental problem area. My colleagues and I see this workshop as one of the ways we can obtain high quality advice and guidance.

Experience in sludge issues since 1977 indicates that an EPA policy for sludge management is going to be necessary if we are to deal with the issues in a workable and environmentally sound manner. The Administrator therefore, established an Agency Sludge Policy Committee in October 1981 to address this problem. The Committee includes the Assistant Administrator for Water, Associate Administrator for Solid Wastes and Emergency Response and the Associate Administrator for Policy and Resource Management. A Sludge Task Force provides the technical support for the Committee. The Task Force is very active now and is getting good cooperation and assistance from EPA offices and regions. It is also obtaining inputs from individuals and organizations outside of EPA who are experts in sludge management issues and technology. The Task Force expects to complete its work with a report to the Policy Committee by this fall. Areas to be addressed include:

- Review of Sampling and Analytical Procedures
- Institutional/Legal Background
- Costs of Sludge Disposal and Reuse Options

Exposure and effects of contaminants of concern will be assessed, as will risks, costs, and benefits. An evaluation of regulatory and economic impacts will be a final step, leading to sludge management guidelines and establishment of EPA sludge management policy. In addition, there will most likely be recommendations for follow-up actions, such as additional research needed.

This workshop is intended to contribute to the Task Force efforts, as well as give us in the research area some sound guidance concerning the critical technology and sciences areas needing additional study or development efforts.

In closing, I want to express my appreciation for your assistance to EPA and other federal agencies in resolving the complex issues and technical problems surrounding the use of the land for waste management. Wastewaters and sludge must be applied to the land as a principal management method, but only in an environmentally acceptable manner. Your assistance will be invaluable in reaching a workable approach. I know that you will work to make the next two days as productive and meaningful as the 1973 workshop. I offer you my encouragement toward that end.

NECESSITY OF REGIONAL RESEARCH

Raymond J. Miller
Dean and Director, College of Agriculture
University of Idaho
Moscow, Idaho

Any program that deals with land, particularly agricultural land and what happens when wastewater or sludge is added to those lands is going to involve agricultural researchers.

Over the last year or so you have probably read or seen several reports on the strengths and weaknesses of agricultural research. I do not recall any of these reports talking about one of the real strengths of the agricultural research system in this country. That is regional research. Each state agricultural experiment station, in the Land Grant Universities receives funds from U.S.D.A. These come to the states by formula through the Cooperative State Research Service. Of these funds at least 25% must be spent on Regional Research. This requirement and the resulting cooperation are unique to CSRS, the State Agriculture Experiment Station, and to this country. These few dollars result in a planning research process and system that results in programs and accomplishments that could be obtained in few if any other way. In addition it brings in scientists from other agencies, such as ARS.

Let me use one example, granted an excellent example of a regional project to explain the benefits. That is W-124, *Optimum Utilization of Sewage Sludge on Agricultural Land.*

This project started 5 years ago in the 12 western states. Since the objectives were similar to those of a project in the North Central region, NC-118, the two projects were combined into one, W-124. Subsequently, the membership was expanded to include the Northeast and the South. So it is really a national project. This project also includes scientists from ARS, EPA, and SCS.

One of the objectives of the project was to obtain data that would provide some uniformity and therefore comparability across the U.S. This project put in place a program that used a common sludge, Chicago, which

was supplied by EPA and Chicago, applied at standard rates in 14 states, using a common crop - barley. Five years data will be collected. Data from 16 soil types and for 5 heavy metals (cadmium, copper, zinc, lead, and nickel) and their concentration in the foliage and grain will be obtained. In addition, they will have data on nitrogen and phosphorus availability, nitrogen mineralization and carbon cycling.

This data will be published as a regional publication, making available a large data bank which can be used as a standard across geophysical and geoclimatic zones in the U.S. In addition, appropriate parts of the information will be published in professional journals.

Where else could you put together such a comprehensive program to obtain so much data, and do it with a minimum of overhead. A relatively small number of dollars in the right system have generated a large, well planned and coordinated program. This project has brought together one of the best groups of soil chemists in this country.

An excellent system - regional research.

POSITION PAPERS

INSTITUTIONAL CONSTRAINTS

INSTITUTIONS CONSTRAINING THE UTILIZATION OF MUNICIPAL WASTEWATER AND SLUDGE ON LAND

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ABSTRACT

The variety of social and regulatory constraints on land application programs identified in the literature is reviewed in this paper. Also summarized are the results of a survey of communities that have mounted such programs. Only a small minority of the surveyed communities identify institutional factors as being responsible for seriously limiting the expansion of their respective programs.

INTRODUCTION

Among the conclusions reached at the 1973 Conference on Recycling Municipal Sludges and Effluents on land was that "unless political and institutional constraints on the land application of effluents and sludges are recognized, identified, and resolved, (land application) projects will likely fail, regardless of their technical, scientific, and economic feasibility" (NASULGC, 1973, p. 232). A variety of potential social, legal, and regulatory constraints were identified. Among them were unfavorable public attitudes toward landspreading activities, a variety of local and state regulations that apply to those activities, as well as the sometimes ambiguous nature of the legal regime within which sludge and effluent landspreaders must function.

The past decade's experience has served to emphasize that social, legal, and regulatory environments are of primary concern to municipalities contemplating a land application program. All of us can think of recent instances where a technically sound program had to be scuttled because it could not overcome the opposition of a county health board, a township council, a state environmental protection agency, or a group of concerned private citizens.

The past decade was also a period of time when the incentives to initiate or expand land application activities grew. Federal laws encouraged its use. Changes in environmental regulations limited the use of other sludge and effluent disposal options. Skyrocketing energy prices reduced the economic attractiveness of incineration and other methods. At the same time, fertilizer price increases and the rising cost of water (both of which were caused largely by energy price hikes) heightened the interest of potential participants in land application programs.

This changing structure of incentives has enhanced the interest of communities in the social, legal, and regulatory constraints to land application. Recent literature has established the types of institutional constraints that have been encountered by communities. Section II of this paper uses that literature to define the concept of "institution" and to identify a taxonomy of institutional constraints. In Section III, an attempt is made to document effects of some of the past decade's institutional changes. Also identified are the likely impacts of federal regulation of land application and of the effectiveness of campaigns designed to win public support for landspreading. Finally in the fourth section, some implications for the design of both landspreading programs and public policy dealing with those programs are identified. Also discussed there are the needs for additional study of the institutions constraining the utilization of wastewater and sludge on land.

INSTITUTIONS FACING LAND APPLICATION

An institution is defined as a collective action in control of individual action (Commons, 1934). Institutions include family, economic sphere (e.g. union or business), bureaucracy, and other social groupings and the behavioral norms of these groupings (Gordon, 1980). In a sense, attitudes and the more formal rules of society are a product of social institutions to which individuals conform.

Institutions tend to be static, inherited from the past, dictatorial, and creatures of habit. They inhibit change. However, new institutions evolve as society re-evaluates its rules and norms in light of new technology (Gordon, 1980).

Attitudes and formal rules toward land application of municipal wastewater and sludges are the institutions dictating the use of this technology. These institutions tend to be inherited, governed by habit, dictatorial, and difficult to change. However, land application technology has changed and continues to change. It forces changes in these static institutions. Thus, the institutions we describe today will be

replaced tomorrow with new ones. Whether they will be more favorable toward land application remains to be seen. That will be determined by those who lead society in its evaluation of this changing technology.

Constraining Attitudes

1. Concern about health risks including pathogens, parasites, metals in food chain: Probably the most constraining attitude is that land application of wastewater or sludge adversely affects public health. Concern is expressed over cadmium, lead, and some organic compounds entering the food chain or being directly injested. The possibility of pathogenic organisms and parasitic ova being distributed by land application also is discomforting.

The available literature suggests that little hazard confronts sewage plant workers. No demonstrated hazard has been documented for communities adjacent to well managed land application systems (Burge and Marsh, 1978; Uiga and Crites, 1980). Yet the public intuitively suspects that land application systems are unsafe. Public opposition is usually the strongest when health risks are perceived to be present. Since most people are risk averse, opponents of land application can often thwart programs by rousing the community with the idea that land application might be unhealthy. If the safety of land application is suspect, there is little chance of gaining public support regardless of its other merits (Bruvold, 1972).

2. Concern about the environment: Concentrations of zinc, copper, nickel and industrial organic compounds may be present in wastewater and sludge. There is the fear that these could accumulate in the soil and become toxic to crops. High salinity in some wastewater may also present long run problems to arid soils. Excessive applications of wastewater nitrogen may cause leaching of nitrogen to underground water supplies. Land application near streams or lakes may result in runoff of nutrients, metals, and organics to surface water supplies.

Well managed systems can avoid these problems. Low application rates and proper placement of wastewater and sludges can assure that land application does not threaten the environment. However, the public may perceive potential environmental damage, and that perception can disrupt the most environmentally safe land application program.

3. Concern about nuisances: Odor is the primary concern. Sludge and wastewater are considered to be foul-smelling and capable of threatening the surrounding community with an odor nuisance. Other

potential nuisances are the damage to roads by trucks and equipment, sludge spills, and the perception of the neighborhood as a "dumping ground."

4. Concern about land values: Threats to the community's health and environment and the threat of nuisances would disrupt the land market. Residential and commercial land prices would plummet. The wealth of most land owners in the community would diminish. Opponents of land application can easily paint a grim picture of economic ruin for the community due to the threat of nuisances, health risks, and environmental damage. Only the possible threat need exist for the most safe, well managed land application to be met with stiff resistance from the community.

5. Historic precedents which shape attitudes: From the 1840's to the 1900's, fears of disease and epidemics pushed communities to adopt waste treatment technologies. Land treatment was the initial technology adopted and was responsible for sharply reducing death rates (Jewell and Seabrook, 1979). In Europe, a trend away from land treatment developed in the late 1800's and early 1900's. Expanding towns put increased loads on land treatment systems, and populations moved closer to the "sewage farms." Sewage treatment technologies such as sedimentation, chemical precipitation, and screening were developed for land application systems. Later these technologies were combined with percolating filter and activated sludge methods to replace land treatment.

Europe, in the last part of the 1800's adopted a philosophy of partial treatment of effluents rather than the complete treatment offered by land treatment. In the United States, this philosophy was soon widely accepted due to the distribution of the population and availability of large quantities of water near most cities. By the 1920's in the United States, land treatment had been replaced by other treatment methods which emphasized partial treatment of effluents (Jewell and Seabrook, 1979).

Today, land treatment of wastewater comprises only a small proportion of the municipal waste treatment. In 1975, total municipal wastewater discharges in the U.S. amounted to 90.8 x 10 m^3/day (24 billion gallons per day). Wastewater treatment totaled only about 0.7 billion gallons per day (Asano and Madancy, 1982). Land application of sludge is more widely accepted than is wastewater treatment. Still in 1979, only 31 percent of the nation's municipal sludge was landspread. Incineration and landfilling stabilized sludge accounted for 22 and 29 percent, respectively. Ocean disposal comprised about 12 percent of the total (Walker, 1979).

Given that attitudes tend to be creatures of habit and inherited from the past, it should be no surprise that the attitudes of the public sanitary engineers and public officials are less than favorable toward land application.

6. Cultural influences: There are general attitudes regarding land application which prevail over nearly any community. Women are less accepting of land treatment than are men (Musselman, et al., 1980; Olson and Brunold, 1982). Due to women's traditional role as the one in the family responsible for the family's health and cleanliness, it is not surprising that they are more reluctant than men to accept a practice with possible health risks.

Age is negatively correlated with acceptance (Musselman, et al., 1980; Olson and Bruvold, 1982). Most new technologies are accepted less readily by the old, and land application is no exception.

Formal education is positively correlated to acceptance (Olson and Bruvold, 1982). Especially when a person's formal education stops at less than twelve years, land application is more likely to be disfavored. Exposure to new ideas is a product of education, and this exposure generally leads to greater acceptance of new technologies.

Experience or exposure increases acceptance. Those who have experience with land application are much more receptive than those who have no experience. Similarly, those who are exposed to land application concepts through educational programs are more favorable toward it than those with limited exposure (Musselman, et al., 1982).

7. Farmer attitudes: A successful land application program usually requires acceptance by the farm community. The following attitudes are characteristic of farmers and may affect land application programs.

Farmers exhibit a strong inclination toward short run economic gain (Napier, et al., 1980). This is a double edged sword for land application programs. On the one hand, it means that farmers are likely to adopt land application when there are short run economic benefits. On the other hand, it may encourage practices which are not conducive to long run soil productivity. For example, the threat of excessive metal accumulation ten or fifteen years from now might provide little incentive for lower sludge application rates.

Farmers perceive themselves to be stewards of the land (Napier, et al., 1980). While this attitude may be contradictory to the previous one in some situations, farmers consider themselves to be stewards of natural resources for future generations. This attitude can assist in the development of a responsible land application program when measures to assure an environmentally safe program are made known to farmers.

Farmers oppose laws or regulations which impose land use controls. They feel they should have the right to determine how to use their land. Zoning and other land use control options can be expected to be met with stiff opposition (Napier and Mast, 1981).

Farmers strive for individualism. There is a strong urge to adhere to self sufficiency. Also, they are conservative; there is a reverence for traditional values. Finally, farmers tend to hold strongly to an "agrarian ideology" (Rohrer and Douglas, 1969). This agrarian ideology is composed of the proposition that all nonfarm occupations rely on farmers for sustenance; there is no like dependency for farmers. Another proposition of the agrarian ideology is that farm or rural life is "better" or "more natural" than urban life.

These attitudes - individualism, conservatism, an agrarian ideology - lead to the often expressed view that municipal waste is an urban problem. The problem ought to be solved in the city by those creating the waste and not by pushing it off on rural communities. Opponents to land application can successively impede a well designed program by turning these attitudes against land application.

8. Municipal officials' attitudes: There is a bias against land application on the part of those responsible for waste treatment (Jewell and Seabrook, 1979). Due to the historic evolution of waste treatment, municipal officials and the technical community are unfamiliar with land treatment. For the past half century, waste treatment texts have ignored or only briefly mentioned land application. Waste is considered a disposal problem, not a resource recovery problem.

Also, municipal officials usually underestimate the community's acceptance of land application (Olson and Bruvold, 1982). When a land application system is proposed, officials are more likely to be confronted by those expressing negative views than those in favor. Thus, officials tend to overestimate the opposition to land application.

Regulatory and Legal Constraints

Federal legislation has encouraged land application and generally has refrained from placing constraints on it. The Federal Water Pollution Control Act Amendments of 1972 (PL 92-500) encouraged the recycling of potential sewage pollutants through the production of agricultural, silvicultural, or aquacultural production. It also contained provisions which require consideration of "appropriate alternative waste management techniques," such as land application, in order to receive federal grants. The 1977 Clean Water Act further encouraged the use of "innovative and alternative" technologies such as landspreading.

EPA has encouraged the consideration of land application through its construction grants program. It is in the process of finalizing a set of regulations for land application and is developing proposed regulations to govern the distribution and marketing of sludge. Generally, these proposed regulations are intended to encourage responsible, well-managed land application systems.

However, the lack of final land application regulations may be impeding progress in developing new land application systems. Communities may be using a wait and see attitude until final regulations are issued.

State legislation and regulation is less supportive of landspreading. In nearly every state, at least one form of license or permit is required. Several agencies, each with different regulations, may be involved where land application is thought to affect water quality, food production, and air quality (Deese, et al., 1980). At least thirty-one states have issued guidelines or regulations concerning land application. Most of the remaining states approve land application on a case by case basis (Morris and Jewell, 1977).

State regulations often constrain the land application of wastewater since they usually require the secondary treatment of waste prior to land application. When secondary treatment is required, land application of wastewater can not be cost effective when compared with conventional treatment systems (Morris and Jewell, 1977).

State and local regulations or guidelines may incorporate permit requirements, application rate restrictions, wastewater or sludge analysis requirements, monitoring requirements, maximum concentrations of metals, nitrogen, and organics, and land use restrictions. While these may appear to be constraints, they are usually established as "rules of the game" for responsible, well-managed programs to achieve safe land

application programs. They are intended usually to constrain only those which might detrimentally affect human wealth or the environment.

Other local constraints may include zoning or other land use controls, health codes, and nuisance laws. These can be effectively used by opponents to delay or completely block the best planned land application program. To assure a successful program, local public officials' attitudes must be assessed early. Their attitudes, like those of the public at large, may need to be changed before a program is viable.

The method of land acquisition may provide constraints to the land application program (Christensen, 1982). Land may be purchased and held in fee simple ownership by a community. While this insures a long term planning horizon, it usually fosters local opposition. The community purchasing a "sludge farm" is likely to raise the ire of neighboring land owners. Also government purchase can result in the local property tax base being reduced and the community's land owners having a heavier property tax burden (Walker, 1979). Another method of acquiring land is by purchasing an easement or a portion of the bundle of ownership rights. Communities might purchase the right to use land for sludge application during certain months of the year, for example (Christensen, 1982). The use of easements might provide a long run planning horizon needed for the community, yet leave the land in private ownership. However, the temporary use of privately owned land is the most common method of land acquisition. Some communities pay land owners for the temporary use of land, but most are allowed to land apply sludge or effluent without charge.

While the land acquisition method has a bearing on the success of a land application program, it is not crucial. More important is the working relationship between the community and rural residents and landowners (Christensen, 1982). Antagonism between these parties can destroy a program even though tight legal provisions have been established.

Finally, management of the land application site may present constraints to program acceptance. Poorly trained operators, odorous conditions, high application rates, and little information about sludge or wastewater content almost guarantee problems. Transporting from one legal jurisdiction to another multiplies potential legal constraints and public acceptance problems. Use of sludge and wastewater near residential areas is likely to generate opposition. Public opposition is highly correlated with land use intensity near land application sites (Deese, et al., 1980; Olson and Bruvold, 1982).

Lessons About Changing Institutions

Communities can influence public acceptance, but they can not control it. Even the most well planned program may face too many institutional rigidities to succeed. The following lessons can help, not guarantee, the break down of institutional barriers by changing attitudes.

The public needs to be involved in the decision making process. In a program's early stages, ideas need to be developed, the attitudes and potential support of affected groups assessed, and rapport with influential individuals established (Donnermeyer, 1977; Dotson, 1982; Ellis and Disinger, 1981; Stitzlein, 1980). The public is then involved in making decisions concerning wastewater or sludge use. Problems are discussed, information is disseminated, alternatives are developed and analyzed and the public participates in decision making. During this process technical and economic data need to be communicated effectively and understandably to the lay public. Avoided is the "public hearing" model which takes the form of a "we-they" winner-take-all battle between wastewater treatment officials and rural residents. Rather, seminars and workshops are scheduled with several small groups to allow favorable group dynamics to operate (Ellis and Disinger, 1981).

Incentives need to be clarified continually. The economic justification of a land application program needs to be emphasized. It is probable that agriculture benefits from receiving wastewater or sludge, and the community benefits from the relatively low cost of land treatment. The size of these benefits needs to be assessed and made public early (Deese, et al., 1980).

Odor, health and environmental issues need to be addressed early and with vigor. Treatment plants must resolve odor problems or a land application program likely is doomed. Demonstration sites can be used to show the lack of odor. Experts from the agricultural community need to assess the proposed program and identify the soils, application rates, and wastewater or sludge contents which provide an environmentally safe program. Public health officials need to evaluate the program and assure the public that the program will not be detrimental to local health or to the food chain.

Advisory groups can be established to facilitate public involvement. Roles which these groups might fill include helping pick suitable sites, bringing farmers and rural residents into planning, promoting the idea that wastewater or sludge is a resource out of place, pointing out local problems or probable constraints, and scrutinizing the programs' technical and management aspects.

An education program can remove much of the communities opposition (Deese, et al., 1980; Dotson, 1982; Ellis and Disinger, 1981; Musselman, et al., 1980). Complete and unbiased information needs to be made available through a wide variety of techniques. Forums for public education might include seminars and workshops with concerned citizens, lectures to community groups, tours of the wastewater treatment plant and land application sites, newspaper articles, television and radio interviews. Written material, varying in scope from brief brochures to university publications, should be made available. The U.S. EPA, state agencies, and many land grant colleges have this written material available. Also, available from these sources are films and tapes concerning land application.

Management of the land application program needs to constantly focus on winning and maintaining public acceptance. Land application at sites removed from residential areas; sludge, wastewater, and soil analysis prior to application; monitoring sites often after application; well kept records; low application rates; soil incorporation of odorous sludges; responsible vehicle operators; clean vehicles; and cleaning up application sites and roads are some of the practices which can help win public acceptance.

Written agreements between the community and landowners should be used when private landowners are involved. The purpose of these agreements is to minimize future misunderstandings. Application rates, sludge or wastewater quality, method and time of application, and legal liability should be clarified.

THE INSTITUTIONAL ENVIRONMENT CONFRONTING A SAMPLE OF LANDSPREADING COMMUNITIES

During the past ten years, rising energy prices, increasing water scarcity, and certain changes in the regulatory environment promoted increased reliance on land application as a sludge and effluent management technique. If anything, these changes have increased the importance of institutional constraints on the growth of land application programs. In order to gain a preliminary idea of the role played by those institutional constraints during the past decade, a mail survey of communities that now have a land application program, have had such a program, or have expressed an interest in mounting such a program was conducted. Some results of that survey are summarized in this section.

During the latter part of 1982, the authors of this paper collected a sample of 177 U.S. communities. Sources contacted during this sample-building process include regulatory officials, cooperative extension personnel, and others who are knowledgeable about land application in

their respective states or regions. Questionnaires to be used in the survey were reviewed by university researchers and municipal officials in the state of Ohio. The initial mailing of the questionnaires occurred in November, 1982. After a follow-up mailing about three weeks later, 110 questionnaires (equal to 62 percent of the original sample) had been returned.

The sample includes a large number of communities from states such as Ohio, Michigan, and Illinois, where sludge landspreading is more common, as well as quite a few cities and towns from California, where land application of effluent is widespread. In addition, one-half of the communities had a 1982 population of less than 25,000; over four-fifths of them had populations of less than 100,000 in 1982. One might keep in mind these two characteristics of the sample when reviewing the results reported below. For example, a midwestern or southwestern town located in an agricultural area might have an experience with local institutional constraints that is vastly different from the local institutional constraints facing a land application community in a more densely populated, less agricultural region, such as New England.

Besides collecting basic information on community size, wastewater treatment and sludge management practices, etc., the survey instrument was designed to answer the following questions:

- What have been the trends in state and local regulation of land application during the past decade?
- Have institutional constraints played a large role in limiting the expansion of land application programs?
- What are municipalities' perceptions of the impacts of future institutional change?
- What role do public relations campaigns play in promoting the success of a land application program?

Changes in Legal and Regulatory Constraints

It is fair to say that major changes in the legal environment for landspreaders occurred during the past decade. Local regulations (e.g., permit requirements, limitations on sites and applicaion rates, etc.) were practically non-existent in 1972. Only 10 percent of the sample recalls encountering such regulations ten years ago. By 1982, though, over 60 percent of the communities reported that their own or neighboring jurisdictions had adopted regulations. This percentage is higher in the northeast and in California. It is interesting to note, however, that local prohibitions have been encountered by relatively few of the communities in our sample. Evidently, even where local communities feel

strongly enough about land application to consider regulating it, they are still willing to accept a well-managed land application program.

State regulation of land application programs were encountered by just 40 percent of the communities in 1972. By 1982, however, more than four-fifths of the surveyed communities reported that state law compelled them to obtain permits from the state and to conform to land application guidelines established by state agencies. Thus, the states are now the major sources of rules and regulations covering land application.

Effects of Regulatory Change

Establishing that a major change in legal institutions occurred between 1972 and 1982 does not necessarily prove that it has greatly constrained the expansion of land application. Economic concerns seem to have motivated communities to move forward with a land application program. Our sample was asked to rate the importance of a variety of factors affecting the decision to initiate or to expand such a program. The relatively high cost of other sludge disposal methods was identified as the most important of these factors.

By and large, the communities that had expanded their programs reported that most groups (owners of land application sites, neighbors of those sites, environmental groups, government officials, etc.) were either supportive or, at the very least, neutral about that expansion. Indeed at first glance, neither social nor legal constraints seem to have played a major role in many communities' decisions to downscale or to avoid increasing land application. (Approximately 25 municipalities responding to our survey reported making such a decision.) High sludge/effluent transportation costs and lack of land application sites were the reasons given most often for such a decision.

Let us note, however, that there might be institutional constraints underlying these two "technical" or "economic" reasons. For example, local opposition might have forced some communities to rely on more distant application sites.

Impacts of Future Institutional Change

We have noted that state governments currently set most of the rules governing land application. Increased federal regulation has been

debated for several years. We were interested in eliciting the municipalities assessment of the likely impacts resulting from federal regulation.

Our survey indicates that knowledge of proposed federal standards is highly imperfect. However, the answers to our questions suggest that community officials disagree with those who argue that the existence of federal standards would greatly ease the task of winning public acceptance for a land application program. Also, whereas relatively few officials fear that promulgation of federal regulations would put their landspreading programs out of business, the great majority of them doubt that those regulations can be designed with enough flexibility to take special local conditions into account. On balance, then, one cannot infer from our survey results that there is strong support among landspreading communities for federal regulation of land application.

Role of a Public Relations Campaign

Over time, a "conventional wisdom" has stressed the need for communities to mount an educational effort in order to win public acceptance for landspreading programs. It might be interesting to note that only slightly more than one-half of the communities expanding their landspreading programs report the use of an educational campaign. Of course, this survey result might be somewhat misleading in that one community can benefit from some other community's or agency's public relations efforts. Among the more effective educational methods identified by the operators was field demonstration plots.

Summary of Survey Results

This survey supplies only preliminary answers to questions dealing with landspreaders' institutional environments and how those environments can be or might be changed. However, one survey result suggests that institutional constraints have not proven to be a significant barrier to most communities wishing to expand land application. In 1972, practically all surveyed communities applied their sludge and effluent to idle land owned by the public sector. Most communities reported that, in 1982, better than half of their sludge and effluent was spread on privately-owned land and that the greater part of acreage used in land application programs was pasture or cropland. This result might be due to a lack of publicly-owned land to support expanded land application programs. More likely is that public attitudes toward this disposal option have changed markedly, thereby allowing for expansion of well-managed land application programs.

IMPLICATIONS

Land application of wastewater and sludge has expanded over the past ten years due to technological and economic forces. Further expansion is expected. Institutional constraints have increased with proliferating federal, state, and local government intervention.

But institutions tend to be static and creatures of habit. As a technology begins to be adopted, institutional constraints are expected. These institutional constraints include public attitudes and a society's formal rules. Public attitudes constraining land application are concerns about health risks, the environment, nuisances, and land values. These attitudes are shaped by historic precedents and cultural influences. Farmers' and municipal officials' attitudes also are constraining.

Society's formal rules include federal, state, and local regulations. While federal legislation has encouraged land application, some state and local regulations have hampered it. Land use controls, health codes, and nuisance laws have been used at the local level by opponents to frustrate land application. Some land acquisition and management strategies have negatively affected land application in communities.

The existence of these institutional barriers inhibit the growth of land application. Institutions frustrate efforts to change and can be expected to construct barriers to change. An important issue is how to build public acceptance that will accomodate an emerging technology like land application. A host of methods are available to build public acceptance. They include public involvement in the decision making process, clarification of incentives for land application, resolution of odor, health, and environmental problems, establishment of advisory groups, aggressive educational programs, and responsible management of land application,

Our survey results indicate that landspreading communities have witnessed substantial growth in state and local regulations during the past decade. In spite of (or possibly because of) this expanded regulatory framework, communities have been able to respond to the economic incentives that are favorable to land application. The public has been generally supportive of this treatment method. The majority of communities have used educational or public relations campaigns to enhance this support. The legal and regulatory frame-work has not been perceived as limiting. However, there is no strong support for new federal regulations.

Future research in this area should center around the question of how acceptance of a safe, well-managed land application program can be better assured. How can programs be designed to win public acceptance? The difficulty of this research is that cultural influences, historic precedents, and public concerns change from community to community. Thus, programs designed to win public acceptance differ across the country.

Other future research should address the formal rules and regulations being adopted. Are state and local regulations sufficient to assure safe land application programs? What rules and regulations could be designed to encourage safe, well-managed programs? How can these rules and regulations be made flexible to accommodate special local conditions and future change?

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QUESTIONS AND COMMENTS FROM THE FLOOR:

Susan Young, INCOG, Tulsa, OK: Have you developed any feel for public attitudes toward state regulators and regulations? U.S. Federal regulators and regulations? (e.g, More confidence in state agencies or federal agencies?)

D. Lynn Forster: The confidence in local, state or federal regulators dependes directly on how much credibility these regulators have with the public. The public is reluctant to grant credibility when regulators have positive attitudes toward new technologies (e.g., landspreading) which promise change. Probably the safest methods for regulators to win public confidence is to support the status quo, to maintain credibility by trying to avoid change. Of course this incentive to avoid change is the problem with any regulatory approach. Regulators supportive of change find gaining credibility to be an uphill battle.

Betty H. Olson, University of California, Irvine: 1. What would be the drawbacks of not having consulted the directors of the controlling agencies in the results of the survey?

Lynn Forster: Local community decision makers were included in the sample. We felt they were likely to have the best perspective on their local situations.

Betty H. Olson: 2. What is the single most important factor in determining acceptance of land application?

Lynn Forster: From the community's perspective, economics is the single most important factor. For the public at large, concern about health risks seems to be the most often expressed. Of ourse, this concern, as well as other concerns, may be a guise for concern about the impacts of land application on property values.

Betty H. Olson: 3. How do you deal with influential individuals who are elected and very much controlled by the electorate?

Lynn Forster: These individuals must be brought into the decision making process early. They should be involved as problems are discussed, information is disseminated, and alternatives are developed.

Ebba Lund, Royal Veterinary and Agricultural University, Copenhagen, Denmark: In your very interesting presentation, it seems that you left out the aspect that land application is not a new idea. For instance, the French literature published more than a century ago, contain the same problems and discussions. Many European countries have had systematic application programs for a century. It seems that the important new factors are the population growth and the difficulties in finding suitable application sites. In Denmark, in spite of careful studies, it is not known what happens to half of the sludge produced. How is the U.S. situation? It seems important when we consider further land application to know when sludges are now being applied. Perhaps without being registered.

Lynn Forster: In many communities, land application programs have been mismanaged. Application rates have been excessive. Sites have been poorly chosen. Recordkeeping has been nonexistent. In some communities, public giveaway programs have been an important disposal methods. In these programs, control over sludge use is completely lost. You are correct in saying that in many communities there has been incomplete knowledge of where sludge has gone.

Merry L. Morris, New Jersey Department of Environmental Protection: Since we are largely dealing with the issues of distrust and credibility (or lack of it), has your study indicated which groups (federal government, state government, county officials, local officials, extension agents, Soil Conservation Service personnel) have credibility? Clearly technical information from a source without credibility will not go far to allay public fears.

Lynn Forster: Our survey did not address the topic of credible sources. However, it is our experience that agricultural institutions are perceived as a credible information source. Personnel from land grant universities, Cooperative Extension Service, and Soil Conservation Service seem to have a high degree of credibility.

Stephen Campbell, BIOGRO Systems, Annapolis, MD: 1. Sludge disposal low bid contract type work creates shortcuts and reduced costs, therefore, utilities or contractors don't do public relations. 2. Public is not being properly educated - engineers and academians talk over heads of average concerned citizens.

Kenneth Dotson, EPA,-MERL, Cincinnati, OH: Who educates the public is more important than the message or program in gaining public acceptance.

Donald B. Aulenbach, Rensselaer Polytechnic Institute: EPA has pushed sewers and conventional treatment. Now, it is difficult to re-indoctrinate the public into the use of land application where applicable.

Regarding nuisance odors from land application, they are frequently much less objectionable than the odors from nearby farms having hogs (unless you are used to the odor of hogs).

Frank Humenik, North Carolina State University: Your list of constraining attitudes was excellent in serving to stimulate recognition of problems faced in getting community acceptance. However, I suggest that instead of health and environment being the primary concerns it is really nuisance and land values which you listed as No. 3 and No. 4. Then, because of preoccupation with these understandable personal interests, either knowing or unknowing, strong arguments are made on an historical and cultural precedent as you noted in No. 5 and No.6. If this is the case, then it is most important to simply address nuisance and land value concern and not overwhelm individuals or public hearings with technical facts on health and environmental protection. Recognition of the real issues, whether openly addressed or admitted, can allow more meaningful public interaction and reduced tension. Sometimes the perception that data on health and environment are neded when the real issue is nuisance and land values causes discussions to get on different tracks and then emotionalism and frustration often results. It appears that another point is to emphasize that the relative risk-benefits of alternatives must be judged on an equal basis. The issue is not whether waste is to handled but what is the best alternative and these alternatives must be evaluated on the same basis, either zero tolerance or a given risk-benefit.

Jack L. Cooper, National Food Processors Association, Washington, D.C.: I complement the preparers and presenters for the excellent reports, and summarize my concerns as follows:

1. *It is the perception of safety that is the major concern of food processors. No matter how safe a particular disposal operation may be, if it is portrayed as unsafe by the press and if the public at large perceives it as being unsafe, food processors accepting crops grown in sludge-amended soils could have their reputations ruined.*

2. *Public involvement in decision making is very important. In discussions with the public, special consideration should be given to concerns of consumers of crops, particularly food processors.*

3. *Enforceable agreements are needed to assign legal responsibility for actions and consequences of improper action.*

4. *Records of sludge, soil and crop testing, application rates and sites, crops grown, etc., must be kept and made available to the public.*

ENGINEERING AND ECONOMICS

ENGINEERING AND ECONOMICS OF APPLYING WASTEWATER AND SLUDGE TO LAND

Charles E. Pound, Douglas A. Griffes, Ronald W. Crites

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BACKGROUND

Land application of wastewater has been practiced for centuries wherever waterborne waste transport systems were developed. Farming with wastewater is the logical extension of the practice, thereby resulting in a beneficial use of the water and nutrients. The practice was introduced into the United States more than 100 years ago. General interest in the subject waned after the turn of the century in favor of more mechanized methods of wastewater treatment (1).

Interest in land application of wastewater, as reflected by the number of references published, began to return in the 1960's (2). As the scientific and engineering community again picked up the subject they asked questions for which there were no documented answers. Studies were initiated to answer some of the questions and included those conducted by the faculties of Pennsylvania State University, University of Arizona, University of California and others (3). Also, agencies such as the U.S. Army Corps of Engineers (USACE), U.S. Department of Agriculture (USDA) and the Federal Water Pollution Control Administration (FWPCA) (forerunner of the U.S. Environmental Protection Agency, USEPA) became interested in conducting or sponsoring investigations into these subjects.

Concurrently, the practice of applying sludge derived from municipal wastewater to land also was questioned. It seemed logical that if the relatively few contaminants remaining in the wastewater after the solid matter or sludge has been removed can cause problems, surely the more concentrated sludge must at least be suspect. Therefore, investigations into the land disposal of sludge also increased in the 1960's.

The Clean Water Act Amendments of 1972 (P.L. 92-500) brought with it a need to develop alternative technologies for treating and disposing of wastewaters and sludges in an economical and environmentally sound manner. This act further encouraged investigative efforts into the land application of wastewater and sludges.

It was with this background that the 1973 Research Needs Workshop was held. Statements of current knowledge and areas of needed technical information were developed that could be used as a guide for establishing design constraints and for directing future research activities (4). Over the next 10 years a great number of research activities were conducted to fill the data and information needs outlined during the Workshop.

With the advent of the energy crises, came a change in conclusions about the most economical sludge stabilization processes. The least energy consuming processes quickly became the most interesting. This resulted in the elimination of sludge incineration in most cases. This change, coupled with the replacement of primary treatment plants with secondary treatment plants, the reduction of marine sludge disposal options, and the concern over sludge give-away programs for private use contributed to ever increasing quantities of sludge.

The following paragraphs present an assessment of the needs at the practical engineering community and the adequacy of the technical tools currently available to meet the challenges presented by a dwindling public works dollar and an increasing volume of wastewater and sludge. To the practical engineer involved in the planning and design of land application systems, it is not only important that specific technical questions be answered, but that information be readily available, concise, and in a useable form. Consequently, both research and informational needs of the engineering community are addressed.

WASTEWATER APPLICATION

Resources available to the community for the planning and design of wastewater application systems have advanced significantly over the past decade. There is now a great deal of practical information available concerning each of the three basic processes:

- Slow Rate (SR)
- Rapid Infiltration (RI)
- Overland Flow (OF)

State of Knowledge

In 1975 and 1976 a series of Technology Transfer seminars were held and design criteria and case studies were presented. One of the first documents developed and published was the "Process Design Manual for Land Treatment of Municipal Wastewater" (5). Published in 1977, this manual was prepared as a joint effort of the USEPA, USACE and USDA. Concurrent with this effort, other comprehensive engineering guidelines were developed by other independent groups. Among the most notable of these was a two volume engineering text prepared at Cornell University (6).

Another important milestone occurred in 1978 with the International Symposium on "State of Knowledge in Land Treatment of Wastewater" in Hanover, New Hampshire (7). This symposium summarized the practical aspects of land treatment and identified suitable approaches for design.

The efforts of the last decade were culminated in the publication of a second edition of the Process Design Manual in 1981 (8). The revision was deemed necessary because of the large amount of research data, design criteria and operating experience made available during the four preceding years. Of particular importance were the results of EPA long-term effects studies (9). The design approach of the manual is essentially empirical, i.e., observation of successful performance followed by derivation of criteria and mathematical expressions that describe overall performance. As a result, the design procedures presented in the manual are expected to be conservative.

Some of the more rational design procedures are presented in the manual. It also describes several currently available mathematical models for evaluating a system's response to a particular constituent or it's overall performance. A more detailed discussion of specific models for land treatment is available (10).

One important engineering subject not covered by the Process Design Manual was that of costs. The first comprehensive reference on this subject was published by the EPA in 1975 (11). This report provided cost curves for the major components of land treatment systems. The information was derived from a combination of:

- Previously published information
- Surveys of then existing facilities
- Consultation with contractors
- Cost calculations based on typical preliminary designs

Although this report was revised in 1979 (12), many of the original cost curves were left unchanged. Several computer programs for cost estimation were developed from this original data (13, 14, 15).

Assessment

An assessment of the information base currently available to the engineering community is given in Table 1. In this table, the availability, completeness, and practicality of information is considered for each of the major planning and design factors. Generally, the information base is considered "adequate" when it:

- Accurately reflects the current research and demonstration data

- Allows the engineer to define acceptable design criteria and to predict performance with a reasonable degree of confidence

Planning Methodology--

Planning is a critical project phase for land treatment systems. The site specific nature of land treatment and the multi-disciplinary approach necessary for planning create special requirements for many typical wastewater agencies and engineering firms. Recognizing this need, planning methodology has been well addressed in the Process Design Manual and other guidelines.

Table 1. Assessment of Existing Engineering Information for Wastewater Application Systems

Factor	Status on Current Design Information		
	Slow Rate	Rapid Infiltration	Overland Flow
Planning Methodology	*Adequate*	Adequate	Adequate
Preapplication Treatment	Mostly Adequate	Adequate	Adequate
Storage Requirements	Adequate	Adequate	Adequate
Loading Rate Determination	Adequate	Mostly Adequate	Adequate
Treatment Performance:			
BOD & SS	Adequate	Adequate	Mostly Adequate
Nitrogen	Mostly Adequate	Limited	Limited
Phosphorus	Mostly Adequate	Limited	Limited
Pathogenic Organisms	Adequate	Mostly Adequate	Adequate
Trace Organics	Limited	Limited	Limited
Distribution	Adequate	Adequate	Adequate
Management of Renovated Water	Adequate	Mostly Adequate	Adequate
Crop Management	Adequate	Adequate	Adequate
Costs	Mostly Adequate	Mostly Adequate	Mostly Adequate

Preapplication Treatment--

Requirements for unnecessarily high levels of preapplication treatment have significantly constrained the implementation of cost-effective land treatment systems in many areas of the country. This problem continues to some degree in spite of the guidelines currently available. Current thinking on appropriate levels of preapplication treatment is summarized in the USEPA 1982 guidance on Facilities Planning (16) and the Process Design Manual.

For slow rate systems a range of preapplication treatment from primary to biological treatment with disinfection has been recommended depending on requirements for:

- Protection of public health as it relates to human consumption of crops or crop byproducts or to direct exposure to applied wastewater
- Prevention of nuisance conditions during storage
- Prevention of operating problems in distribution systems

Of these three factors, public health protection is the most difficult to define. Because of the uncertainties that are often involved and the emotional nature of this issue, conservative standards generally have been applied by State or local public health or water quality control agencies. Continuing system demonstrations will be valuable for further refinement of these standards.

The degree of public contact is much less with rapid infiltration and overland flow systems so preapplication treatment guidelines are related primarily to required system performance and the avoidance of nuisance conditions. Recommended pretreatment levels range from primary to biological treatment for rapid infiltration systems. Nitrogen removal by nitrification-denitrification in the soil can be maximized by limiting preapplication treatment to primary levels (17). For overland systems minimal preapplication treatment, such as screening or comminution, can be acceptable at isolated sites.

Storage Requirements--

There is little design guidance available for determining storage requirements. This limited guidance is aimed primarily at slow rate and overland flow systems. Three computer programs were developed by the National Climatic Center to estimate storage days when inclement weather days preclude operation. Generalized information generated from these program is available (18), or the programs can be run for a specific location using climatic criteria set by the user.

Loading Rates--

Systematic procedures for estimating design loading rates are presented in the Design Manual for all three processes.

For slow rate systems, design loading rates may be limited by either soil permeability or nitrogen concentrations in the groundwater. Insitu testing is the recommended procedure for

determining soil permeability. Hydraulic loadings are based on a monthly water balance prepared for the design climatic conditions, using the estimated soil permeability values. For nitrogen loadings, an annual water and nutrient balance is constructed which accounts for nitrogen concentrations in applied and percolating wastewater, crop uptake, and denitrification and volatilization.

The critical loading rate for rapid infiltration systems is usually limited to the hydraulic capacity of the soil. BOD, nitrogen, and/or suspended solids loadings may be critical in some cases. Ranges of hydraulic loadings are suggested in the Design Manual based on results from different types of field measurements.

For overland flow systems, guidelines were developed which relate the hydraulic loading rate to the level of preapplication treatment and desired treatment performance. Application rates and slope length were found to be design factors of greatest importance.

Treatment Performance--

BOD and Suspended Solids--All three processes are capable of providing excellent treatment performance with respect to BOD and suspended solids. It is only with overland flow systems that some different required performance levels for removal of these constituents will affect design criteria. A strong correlation between detention time of the wastewater on the slope and BOD removals has been well documented (8, 19). Overland flow systems have experienced problems in removing algae from stabilization pond effluents.

Nitrogen--Nitrogen removals and transformations are often a critical design consideration for all land treatment processes. Consequently considerable research effort has been directed in this area during the last ten years.

Slow rate system designs are often limited by the allowable nitrate concentrations in the underlying groundwater. Crop uptake is the predominate nitrogen removal mechanism with denitrification and volatilization of ammonia playing lesser roles. Each of these factors has been addressed in recent design guidelines and sufficient information is now available to predict nitrogen performance reasonably well. Nevertheless, refinement and optimization of this information should continue to be an important objective in the future because of the direct impact on hydraulic loading rates, land area, and system cost. Procedures to estimate dilution of nitrates due to mixing and dispersion in the groundwater also need to be developed.

Nitrification-denitrification is the primary nitrogen removal mechanism for rapid infiltration. Relationships have been demonstrated between nitrogen removal and total organic carbon concentration, infiltration rates, lengths of flooding and drying cycles, soil temperature and pH. Although these relationships are now at least partially understood, more effort is needed to develop criteria for design optimization.

Much the same thing can be said for overland flow systems. Nitrification-denitrification is also the major removal mechanism here, although crop uptake and ammonia volatilization can also be significant. General relationships have been demonstrated for several design and operating parameters, however criteria for design optimization have not yet been established.

Phosphorus--At the application rates typically encountered with slow rate systems, phosphorus removal generally is not a matter of concern. Phosphorus removal can be accounted for in terms of crop uptake and adsorption and precipitation in the soil. Conservative design techniques for determining removal capabilities and the useful life of slow rate systems have been developed (8, 20).

For rapid infiltration systems, phosphorus removal capabilities and predictions of useful life can be more critical because of the lower adsorption capabilities of granular soils and the high hydraulic loadings involved. Long-term precipitation of phosphorus in the soil column is significant but relatively unpredictable mechanism. An empirical model has been developed which shows removal is largely a function of detention time in the soil (21).

Phosphorus removal from overland flow is generally poorer than it is for the other systems because most of the applied wastewater flows over the surface and does not contact the soil matrix and adsorption sites. Enhancement of phosphorus removal has been achieved by aluminum sulfate addition before application of wastewater. Design information is limited and optimization criteria are required.

Pathogenic Organisms--Removal of microorganisms, including bacteria, viruses, parasitic protozoa, and helminths has been a continuing subject of interest and investigation for all three processes. Although with each process some significant questions remain, concern with pathogenic organisms has greatly decreased as performance of these systems has been demonstrated. For slow rate systems, the consensus reached in the development of the Design Manual was that microorganism removal is not normally a limiting factor in the design procedure (8).

Considerable research has been conducted on wastewater aerosols in the last 10 years (8). The reported relationship between wastewater aerosols and incidence of disease in Israel was subjected to further research and discredited (22). The risk to health from wastewater aerosols appears to be low and land treatment systems do not present any more of a health hazard from aerosols than do conventional treatment plants (8, 23, 24).

Microorganism removal is more critical for rapid infiltration system design than for the other two systems because of the granular soils and high hydraulic loading rates typically encountered. Study of virus removal has been limited to date, although indications are that effective reductions can be achieved (25). Further demonstration and study may be required.

Because most of the renovated wastewater from overland flow is collected and discharged to surface waters, the concerns are different than with the other two processes. In general, the removal efficiency of overland flow systems for pathogenic organisms is comparable to that of conventional secondary treatment systems. Demonstrations of microorganism removal have been limited so far, particularly for viruses. Reduction of enteric viruses by approximately 85% was reported in a 1976 study (26).

Trace Organics--Increasing attention has been given recently to the ability of land treatment systems to remove trace organic compounds. Based on results of investigations conducted so far, it appears that all three processes can provide effective removal of trace organics. A model has recently been developed for overland flow systems which relates to trace organic removals to molecular weights, octanol/water partition coefficient, and Henry's constant for each individual compound (27). In general, though, the modeling or trace organic performance is in a very preliminary stage, consequently very little design guidance is currently available. Such guidance is particularly important for communities with large industrial wastewater contributions.

Distribution--

There is an ample amount of guidance available on the distribution techniques for all three processes. To a large extent, this information was developed by adapting conventional agricultural irrigation techniques to wastewater application.

One special consideration for wastewater systems has been distribution design for winter operations. Various techniques have been demonstrated which allow winter operation of forest (slow rate), rapid infiltration, and overland flow (to a limited extent) systems.

Management of Renovated Water--

Management of renovated water can take various forms, including underdrainage of agricultural land, runoff collection for overland flow, and recovery wells for rapid infiltration. As with distribution systems, many of these techniques are conventional ones employed by the agricultural community or for groundwater control. There are special considerations for wastewater application, of which the following stand out:

- Stormwater collection and/or release for overland flow systems
- Using the groundwater aquifer for seasonal storage of irrigation water following rapid infiltration
- Agricultural drainage problems in previously unirrigated areas

Crop Management--

Although many standard agricultural practices are directly applicable, there are some special considerations for crop selection and management of wastewater application systems (primarily slow rate but also overland flow to a lesser extent). The most significant requirements commonly encountered are:

- Maximizing nutrient uptake
- Tolerance of crops to high soil moisture conditions
- The need to schedule applications according to wastewater availability rather than crop need

Another aspect of crop management is the method of ownership and control of the agricultural system. There is considerable guidance available for situations where the wastewater agency controls the entire operation. However there is less guidance available for planning alternative arrangements such as contractual relationships with independent farmers, and leasing arrangements.

Costs--

As mentioned earlier, comprehensive cost information for facilities planners is currently available (12). The methodology and much of the actual cost information is still adequate for most planning purposes. However, the information is somewhat outdated as it was prepared in 1974 and 1975 at a time when there was very little cost data available from recently constructed systems.

Future Needs

Upon consideration of the practical problems faced by the engineering community in the planning and design of wastewater systems, there are certain deficiencies in currently available engineering guidelines which still must be addressed. These include both research needs to fill in specific technical gaps and information needs to synthesize existing knowledge into a practical format. The following is a summary of major future needs.

Information Needs:

1. Updated cost information based on surveys of recently constructed facilities and current unit costs.
2. A better definition of grant procedures for alternative forms of management and control (such as leasing, contract operations, etc.).

Research Needs:

1. Determination of trace organics removal performance and mechanisms (particularly for rapid infiltration).
2. Prediction and optimization of nitrogen removals, particularly for rapid infiltration and overland flow. Prediction of denitrification is the most important need.
3. Prediction and verification of phosphorus removal by rapid infiltration.

4. Optimization of phosphorus removal by overland flow, especially using supplemental techniques.

5. Prediction of algae removal by overland flow.

6. Post closure performance of land treatment sites. Long term fate of pollutants.

7. Refinement of the relationship for rapid infiltration between long term hydraulic loading rates and initial field tests of infiltration rates.

8. Prediction of dilution of percolate with groundwater by mixing and dispersion.

SLUDGE APPLICATION

In the last 10 years the use of sludge on agriculatural land has been an increasingly controversial subject. Less controversial has been the use of sludge on turf, forest, and drastically disturbed land. The state of knowledge on land application of sludge has increased significantly in the past decade. However, many more informational and research needs remain.

State of Knowledge

For discussion purposes, the technical knowledge associated with sludge application to land was divided into seven areas:

1. Sludge Properties
2. Planning and Site Selection
3. Application Methods
4. Trace Metal Loadings
5. Nitrogen Decay Rates
6. Stabilization and Disinfection
7. Dedicated Land Application

A significant amount of information has been generated for the first three areas. This information is considered as adequate for purposes of design.

In 1978 EPA sponsored a technology transfer publication on Sludge Treatment and Disposal that included a section on Principles and Design Criteria for Sewage Sludge Application on Land (28). That document contains information on planning and site selection, process design, facilities design, management, monitoring and costs. The emphasis was on recycling of sludge to land in a useful or beneficial manner. A discussion of planning and design for disposal of sludge on land was included in one chapter of the EPA Process Design Manual, Sludge Treatment and Disposal (29). In this case the emphasis was on maximizing the placement or application of sludge on lands without consideration of beneficial use.

In 1983 EPA plans to publish a Process Design Manual on Utilization of Municipal Sewage Sludge on Land (30). In addition to the subjects covered in the 1978 EPA publication the new design manual will have chapters on application to drastically disturbed lands and dedicated land application.

In the remaining four areas the state of knowledge for design of sludge application systems also has advanced. Nevertheless, the mechanisms and processes involved are only partially understood.

Sludge Properties--

Recognition of the importance of nitrogen and heavy metal concentrations in design led to increased emphasis on characterizing sludges for these constituents. Phosphorus, potassium and trace organics such as PCB's also are important constituents for agricultural use systems. The presence of toxic organics and the chemistry of organic transformations in sludges requires further research. Data are available in publications (28, 30, and 31) on median concentrations of these constituents in anaerobically and aerobically digested sludge.

Planning and Site Selection--

Site selection criteria have been established for topography, soil permeability, and depth to groundwater. Site identification procedures are available to avoid undesirable soils, geology, hydrologic features and land use.

A usual procedure is to draw concentric circles of increasing radius around the source of the municipal sludge. A preliminary area requirement is calculated by dividing the annual sludge production by a nominal loading rate such as 10 tons/acre/ yr. Starting within the nearest circle, the unsuitable land is excluded and the procedure continues moving progressively outward from the inner circle until sufficient land is located.

Application Methods--

Liquid sludge (1 to 10 percent solids) can be applied by sprinklers, surface irrigation, trucks, or injection equipment. Injection equipment can be either truck-mounted tanks with injection shanks or tractor-pulled devices connected to a sludge pump by an irrigation hose. Many varieties of injection equipment are available commercially.

Dewatered sludge and composted sludge can be applied with a tractor-drawn agricultural manure spreader or similar device mounted on a truck (32, 33). Sludge cake should be allowed to dry on the soil surface after spreading and then be incorporated into the soil (34).

Trace Metal Loadings--

Land application of sludge can add significant quantities of trace metals to the soil. From a phytoxicity standpoint, zinc, copper, and nickel are the metals of concern. From a human health standpoint, cadmium is the metal that has received the most attention (35, 36).

The concern for cadmium arises from estimates that dietary intake of cadmium in the United States is nearly 50 percent of the limit set by the World Health Organization. The level of cadmium in many crops is increased following application of cadmium-laden sludges to soils. The EPA has established annual loading limits for cadmium to avoid excessive accumulation of cadmium in food-chain crops (37, 38).

Crops vary in their uptake of cadmium. Some cultivars (varieties) of crops exclude more cadmium than others. Available data suggest that cadmium uptake can be genetically controlled and that cultivars can be selected for their low uptake and little translocation to edible parts (36).

According to EPA regulations, the pH of the soil-sludge mixture must be maintained at or above 6.5 (38). Recent research with sweet corn showed, however, that kernel cadmium did not increase with a soil pH of 5.7 to 6.5 (39). The authors suggest that the 0.5 kg/ha limit may be unnecessarily restrictive for non-accumulator crops such as sweet corn.

Alternatives to food crops, such as forest, turf, cotton, or biomass production have been investigated (40, 41). Such alternatives are attractive because nitrogen loading is the primary constraint hence much of the current use of sludge is on non-food-chain crops.

Ultimate site capacity often is estimated using a cumulative metal loading (28, 42). Current research tends to indicate that metal loadings are not necessarily cumulative in their effects on crops and that the phytotoxicity levels may be overly conservative (36, 39). The long term effects of sludge application on crop-metal content should be researched.

Nitrogen Decay Rates--

Nitrogen in most sludges occurs primarily in the organic form. The organic nitrogen must "decay" or become biologically transformed into inorganic forms that are available to plants. The rate of decay may range from 15% to 35% per year or more. The difference can be significant in determining the allowable loading rate based on crop nitrogen use.

The new Design Manual (30) prescribes the following decay rates for use in design procedures although these values may vary depending upon site specific conditions:

Type of Sludge	Percent available in the first year
Raw primary or waste activated	40
Aerobically digested	30
Anaerobically digested	15
Composted	10

After the first year the amount of nitrogen decay or mineralization decreases by approximately 50% each year until it stablizes at about 3% of the remaining organic nitrogen. For a given sludge, the lower the decay rate the higher will be the allowable sludge loading rate.

Stabilization and Disinfection--

Considerations of odor control, public health and safety, and ease of handling have led to considerable developments in sludge stabilization and disinfection. Composting is a process that can both stabilize and disinfect sludge (42). The Beltsville aerated-pile method converts undigested sludge into a composted product that is aesthetically acceptable and avoids odor production (43).

Irradiation has been proposed as a disinfection alternative or supplement to anaerobic digestion prior to land application (44). Pathogen reduction can be achieved with liquid, dried, or composted sludge (45).

Dedicated Land Application--

Sludge application rates at dedicated sites are significantly higher than rates for agricultural or forested sites (30). The concept is to provide minimal pretreatment of the sludge and apply at the maximum rate consistent with organic decomposition. In order to oxidize all available nitrogen (nitrification), it has been estimated that a limiting nitrogen application rate would be 3.8 Mg/ha/d which translates to approximately 1.7 dry tons of sludge per acre per day.

Information Needs

Engineering needs that are not normally classified as research needs are discussed in this section. The major needs are for comprehensive, up-to-date information on costs, energy, monitoring and public attitude assessment.

Cost Estimation--

Pretreatment and transportation are generally the largest costs in land application of sludge. Information is available for most pretreatment processes (29, 47) and for transportation (28, 30). However, the costs of land application are less well documented. Capital and operating costs are available in the literature for only a few individual systems. Subsurface injection costs (48), dedicated land application costs (30, 46) and operating costs of landspreading (49) have been reported. What is needed is a comprehensive report on costs of land application including all types of land application techniques and a wide range of sludge quantities.

Energy Requirements--

Data on energy requirements have been assembled (50) but there remain deficiencies in energy conservation assessment. Energy requirements of distribution systems such as injection and spreading have not been presented in a comprehensive manner that includes currently used equipment and a wide range of sludge types and quantities.

Monitoring Requirements--

The design engineer needs to know what monitoring procedures and schedules are realistic for potential environmental effects. A detailed and extensive monitoring program, proposed in the absence of practical guidance, can be very expensive and may not be warranted. Guidance in this area also would be helpful to regulatory agencies. Guidance should include the need and extent of monitoring of soils, groundwater, sludge properties, and crop quality (28).

Public Attitude Assessment--

The subject of land application of sludge has been fraught with controversy about cadmium loadings, risk assessments, and agruments over safety. Such controversy leads the public and many engineers to assume the worst and decide that land application is not an appropriate disposal method. Issues like the Del Monte ban on sludge application (51) and the controversial statements by Dr. Lisk over safety of land application need to be put into perspective (52, 53).

Positive programs such as those in Madison, Wisconsin, (51) and at Columbus, Ohio (52) need to be capsulized and published widely. The Ohio project, conducted by the Ohio Farm Bureau Development Corporation, concluded that the more efficient methods for reaching the public are:

1. Small, easily readable information pamphlets.

2. A film on land application which could be used by community, farm, and interested public groups.

3. A speaker's bureau to provide knowledgeable speakers on land application to community organizations who frequently seek topics for their meetings.

Research Needs

Research and development related to land application of sludge is needed in several areas.

1. Application methods for forestland
2. Long term heavy metal loadings
3. Design criteria for forest application
4. Refinement of nitrogen availability
5. Low cost methods of sludge stabilization and disinfection

Application Methods for Forestland--

Forestland has significant potential for land application, however, the methods of application are relatively limited, especially for dewatered sludge. Liquid sludge can be sprayed on the land, either from moving vehicles or from sprinkler systems. Sprinkler systems using modified nozzles have proven most effective (56). Dewatered sludge is difficult to apply to developed forestland and is generally restricted to initial applications on new plantations. New methods of application could overcome this restriction.

Long Term Heavy Metal Loadings--

Sites where land application has occurred over a number of years have been analyzed (57). Only one site out of 9 produced crops with metal concentrations in the phytotoxic range and the surface soil at that site had a pH of 3.7. More controlled research on the phytotoxic effects of long term heavy metal loadings is necessary. The entire question of cadmium as a measure of public health problems and use as a design guideline should be reviewed, including the basic assumptions implicit in the criteria for daily ingestion by humans, the actual levels of intake, and the amount potentially attributable to crops grown on sludge-amended soils.

Design Criteria for Forest Application--

Application rates to agricultural land are limited either by nitrogen or heavy metal loadings. For forest applications, rates have not developed sufficiently to allow straight-forward process design. Denitrification rates, nitrogen storage in acidic soils, and nitrogen availability rates have not been widely established. Cadmium loading restrictions do not apply and phytotoxicity limits for zinc, copper, and nickel have not been established. Loadings of trace elements such as boron may turn out to be limiting to some tree species.

Refinement of Nitrogen Availability--

There is a need for test methods to determine, for a given sludge and soil, how fast the organic nitrogen will become available to the crop and to soil bacteria. Does soil texture, temperature, or pH affect this rate? Is the current method of relating decay rates to type of pretreatment adequate? Initial organic nitrogen concentration has been correlated with nitrogen availability in recent research (55).

Stabilization--

Low cost stabilization methods are needed specifically for forest, drastically disturbed land, and dedicated land application. Lime stabilization, for example, may be desirable for stabilization of raw sludge prior to application to drastically disturbed land. Pretreatment methods need to be optimized to provide the necessary stabilization and disinfection for the specific application at the least cost. Disinfection is not necessary prior to dedicated land application and may not be required prior to forest or disturbed land application.

CONCLUDING STATEMENT

The last 10 years have brought a dramatic increase in energy costs which affected our comparison of alternative technologies for treatment and disposal of both wastewater and sludges. Assuming energy costs will rise relative to other costs during the next decade, we will see efforts continuing toward development of low energy consuming or even net energy producing technologies. Such technologies may be particularly applicable for sludge management in large population centers. Nevertheless, the use of land application of wastewater and sludges, especially for beneficial reuse of water and nutrients, will continue to offer cost competitive options and play a predominant role in solving our waste management problems.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

Betty H. Olson, University of California, Irvine: How much consideration is being given to making land application of wastewater or sludge application economically feasible at local, state, and federal levels?

Charles Pound: I am not in the best position to answer your question since Metcalf & Eddy is not involved directly in the review of proposed projects. However, since 1977 the EPA has offered economic incentives for land treatment of wastewater and land application of sludge as Innovative and Alternative technologies. On the local and state levels land application must compete as a low cost technology with other waste treatment technologies. Most of these agencies are primarily concerned with protecting the environment and the public health rather than promoting the economics of any given process. Therefore, a conservative approach is usually taken, sometimes leading to the detriment of the economic issues of land application.

Merilyn B. Reeves, League of Women Voters: What is the state of knowledge in regard to level of treatment prior to land treatment? The discussion paper referred to the fact that nitrogen removal can be done with primary treatment, or in the case of overland flow, less than primary. For phosphorus removal or pathogens is secondary treatment necessary? For areas like the Chesapeake Bay the issue of nutrient removal, and its cost, is of prime importance. Is land treatment considered advanced treatment, and is it made too expensive by requiring secondary treatment first?

Charles Pound: To answer your first question first, I believe that we have a reasonably well developed state of knowledge with regard to pretreatment prior to most land application systems. Experience has shown us that acceptable performance in rapid infiltration, overland flow, and surface irrigation systems, can be achieved with primary effluents. However, from a practical standpoint, additional treatment is often necessary to minimize the possibilities of odor production and nuisance conditions. Secondary treatment and disinfection is often beneficial for pathogen removal prior to irrigation particular for systems where there may be some degree of public contact.

On your second question: Please note that conventional secondary treatment will not provide significant amounts of phosphorus removal.

Dr. Merry L. Morris, New Jersey Department of Environmental Protection: What do you include in the category of "monitoring"? The range of monitoring activities or requirements is quite extensive. What do you consider could be case specific and which could be regional or national requirements?

Charles Pound: It is very difficult to develop general monitoring requirements for all land application systems since the requirements will vary considerably from site to site and will depend upon the process employed, the application rates, and other factors. In general, though, the monitoring program for most land application systems should include at least some monitoring of groundwater at the perimeter of the site. Water quality samples should be taken periodically, perhaps as frequently as monthly during the initial period of operation. Later, the frequency can often be reduced to quarterly or less. Typical parameters of concern for most land application systems are nitrogen, TDS, chloride, and possibly total organic carbon (TOC). Case-specific monitoring requirements may include constituents known to be contributed from local industrial sources.

Merry Morris: What do you consider to be the advantages of forest application of wastewater effluents or sludges?

Charles Pound: There are several advantages which can sometimes be realized by the application of wastewater or sludge to forest lands. First, forest lands are often less expensive than other land which may be available. Also, steeper, less desirable areas can be utilized as compared to open land. Quite often the water can be applied year round, including during winter conditions. Finally, forest application systems can often enable one to minimize site disturbance and to isolate the systems from public view and public access.

Charles F. Jelinek, Food and Drug Administration: In general, sludge should not be applied by spraying to food or feed (including forage and fodder) crops. Studies have shown that dried sludge adheres to plants with great tenacity, even through heavy rains. Animals eating such crops can ingest organics and pathogens directly. Many organics bioaccumulate in fatty tissues, and can thus be present in muscle or milk in comparatively high levels. Likewise, organics, toxic metals and pathogens present on food eaten by humans, such as vegetables and fruits, can be consumed directly by the individual.

Robert B. Dean, Lun-Dean Environmental Company, Copenhagen, Denmark: Further to the question of establishing monitoring requirements - engineers would like a complete set of monitoring specifications before they start. However, this is not the most-effective monitoring system. A good quality control program should be used to establish monitoring requirements based on performance. Monitoring of only a few parameters can give essentially all of the useful information at a greatly reduced cost. The problem is that we do not know in advance which will be the critical parameters for a new site. They can only be found out by experience.

Terry J. Logan, Ohio State University: We do not have adequate knowledge of sludge properties, especially sludge metal chemistry and content of organics.

Charles Pound: For planning purposes there is sufficient information in the literature on sludge properties. For design purposes each sludge to be land applied should be characterized for constituents of concern. Municipal sludges should be analyzed for nitrogen, phosphorus, metals, and potentially toxic organics. Characterization for all toxic organics is very expensive and is seldom done in practice. Generally, the extent of organics content determination is limited to PCB's and pesticides.

HYDROLOGIC MANAGEMENT: NUTRIENTS

HYDROLOGIC AND NUTRIENT MANAGEMENT ASPECTS OF MUNICIPAL WASTEWATER AND SLUDGE UTILIZATION ON LAND[1/]

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INTRODUCTION

The biosystem of soil, organic crop residues, and growing crops can effectively renovate municipal wastewater effluents and sewage sludges. Soils and crops have proven to be effective treatment systems in research and in practice. Treatment systems have been designed to handle a wide variety of materials from clear effluents to sludges under various crop, soil, and climatic conditions. The success of these treatment systems is often highly dependent upon decisions during operations. Adequately designed systems can prove ineffective when management decisions fail to make adjustments for changing conditions. A critical management need for the successful operation of most waste renovation systems is the control of the hydrology and hydraulics for water movement at and near the soil surface. Successful renovation of wastes and protection of the environment are often linked to this critically important, biologically-active, and man-controllable zone of the treatment system.

The purpose of this paper is to discuss the hydrologic and nutrient management needs of operational land treatment systems and to give specific examples of management adjustments that have been used to control and improve the renovation of wastes. The term land treatment system covers a wide range of system types, including slow rate, rapid infiltration and overland flow effluent systems, as well as dry and liquid sludge applications for utilization or disposal. Some management considerations are unique to a particular waste renovation

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2/ Research Soil Scientist, Research Chemist, and Research Soil Scientist, respectively, USDA-ARS, St. Paul; and Assistant Professor, Professors, respectively, Dept. of Soil Science, Univ. of ıesota, St. Paul, MN.

system, whereas others can be manifested in general terms to apply to many systems. For example, the necessity to avoid water-logged soils for crop production applies primarily to effluent irrigation systems, but concern for crop uptake of nitrogen (N) applies to almost all waste renovation systems. For this reason some background information on the similarities and differences in systems will be presented before discussing several key problems requiring management consideration.

DESIGN AND OPERATION OF A LAND TREATMENT SYSTEM

There are three important aspects to the design and operation of a waste renovation land treatment system. The first aspect is the type of material to be utilized. There are obviously different requirements for handling sludges than effluents. Type of delivery system, method of application, need for incorporation, and rate of application are largely functions of the type of material. Only small quantities of sludge need to be applied to land in comparison with dilute effluents, for comparable fertilizer value.

The second aspect of the design and operation of a land treatment system involves the soil, geology and climate of the area. Different management is required when faced with shallow or deep soils, slow or high permeability, steep-sloped or level surfaces, humid or arid climate, cold or warm conditions, and crop or forest lands. Effluent should be applied differently to a moderately permeable soil in a cold, semi-humid climate than to a highly permeable soil in a warm, arid climate.

The third aspect for the design and operation of a land treatment system is the economic constraints for the design combined with the philosophy of the designer. Systems have been designed and operated for a final treatment process whereby maximum application rates are desired. Systems have also been designed and operated as direct resource recycling systems wherein either nutrients or water for crop production have dictated lower application rates. The Beltsville sludge trenching system (Sikora et al., 1982) and the Minnesota agricultural watershed (Duncomb et al., 1982) are examples of different systems.

Soils and crops can renovate wastes because they act as a physical filter for the removal of solids, a chemical reaction medium for immobilization of dissolved constituents, and a biological reaction medium for the conversion of constituents by living organisms. The filtering activity is caused by water flow through small openings (pores) which trap large particles. Chemical reactions include precipitate formation, sorption and others. Biological reactions

include for example, nitrification, denitrification and crop uptake. These processes are the core of a soil-crop renovation system. The predominant renovation mechanisms depend upon the material to be renovated, the soils, and climate, and the philosophy of the system. These are the processes that can be influenced by the management of the system. It is also the observation of these processes and their effects on renovation that will dictate the need for management decisions.

Land application of wastewater effluents can be categorized into three systems: slow rate or crop irrigation systems, rapid infiltration or groundwater recharge systems and overland flow systems. Crop irrigation systems such as Penn State (Sopper and Kardos, 1973a), Apple Valley, MN (Clapp et al., 1977b), Muskegon, MI (Ellis et al., 1982) and Michigan State (Tesar et al., 1982) usually involve ordinary agricultural or silvicultural techniques while utilizing effluent as a nutrient or water resource. The final renovation of N from the water primarily involves management of the crop. Other constituents of effluent may also be affected by the crop, but the soil usually acts as the primary physical filter and chemical reaction medium. Management of this type of system primarily needs to provide a favorable environment for crop production (Linden et al., 1978).

Groundwater recharge systems such as Flushing Meadows (Bouwer et al., 1974) involve ordinary hydrologic and hydraulic considerations for maximizing the chemical and physical filtering capacity of the soil. Considerable attention must also be given to the micro-movement for biological renovation within the soil mass. The primary management needs of this type of system are to maintain both a high hydraulic capacity and a favorable environment for biological denitrification.

Overland flow systems such as those at WES (Lee and Peters, 1979) and CRREL (Palazzo et al., 1982) utilize only a small portion of the soil profile, while effluents run over the soil surface through a standing crop. Management considerations involve both the crop (NO_3-N uptake) and the micro-environment near the soil surface for biological denitrification.

Land treatment systems for sewage sludges are designed for either disposal or utilization. There is a wide variety of sludge types, including liquid (less than 5% solids), slurries, filter-cake and dried, and composted materials, which can be applied to land in one of the two systems. A land disposal system such as the Beltsville sludge entrenchment study (Sikora et al., 1982) involves the placement of a maximum amount of material in a minimum amount of space. The primary management consideration is to minimize the migration of materials away

from the disposal site. Sludge utilization land treatment systems such as the Rosemount, MN watershed (Clapp et al., 1977a; Duncomb et al., 1982) and the Fulton County, IL, project (Peterson et al., 1980) involve using the nutrients and organic matter in sludges as a resource for crop production and soil improvement. The primary management considerations of these systems are to incorporate and keep the material in the root zone until it is utilized for crop production.

With this wide diversity in types of effluent and sludge systems in mind, the remainder of this paper will deal with some very specific management considerations and problems for meeting the renovation requirements of a system. Some of these key problems are unique to a specific system, whereas others may be involved in more than one type of system. Specific examples from cases reported in the literature and from our own experiences of more than 12 years of work on land treatment systems will be used.

SPECIFIC PROBLEMS AND MANAGEMENT CONSIDERATIONS

Soil Sealing and Hydraulic Degradation

The effect of the applied material and the method of application on the ability of the soil to conduct water and air is important in slow rate and rapid infiltration effluent treatment systems, as well as liquid sludge treatment systems. Soil pores can be blocked by suspended solids, gas bubbles from chemical and biological reactions, microbial biomass (bacteria, fungi, and algae), and soil solids from compaction, erosion, or reorientation during wetting (especially from rain or sprinkler-drop action), thereby restricting water and air movement. Such restrictions can cause problems in maintaining the designed hydraulic loading rates in rapid infiltration effluent systems, poor soil aeration for crop production, reduced applications of effluent in crop utilization systems, and increased risk of runoff. Fortunately, these problems can be minimized by proper management. Two examples will illustrate how proper management minimized soil sealing.

At the Flushing Meadows rapid infiltration effluent project in Phoenix, AZ. (Bouwer et al., 1974; Rice, 1974), special management techniques were developed in order to minimize the causes and effects of soil sealing. Effluent was allowed to flow overland through a grassed area, to remove suspended solids in the effluent, before ponding in infiltration basins. Thus a major cause of sealing was removed before final land treatment. Even with pretreatment, the soil infiltration capacity decreased with time in the basins, necessitating further management adjustments. Periodic drying cycles, occasional physical disturbances of soil (removal of deposits), and grass cover

crops were necessary in order to restore and maintain high infiltration rates. Drying cycles caused the surface layer to dry and crack, thus reopening the surface. Grass cover promoted root channelization, keeping the soil open and porous.

At the crop irrigation effluent study at Apple Valley, MN (Clapp et al., 1977b; Marten et al., 1979; Marten et al., 1981), low infiltration rates under conventionally-managed corn (Zea mays L.) culture restricted water and nutrient supplies for optimum crop production. Grass or small grain cover-crops prevented sealing and allowed larger and more frequent irrigation with effluent. We found that corn could be grown in rye or perennial grass covers. In both systems strips were tilled in order to partially suppress the growth of the grass and to prepare a good seed bed for the corn. With annual rye no other competition suppression was needed because the rye was maturing as the corn began rapid growth. Some additional herbicidal suppression was required with perennial grass as the cover crop. The grass biomass protected the soil surface from rain and irrigation drop action and promoted the formation of biomass-related porosity of the soil surface layer. The combined cropping system allowed good corn production while maintaining a high infiltration capacity of the soil. The combination grass and corn crop system was a management adjustment made after operations began in order to prevent soil sealing. Such adjustments may be necessary for other land waste treatment systems in order to optimize the performance of the system.

Crop Management

Crops are an integral part of most land treatment systems and require special management. Important management criteria for the successful operation of a land treatment system includes: (1) selection of the best adapted crop, (2) pretreatment of soil with lime and fertilizers, (3) attention to soil drainage and aeration requirements, (4) use of higher seeding rates than normal, (5) allowance of sufficient time for stand establishment, (6) selective or complete reduction of competition from weeds or grasses, and (7) timely harvesting. Some attention has been given to each of these aspects of crop management in waste renovation research and a complete review will not be given here. Crop management needs can be summarized, however, as follows: the manager should attempt to provide the best possible physical, chemical, and biological environment for the best adapted crops that have some economic value. In some cases this may require more management skill than simply controlling the application of the waste material. Several examples from our experiences with wastewater irrigation will help illustrate the need for these special management skills.

During the fall and early spring, following an August seeding of perennial forage species, only light and infrequent irrigations of wastewater were applied (Marten et al., 1979). This practice allowed time for seed germination and seedling establishment in order to insure a good stand of each species. The established stand thus had the best opportunity to withstand heavy and frequent wastewater irrigations begun later in the spring and continued for several years. Differences in persistance of the various species which occurred later was not related to ability of the plants to become established under wet soil conditions. The same practice was followed each year with the annual corn crop (Marten et al., 1980).

Forage harvesting schedules had to be adjusted to take advantage of the regrowth and N uptake characteristics, while not reducing the regrowth potential of the plant (Marten et al., 1979). Such management adjustments require considerable knowledge of the growth characteristics of forage species, and depended on the amount of wastewater applied in our studies (Linden et al., 1981).

Runoff

All land used for agricultural crop production should be managed in such a way as to minimize soil erosion and water runoff. This maxim is particularly true for landscapes receiving wastewater sludges and effluents because relatively greater loss of plant nutrients will reduce the productive potential of a given site, and the risk of pollution of surface waters will be magnified. Obviously, overland flow systems designed for wastewater treatment are special cases where controlled runoff is desirable.

Most sewage sludge constituents that possess a potential for surface water contamination are only sparingly soluble, with the exception of N. In this context, conservation practices and structures that minimize sediment runoff will also greatly reduce the hazard of surface water contamination. Using terracing and conservation tillage practices, we were able to hold sediment losses to a minimum within a watershed that contained slopes of up to 12%. On the areas cropped to continuous corn, sediment losses occurred only during high intensity storms in the spring and early summer prior to canopy closure (Duncomb et al., 1982).

Sediment yields never exceeded 1.0 Mg/ha per storm on areas receiving 8 to 15 Mg/ha of sewage sludge annually by injection. Copper (Cu), chromium (Cr), and zinc (Zn) concentrations (Table 1) did increase slightly in the sediments removed from sludge-treated areas, but the concentrations of cadmium (Cd), Cr, Cu, nickel (Ni), lead (Pb), and Zn (Table 2) were not increased in the runoff waters.

Table 1. Elemental content of selected stormwater sediments from corn areas after 5 years of sewage sludge applications at the Rosemount watershed in 1978.

Date	Treatment	Sediment Yield	Element							
			N	P	Zn	Cu	Cd	Ni	Pb	Cr
		kg/ha	mg/g	----------μg/g----------						
6/30	Control	830	10.0	626	52	27	0.8	18	27	45
	Sludge	552	7.7	862	73	46	0.8	18	35	97
7/10	Control	170	0.4	---	40	14	0.5	--	44	16
	Sludge	44	0.3	775	76	45	0.8	18	34	90

The losses of metals from grass sod areas were confined essentially to snowmelt runoff (Dowdy et al., 1980), since sodded areas yielded very little sediment. Trace metal concentrations in snowmelt waters were low, in the parts per billion range. Surface applications of sewage sludge during the growing season to grass areas did not cause increased Cd, Ni, or Pb concentrations in snowmelt runoff. Copper concentrations were consistently higher in snowmelt from sludge-treated areas than from control areas. However, winter applications of sludge on frozen, snow-covered grass areas, that had previously been sludge-treated, did result in increased Cr, Cu, Ni, Pb, and Zn concentrations in snowmelt runoff. Copper concentrations, which exhibited the largest change, were increased from 10 to 100 μg/L. The significance of this increase must be considered in light of USPHS drinking water standards of 1000 μg Cu/L.

The fact that Cu was the first and most consistent metal to appear in snowmelt runoff from grassed areas was not unexpected from a chemical point-of-view. All sludge applications were applied on the grass sod and accumulated as a mat on the soil surface. As snow melted, water filtered through the sludge mat, leaching organically-complexed metal ions out of the residual sludge and into runoff waters. Copper complexes are the most stable and soluble of the trace metals we measured and may be the fulvic acid types studied by Sposito et al. (1979).

Metal contamination of surface and groundwaters as a result of land applications of wastewater effluents is unlikely, particularly

Table 2. Elemental content of selected storm runoff water under corn and grass crops after 5 years of sewage sludge applications at the Rosemount watershed in 1978.

Date	Treatment	Runoff	N	P	Zn	Cu	Cd	Ni	Pb	Cr
			Element							
		cm	--mg/L--		----------µg/L----------					
					Corn					
4/7	Control	2.1	24.0	0.15	19	10	<2	4	<12	3
	Sludge	3.9	47.8	0.87	27	18	<2	9	<12	5
6/28	Control	0.8	3.7	0.14	28	4	1	4	22	12
	Sludge	0.6	3.1	0.82	21	14	<1	3	<15	< 2
6/30	Control	4.4	1.4	0.15	13	5	<1	6	<15	3
	Sludge	1.6	0.9	0.97	17	8	<1	4	<15	< 2
7/10	Control	0.4	0.5	0.23	16	5	<1	5	<20	< 3
	Sludge	0.4	0.6	0.51	12	7	<1	7	<20	< 3
9/14	Control	2.5	1.7	0.20	72	7	<1	2	<27	<10
	Sludge	1.8	2.1	0.61	19	12	<1	<11	<27	<10
					Grass					
4/7	Control	2.2	59.2	0.99	46	20	<2	12	<12	3
	Sludge	2.8	24.9	5.22	57	74	<2	9	<12	25
	Sludge†	3.1	49.6	7.78	52	92	<2	14	<12	26
6/26	Control	0.7	6.4	0.99	22	9	<2	4	<15	< 1
	Sludge	0.4	18.0	6.39	139	24	<2	8	<15	3
	Sludge†	0.5	47.3	7.64	30	29	<2	12	<15	4
7/10	Control	0.1	2.4	2.82	122	14	<1	7	<21	7
	Sludge	0.8	18.5	9.54	39	37	<1	19	<21	27
	Sludge†	0.7	7.2	8.03	34	63	<1	12	<21	16
9/14	Control	2.9	2.5	1.09	14	6	<1	< 3	<25	< 9
	Sludge	3.1	11.8	5.80	42	41	<1	9	<25	< 9
	Sludge†	2.7	13.1	6.41	15	45	<1	< 3	<25	< 9

† Received winter application of 4.0 Mg/ha sewage sludge in January 1978.

when renovation systems are managed in such a way as to maintain a vegetative soil cover. In addition to management techniques that limit environmental degradation due to land application of sewage sludge, municipal effluents have very low trace metal concentrations (Dowdy et al., 1978).

Nitrogen

Nitrogen is the element of major concern in land application of municipal wastewater effluents and sewage sludges. Average concentrations of 20 to 40 mg N/L for effluents and 3 to 5% N (30 to 50 g N/kg) for sludges are typical. When these amounts of N are applied to land at accepted agricultural rates and undergo biochemical and microbial transformations in the soil, questions immediately arise as to possible environmental pollution of surface and groundwaters. Much of the research on municipal waste utilization conducted during the past ten years has dealt with these potential problems of N management.

Application rates of N for most waste utilization experiments have covered the range from low (deficiency) to very high (excess) amounts, depending on the crops and soils under investigation. Our studies at Apple Valley with wastewater effluent applied to corn and forage grasses used annual N applications ranging from 190 to 830 kg N/ha/yr (Dowdy et al., 1982). Sewage sludge experiments have provided even greater N rates, including 620 to 1380 kg N/ha/yr for corn and reed canarygrass on a terraced watershed (Duncomb et al., 1982), 250 to 1460 kg N/ha/yr for turfgrasses (Clapp et al., 1982), and 1600 to 6300 kg N/ha/yr on small corn plots (Stark and Clapp, 1980).

Emphasis in waste utilization studies is placed on N because its transformations in the soil-plant-water system encompass the whole N cycle. Forms of N in effluents are especially important, having ratios of NH_4-N:NO_3-N:organic N ranging from 11:1:2 (Clapp et al., 1978) to 6:7:1 (Deese et al., 1977). A high NH_4-N to NO_3-N ratio appears to be desirable from both soil storage and water quality aspects. The NH_4-N form of N accumulates in the cation-exchange complex of soils, delaying its conversion to the NO_3-N form which can be leached into groundwater. The Penn State project (Sopper and Kardos, 1973b) experienced yield reductions of corn and high NO_3-N leaching when effluent-N changed to higher NO_3-N contents.

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, 1973b; Clapp et al., 1977b; 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, 1973b; Marten et al., 1978).

For rapid infiltration wastewater effluent treatment systems, N transformations in the soil below the infiltration basins were governed by flooding and drying sequences (Bouwer et al., 1980). If short, frequent flooding periods (~2 da) were used, the soil remained sufficiently aerobic for essentially complete conversion of the N in the effluent to NO_3-N in the renovated water. With longer flooding periods (~2 wk), O_2 in the soil was depleted during flooding, as NH_4-N in the effluent was adsorbed by the cation-exchange complex of the soil and then nitrified during drying. Resulting NO_3-N was partially denitrified and the rest leached out during the next flooding period.

With nitrification-denitrification as the main mechanism for N removal in rapid infiltration systems, the flooding and drying sequence that maximizes denitrification and N removal percentage depends on various factors which must be evaluated for each particular system. Pertinent factors include the NH_4-N and C contents of the effluent, infiltration rates, CEC of the soil, exchangeable NH_4^+ percentage, depth of O_2 penetration in soil during drying, and temperature. The combined laboratory and field data from the Flushing Meadows experiments (Bouwer et al., 1980) showed that to achieve high N removal percentages, the amount of NH_4-N applied during flooding must be balanced against the amount of O_2 entering the soil during drying. Flooding periods must be long enough to develop anaerobic conditions in the soil. Infiltration rates must be controlled to the appropriate level for the particular effluent, soil, and climate at a given site.

In a 30-year study of a rapid infiltration site at Hollister, CA (Levine et al., 1978) effective N removal was provided by 7 m of soil. Input total N levels of 40 mg/L were reduced to less than 4 mg/L within a shallow aquifer. No pollution hazard was posed by NO_3-N. A comparison of total N input to that gained by the soil revealed that only 2% of the wastewater N could be accounted for in the upper 300 cm of the soil profile.

For overland flow wastewater treatment systems, a number of mechanisms are involved in N removal, including volatilization, biological nitrification-denitrification, adsorption, soil storage, and plant uptake (Martel et al., 1982). The NH_4-N form can be removed by any of the above mechanisms. Most of the organic N is initially removed by sedimentation, then incorporated into the soil or converted to NH_3. The major losses other than crop uptake measured at Utica, MS

(Peters et al., 1981) were volatilization of NH_3 (about 9% of applied effluent N) and nitrification-denitrification (34 to 42%).

Specific design and operating criteria to optimize N removal or NH_3 conversion in overland flow systems have not been established, however, the following general relationships can be stated: (1) total N and NH_3 removed is inversely related to application rate and directly related to slope length, (2) rate of nitrification is reduced if wastewater is applied continuously, and (3) the overall N removal and NH_3 conversion efficiency is reduced as soil temperature drops below 13 to 14°C, due to decreased biological activity.

In contrast to wastewater effluents, most N in sewage sludges occurs in the organic form, with NH_4-N comprising about 10 and 30% of the total N for aerobically- and anaerobically-digested sludges, respectively. The rate of organic-N mineralization, therefore, becomes a top priority research objective in terms of determining N availability. Organic-N mineralization for freshly applied sludges has been studied by several investigators in the laboratory. Premi and Cornfield (1971) found from 2.3 to 4.2% of the added organic N had mineralized after six weeks of incubation. Epstein et al. (1978) reported that 41% of a digested sludge's organic N had mineralized after 15 weeks of incubation. Magdoff and Chromec (1977) found that 14 to 25% of the organic-N from an applied anaerobically-digested sludge mineralized during a 13-week period, whereas 36 to 61% of the organic-N mineralized from an applied aerobically-digested sludge. Ryan et al. (1973) showed that 4 to 48% of the organic-N in an anaerobically-digested sludge had mineralized after 16 weeks of incubation. Pratt et al. (1973) found that 35, 10, 6, and 5% of the residual organic-N is mineralized for the first, second, third, and fourth year, respectively, following application of sludge to agricultural land in California, whereas sludge application guidelines from Wisconsin cite a decay series of 15 to 20, 6, 4, and 2% (Keeney et al., 1975).

Our studies on residual-N availability from soils treated with sewage sludge in a field experiment indicated that application rate was more important than sludge type (Stark and Clapp, 1980). Those experiments proved that the chemical indexes of N availability were as reliable as more time-consuming incubation procedures for determining the N-supplying capacity of soils and sludges. Later studies on N availability from field soils five years after incorporation of sewage sludge have shown the importance of residual mineral-N in assessing long-term N supplying power of heavy sludge applications (Harding et al., 1981).

Concern about groundwater contamination by NO_3-N from application

of excessive amounts of effluents and sewage sludges has led to extensive monitoring of experimental sites with shallow wells and porous ceramic cup water samplers. Our Apple Valley wastewater effluent experiment showed that NO_3-N levels in soil water at 125-cm depths increased significantly with increased effluent application rates, especially under corn, but only occasionally exceeded 10 mg/L during early and late season (Fig. 1), when applications surpassed crop uptake demands (Larson et al., 1980). The Rosemount sewage sludge watershed study (Duncomb et al., 1982) illustrated N overloading problems where NO_3-N contents of soil water at 150-cm depths increased to 200 and 60 mg/L for corn and reed canarygrass areas, respectively, when annual N applications approached 800 kg N/ha. The data from this study suggest the ratio of total N application to crop removal should not exceed approximately 2.0 to prevent NO_3-N buildup at depths below the rooting zone of crops.

When municipal effluents and sewage sludges are applied to agricultural land, the primary objective becomes their use as a source of plant nutrients for crop production. Similarly, the most effective method of removing an element, e.g., N, is to harvest the plants that have taken up the element, and remove them from the area. Crop yields serve as an indicator of effectiveness of a waste utilization system and are influenced by many additional factors other than those directly related to the waste material, including climate, soil physical and chemical properties, drainage, crop species, crop and soil management, and application rates and schedules. The quantity of an element removed by a waste utilization system is determined by total dry matter produced (crop yield) and the content of that element in the harvested part of the crop.

Total N uptake for corn under the effluent irrigation system at Apple Valley (Clapp et al., 1978) ranged from 110 to 170 kg N/ha/yr for applications of 200 to 650 kg N/ha/yr (55 to 26% uptake). Other researchers in Pennsylvania (Sopper and Kardos, 1973b; Hook and Kardos, 1977) and in Florida (Overman and Nguy, 1975) obtained average values of 100 kg N/ha/yr uptake from 200 kg N/ha/yr applied (50%) and 180 kg N/ha/yr uptake from 580 kg N/ha/yr applied (31%), respectively. Corresponding uptake values for reed canarygrass in Minnesota (Clapp et al., 1978) ranged from 235 to 420 kg N/ha/yr for applications of 235 to 800 kg N/ha/yr (100 to 52% uptake). Reed canarygrass in Pennsylvania (Hook and Kardos, 1977) gave values of about 390 kg N/ha/yr uptake from 630 kg N/ha/yr applied (62%), and in Alberta (Bole and Bell, 1978) gave 59 to 237 kg N/ha/yr uptake from 52 to 410 kg N/ha/yr applied (113 to 58%), respectively. Similar N uptake values were obtained for reed canarygrass and orchardgrass in New Hampshire (Palazzo and McKim 1978; Palazzo, 1981). The 4-year average uptake of N by corn and reed canarygrass following sewage sludge application (Duncomb et al., 1982) was 225 kg N/ha/yr (26%) and 355 kg N/ha/yr (34%), respectively. Our results indicate that high yielding and good quality crops of corn and

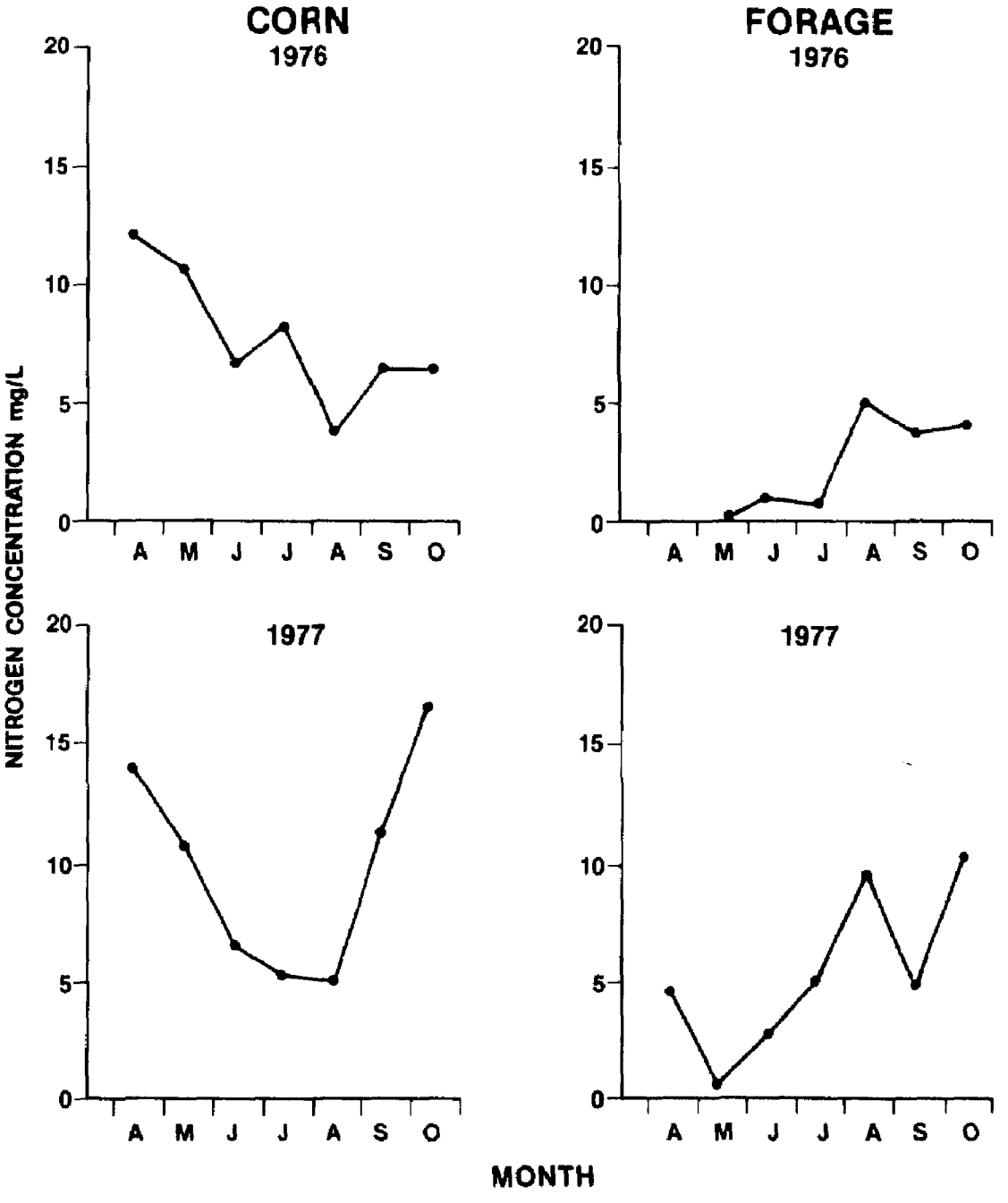

Figure 1. Monthly mean nitrogen concentrations at 125-cm depth under the high effluent treatment of corn and forage crops for 1976 and 1977 at the Apple Valley wastewater effluent project.

forage grasses can be produced under very high levels of N applied in wastewater effluent or sewage sludge, providing best management practices are followed to decrease adverse environmental effects.

Uptake of N by crops during treatment of wastewater effluent by overland flow systems depends on plant species as well as other factors influencing biological interactions. Water-tolerant forage grasses generally have high N uptake capacities. At the Utica, MS site (Peters et al., 1981), a grass mixture of reed canarygrass, tall fescue, perennial ryegrass, and common Bermudagrass removed between 179 and 222 kg N/ha/yr. At Hanover, NH, annual plant uptake by a mixture of tall fescue, orchardgrass, reed canarygrass, and perennial ryegrass ranged from 210 to 332 kg N/ha/yr (Palazzo et al., 1982). Proper management of the crop is extremely important in overland flow systems, especially with regard to stand establishment, weed control, nutrient status, and cold tolerance.

Phosphorus

Phosphorus is another component of wastewater sludges and effluents that theoretically could degrade surface and groundwaters. Like trace metals, P compounds are only sparingly soluble, and the greatest threat for degradation of the environment would be associated with P as a component of sediment eroded from the landscape. However, significant increases in the P concentrations of runoff waters from sludge-treated areas on our watershed have been observed (Table 3). Phosphorus concentrations and total P losses were considerably higher from the sludge-treated grass areas than from comparable corn areas. Incorporating sludge into the soil on the corn areas lowered P losses. Dunigan and Dick (1977) also reported that incorporation of sludge reduced P losses by 5 to 18 times. Maximum P loss occurred from the grass area that received winter applications of sewage sludge. Ketcheson (1978) and Shelton and Lessman (1978) also found greater soluble P losses during winter months after sludge was applied during the same winter period. The same general trends of increased P in storm runoff from sludge-treated areas appear to hold for runoff events occurring during the growing season.

Although in a soluble form, P loadings by wastewater effluent irrigations on croplands are much less than P additions via sewage sludge applications. Phosphorus additions were 2 to 4 times less than those made in our sludge utilization studies, even with an annual mean addition of 2.4 m of effluent. Our 6-year field study of wastewater renovation indicated that soils receiving comparable quantities of P as effluent or mineral fertilizer (rates for good crop management) behave in a similar manner with no apparent leaching of P to a 60-cm depth

Table 3. Phosphorus applied versus phosphorus in snowmelt runoff from terraced corn and grass areas at the Rosemount watershed.

		Corn			Grass		
			Runoff			Runoff	
Year	Treatment	Applied	Concen-tration	Quantity	Applied	Concen-tration	Quantity
		kg/ha	mg/L	kg/ha	kg/ha	mg/L	kg/ha
1976	Control	65	0.1	0.0	58	0.3	0.1
	Sludge	614	0.3	0.1	267	1.5	0.3
1977	Control	16	0.2	0.0	20	0.7	0.3
	Sludge	332	0.8	0.5	371	2.6	1.4
1978	Control	17	0.4	0.1	17	0.8	0.2
	Sludge	301	1.1	0.4	533	4.3	1.2
	Sludge†	---	---	---	648	10.4	3.1
1979	Control	17	0.2	0.1	17	0.6	0.9
	Sludge	579	1.0	0.4	405	2.5	1.2
	Sludge†	---	---	---	521	6.7	8.6

† Received winter applications of 4.0 and 5.3 Mg/ha sewage sludge in January 1978 and 1979, respectively.

(Latterell et al., 1982). For crops receiving twice this P loading, P leaching to the 60-cm depth was observed.

Phosphorus ranks second only to N in nutritional importance to crops. Some soils require additions of P fertilizer to sustain plant growth. Effluents and sludges, when applied in sufficient amounts to satisfy N needs for plants, will often provide P in excess of crop uptake, although P uptake by crops is neither as high nor covers as wide a range as N uptake. Corn and reed canarygrass removed about 35 and 55 kg P/ha/yr, respectively, with wastewater effluent application rates of 65 and 160 kg P/ha/yr at Penn State (Sopper and Kardos, 1973b) and 190 and 250 kg P/ha/yr at Apple Valley (Clapp, et al., 1978). Sewage sludges applied over a 4-year period on corn and reed canarygrass at the Rosemount watershed removed an average of 38 and 42 kg P/ha/yr, respectively, for applications of 460 kg P/ha/yr (Duncomb et al., 1982). 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.

Phosphorus removal from wastewater effluents by rapid infiltration and overland flow renovation systems is often less efficient than was observed in our slow infiltration studies. Since P removal in these systems is a result of sorption on soil-colloids and/or precipitation on the soil matrix, renovation efficiency is a direct function of effluent application rate and retention time within the treatment zone. Bouwer et al. (1980) reported a P removal of about 40% when secondary effluent was applied at maximum hydraulic loadings at the Flushing Meadows project. By reducing hydraulic loadings (i.e., increase retention time), these researchers were able to increase P removal to about 80%.

Literature summarized by the USEPA (1981) suggests that overland flow systems generally remove 40 to 60% of the P from wastewaters. Jenkins et al. (1978) observed an 80 to 90% P removal when effluent was applied to their overland flow system at approximately 0.05 m/wk during the summer months, but P removal decreased to about 30% during winter applications in New Hampshire. In Mississippi, increased retention times and/or reduced effluent application rates significantly increased P removal from applied wastewater (Peters and Lee, 1978; Lee and Peters, 1979). Similar trends have been reported by Thomas et al. (1974) and Abernathy (1983). However, additions of aluminum sulfate to the effluent prior to land application greatly improved P reductions in overland flow systems (Thomas et al., 1976; Peters and Lee, 1978).

Other Macro Nutrients

The macro nutrients K, Ca, and Mg, and the element Na, usually vary in concentration in effluents and sludges with respect to water source and wastewater treatment processes. Levels of K in most waste materials are not high compared to crop K requirements. Wastewater effluents can have a range in K concentrations of 10 to 15 mg/L, and with annual applications of 2.5 m should provide 250 to 375 kg K/ha/yr. High producing forages removing up to 350 kg K/ha/yr will require additional fertilizer K to maintain soil K levels and avoid deficiency (Marten et al., 1979). Sewage sludges are commonly low in K (<1%) and crops must be supplemented with K fertilizer for optimum growth when insufficient K is available from the soil. Plant K uptake may also be adversely influenced by high-lime (Ca) sludges or by soils containing free $CaCO_3$. Waste utilization experiments have not been designed to study K interactions and uptake because it is not a serious threat for water contamination. Usually fertilizer K is added in ample amounts, so as not to limit plant growth.

Most effluents and sludges contain sufficient amounts of Ca and Mg, and these nutrients normally do not adversely affect crop response. Effluent Ca and Mg levels can range from 10 to 20 and 5 to 25 mg/L, respectively. Sludge Ca and Mg values range from 3 to 5% and 0.5 to

1.0%, respectively, except for high-lime materials, which can contain 15% Ca. Corn and reed canarygrass may remove up to 25 and 50 kg Ca/ha/yr and 25 and 40 kg Mg/ha/yr, respectively (Clapp et al., 1978; Duncomb et al., 1982).

There has been concern that high concentrations of Na and associated salt anions in effluents could lead to degradation of soil structure and permeability and that Na could compete with K for exchange sites in the soil and for plant uptake. We observed that an equilibrium was established between effluent, soil, and soil water after four years to give an exchangeable Na percentage of 6.0% in the Apple Valley study, even though Na concentrations of 275 mg/L and an application rate up to 8,000 kg Na/ha/yr were used (Larson et al., 1979). We know of no problems associated with Na for either effluent or sludge experiments. The higher salt concentrations of sewage sludges (2.5 to 6.0 dS/m) may limit plant growth (Larson, et al., 1981; Clapp et al., 1982). This effect is not relatively wide-spread, but will bear observation in the course of waste utilization studies.

The potential for groundwater contamination by sludge-borne trace metals is extremely limited (Dowdy and Volk, 1983). Although some studies have noted movement of metals to layers below the zone of incorporation, it appears that Zn is the element that most consistently has potential to move in soils. The mechanisms by which it moves, whether as Zn^{2+}, as an anionic form, or as a chelated compound, have not yet been established. Trace metal movement will most likely occur with large applications to a sandy, acid, low organic matter soil that receives high rainfall or irrigation. Even under these conditions the extent of movement will be limited, but metals may move through open soil channels or cracks where the soil has no opportunity to attenuate them.

PERSPECTIVES ON FUTURE RESEARCH

Short term research on the management of land treatment systems for the renovation of municipal sewage wastes has resulted in the development of systems that are successfully operated throughout the world. Many short term and site specific management problems have been encountered and solved. Infiltration rates, N leaching, crop response, and runoff, among other potential problems, can be controlled through proper management. Proper management, however, requires some very specific management skills and continuous, perceptive observation of the system. It is the necessity for transferring such skills from successful operation of land treatment systems to new designs that dictate future research in this area. Research information must be broad enough to be applicable to new sites and conditions in order to allow design of improved systems.

Needs for research over the long term are more numerous. Site aging with respect to trace metals and P are examples of these longer term needs. The fate of applied materials such as N and trace metals after cessation or during long periods of application also require longer term research. Will soils continue to immobilize P and metals in forms that prevent water contamination or plant uptake? While we may have a reasonable understanding of how much of various materials will move within the soil mass over the short term, an understanding of the chemistry controlling such phenomena is very limited. This information is essential to development of sound management criteria applicable to a wide range of soil and climatic conditions. Simulation models (Gupta et al., 1978; Iskandar, 1981) could provide many of the answers required for solving the long term research problems of land treatment systems.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

Donna J. Griffin, Mintech, Inc., Tulsa, OK: Dr. Linden showed a slide reflecting more runoff from sludge plots than control. I questioned it as I would anticipate greater infiltration of sludge-amended soils.

Dennis R. Linden: In theory and practice sludge applications to land have two conflicting effects on infiltration. First of all, the organic materials in sludge do promote soil structure and to the stability of soil structural units during wetting which should be an enhancement to infiltration. The second effect is that of sealing or clogging of soil pores because of the deposition of fine-grained materials. This effect is a detriment to infiltration, but is of minor significance if sludge applications are incorporated into the soil.

Little experimental evidence exists to show major impacts of sludge applications on the infiltration characteristics of soil.

Kenneth Dotson, EPA, Municipal Environmental Research Laboratory, Cincinnati, OH: Were concentrations of nitrogen in soil-water under sludge application plots NO_3-N or N?

Dennis R. Linden: Concentrations were of NO_3-N, reported as mg of NO_3-N per L in water.

Stephen Campbell, BIOGRO Systems, Anapolis, MD: 1. Does soil compaction from application equipment affect the percolation of water through soil, and to what extent is it a management factor?

Dennis R. Linden: Yes, Soil compaction from traffic will reduce percolation of water and is a major management factor. The moisture condition and "trafficability" (ability to withstand loads without serious compaction) are factors that need to be considered. Scheduled drying-out periods must be allowed in order to permit farming (planting, cultivation, harvest, etc.) practices without damage to the soil.

In general a soil with good internal drainage will have suitable trafficability after about two days of drainage and drying.

Stephen Campbell: 2. Was sludge actually applied on top of snow for your runoff studies?

Dennis R. Linden: This is not a generally recommended practice, but in this carefully monitored experimental situation, there were no adverse effects to the environmental quality from the "on snow" application of sludge. "On snow" applications of animal wastes (manures) have been shown to reduce soil erosion because of the soil surface stabilizing influence.

HYDROLOGIC MANAGEMENT: TRACE METALS

FATE OF TRACE METALS DURING LAND TREATMENT OF MUNICIPAL WASTEWATER

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Application of municipal wastewater to land has been practiced since the advent of wastewater treatment (Iskandar, 1978; Jewell and Seabrook, 1979). Through this practice, wastewater receiving various degrees of pretreatment has been introduced into the soil system with waste disposal being the primary objective. The public health and hydraulic aspects of spreading wastewater have been extensively investigated. However, only within the past two decades has attention been given to their effects on water quality. Although soils have been shown to be an effective medium in attenuating suspended solids, biodegradable organic matter and pathogens, land treatment systems have not performed in a consistent manner under diverse conditions. During the 1973 workshop on Recycling Municipal Sludges and Effluents on Land the need to better characterize soils' performance as a wastewater treatment medium was keenly recognized. It was also recognized that phytotoxic levels of trace metals accumulate in soils and that accumulation of hazardous trace metals in harvested plant tissue may restrict land treatment of wastewater.

Conceptually, land treatment differs from conventional municipal wastewater treatment in two aspects. In a conventional wastewater treatment system, each unit process is designed to deal with one category of pollutants in water. As the wastewater is being purified in a treatment plant, it passes through a treatment train consisting of several selected unit processes arranged in a logical sequence. For land treatment, a system is expected to remove the multitude of pollutants from applied wastewater in one step. Unlike a conventional wastewater treatment system which consists of a treatment mode and a discharge mode, the treatment and the discharge are frequently combined into a single operation in land treatment (except overland flow systems). Because effluents from land treatment may directly enter surface or groundwater, it is important that final quality of the discharge can be controlled and predicted if operational conditions change.

Existing land treatment systems are usually designed and operated to optimize the removal of suspended solids and BOD. Although trace metals are not expected to be a limiting factor in land treatment, it is essential that potential detrimental effects of trace metals are not overlooked. This report reviews and summarizes results from case studies concerning the fate of trace metals during land treatment of wastewater effluents.

TRACE METALS IN WASTEWATER

"Site specific" is an inherent feature of all land treatment systems. It is essential that the composition of wastewater to be treated is carefully matched with the characteristics of the site and planned system. Successful application of land treatment methods depend on several interrelated factors such as soil type, topography, groundwater characteristics, climate, etc. Since climatological, hydrological, and pedological features of a land treatment site can not be easily altered, pretreatment of wastewater or selective management techniques are often necessary to make up for deficiencies of the site. In this manner, contaminants not compatible with the prescribed land treatment scheme may be removed prior to water application or by imposing special management of the system.

Pretreatment requirements for land treatment vary for overland flow, rapid infiltration and slow rate systems. Although overland flow systems designed for treating municipal wastewater are comparatively new, they will accept a wastewater with minimum pretreatment and produce an effluent similar in quality to secondary effluents from conventional wastewater treatment systems (Hall et al., 1979). High rate systems characterized by rapid infiltration/percolation through porous soil medium, are frequently used for effluent polishing. Unless the influent to a high rate land treatment system is properly pretreated, the hydraulic loading and the treatment performance will be difficult to maintain. Many operating high rate application systems in the U.S. renovate secondary effluents (Bouwer et al. 1974; Leach et al., 1980). The function of slow rate systems may range from simply providing water for crop irrigation to substituting for advanced treatment. The degree of pretreatment required for slow rate systems must, therefore, be adjusted accordingly.

Trace metals are minor constitutents in municipal wastewater. Trace metal contents of municipal wastewater are, however, almost always higher than its source water due to industrial waste contribution and input from domestic sources such as human wastes and consumer products used daily (Hathaway 1980). For a typical municipal wastewater in the U.S., non-industrial sources' share of total trace metal load usually are <25% of the total metal loading (Klein et al., 1974; Davis and Jacknow, 1975; Levins et al., 1979).

Once trace metals enter the wastewater stream they tend to be adsorbed by wastewater solids or form inorganic precipitates (Blakeslee, 1973; Chen et al., 1974). During treatment, trace metal concentrations of wastewater will be reduced as suspended solids and biodegradable organic matter are separated from the water. Since wastewater treatment systems are not designed and operated to optimize the removal of trace metals, consistent removal of metals is not always achieved. Table 1 summarizes concentrations of selected trace metals in municipal wastewater and treated effluents. Even after secondary treatment, significant amounts of trace metals present in effluent are present as finely divided suspended particles (Chen et al., 1974).

Table 1. Concentrations of trace metals in municipal wastewater and their treated effluents.

	Raw Wastewater[1]		Primary Effluent[2]		Secondary Effluent[2]	
Element	Range	Median	Range	Median	Range	Median
	mg·L^{-1}					
As	<0.0003-1.90	0.085	<0.005-0.03	<0.005	<0.005-0.023	<0.005
B	<0.123-20.0	-	<0.01-2.4	1.0	<0.1-2.5	0.7
Cd	<0.0012-2.14	0.024	<0.02-6.4	<0.02	<0.005-0.15	0.005
Cr	<0.0008-83.3	0.40	<0.05-6.8	<0.05	<0.005-1.2	0.02
Cu	<0.0001-36.5	0.42	<0.02-5.9	0.10	<0.006-1.3	0.04
Hg	<0.0001-3.00	0.11	<0.0001-0.125	0.0009	<0.0002-0.001	0.0005
Mo	<0.0011-0.874	-	<0.001-0.02	0.007	0.001-0.0018	0.007
Ni	0.002-111.4	0.23	<0.1-1.5	<0.10	0.003-0.6	0.004
Pb	0.001-11.6	0.12	<0.02-6.0	<0.2	0.003-0.35	0.008
Se	<0.002-10.0	0.041	<0.005-0.02	<0.005	<0.005-0.02	<0.005
Zn	<0.0001-28.7	0.52	<0.02-2.0	0.12	0.004-1.2	0.04

[1]Derived from data in Minear et al., 1981.
[2]Chang and Page, 1983.

ATTENUATION OF TRACE METALS IN SOILS

In terms of land application of wastewater, the soil may be viewed as a dynamic filter. It is a chemically and microbiologically active porous medium consisting of weathered minerals, organic residues, microorganisms, water and air. Passage of wastewater through the soil matrix will induce a variety of reactions which are the basis for soils to renovate wastewater.

Mechanisms responsible for removing constituents from wastewater during land treatment depend on the physical characteristics of the impurities present. Straining and filtration are the the primary

mechanisms for removal of suspended solids present in applied water. If wastewater particles are larger than soil pores, particulates are strained off at the soil-water interphase. High hydrualic loading and/or high suspended solid concentrations may result in rapid accumulation of a layer of finely divided particles at the soil surface and thereby restrict the infiltration of water. From an operational point of view, straining to remove undesirable constituents during land treatment frequently is undesirable when organic particles accumulate much faster than they are biologically decomposed. For those suspended solids that are not retained at the soil surface, retention by filtration at lower soil depths is quite effective. As suspended particles travel with water through the porous medium, they are trapped and brought into contact with the surface of the soil solid phases by various physical and hydrodynamic forces which control behavior of particles suspended in fluids (Ives, 1973). Suspended solids coming in contact with soil particles may then be immobilized by surface adsorption. The effectiveness of filtration is a function of suspended solid concentrations, the soil's mechanical characteristics and the hydraulic loadings (Yao et al., 1971). Generally, lower concentration, finer soil texture, slower flow velocity and longer travel distance enhance suspended solids removal during filtration. Since significant amounts of trace metals are present in wastewater and their treated effleunt as finely divided suspended solids, there is reason to believe that filtration is probably the most effective mechanisms to attenuate trace metals during land treatment. In overland flow, the majority of the applied water flows across the relatively impervious slope in a thin sheet. Suspended solids are removed as water travels through the vegetative mat. With the removal of suspended solids, the concentrations of trace metals in the water applied should also be reduced.

Trace metals also are present in wastewater in dissolved forms. The physical reaction of straining and filtering would have little effect on the removal of dissolved constituents during land treatment. If dissolved impurities are to be effectively retained by soil, chemical and microbial reactions are required to immobilize them. Chemical reactions that determine the forms and fate of trace metals in soil receiving wastewater have been extensively reviewed (Lindsay , 1973; Chang and Page, 1983). Trace metals present in the soil solution may enter the stationary phase by exchanging with ions already on exchange sites or adsorbing onto specific surfaces of amorphous soil constituents. If the solution concentration of a trace metal element exceeds the solubility product of solid phase compounds precipitation will take place. The trace metal containing soil minerals may also dissolve when the trace metal element in question becomes under-saturated in the solution phase. Constituents dissolved in soil solution are also subject to chelation by soluble organic matter, removal by plant, microbial uptake and leaching by water movement. Under certain circumstances, a few elements such as As, Se, Hg, etc., may volatilize. Recent advances in understanding soil chemical equilibria have contributed significantly to our understanding

of the interactions of various inorganic chemical constituents with the soil matrix (Page et al.,1981). However, the rate and degree of ion exchange, specific adsorption, precipitation, dissolution and organic chelation reactions depend on kinetic and thermodynamic factors in soils. The shifting in soil pH and the redox potential are especially important in defining the chemical reactions between trace metals and soils (Sims and Patrick, 1978). Attempts have also been made to model the trace metal equilibria in the soil solution based on thermodynamic association constants and solubility products of various chemical reactions in soils (Mattigod and Sposito, 1979). Ideally, the outcome of chemical equilibrium should be coupled with the hydraulic aspects of the land treatment to project trace metal solute movement in soils. Numerous studies on chemistry of trace metals introduced into soils through wastewater application tend to indicate strong immobilization of trace metals in soils (Lindsay, 1972). Since chemical equilibria occur in all soil systems, although substantial attenuation occurs, complete removal of trace metals during land treatment of wastewater is not likely to take place.

FATE OF TRACE METALS DURING LAND TREATMENT

In the soil environment, chemical reactions described in the previous section occur simultaneously. Because of the interacting nature of these chemical pathways in soils, it is difficult to distinguish the contribution of each chemical reaction in soils experimentally. To complicate the problem even further, trace metal additions to soils during land treatment usually are quite small. As a result, impacts of trace metals to the waste-receiving soils and on the quality of finished water are difficult to evaluate especially under short term experimental conditions. Any adverse effect of land treatment on soil, crop or groundwater most likely will only be detected from studying long-term wastewater application sites. Through efforts by the U.S. Environmental Protection Agency, long-term environmental effects of wastewater land application sites have been extensively investigated in the past ten years.

During the past 10 years, treatment performance of several large scale slow rate wastewater land application treatment systems have been studied (Seabrook, 1975; Hossner et al., 1978; Demirijian, 1980). Environmental impacts at many long-term domestic wastewater irrigation sites throughout the U.S. were investigated (Benham-Blair and Associates, Inc. and Engineering Enterprises, Inc., 1979a; Koerner and Haws, 1979a; Overman, 1979; Stone and Rowland, 1980a; 1980b). Trace metal loadings did not limit the land application of wastewater in any of the above systems studied. Although metal contents of applied wastewater and hydraulic loading varied extensively, all studies reported significant reduction of trace metals by land treatment (Table 2).

Table 2. Trace metal behavior during slow-rate land treatment.[1]

Element	Max. Contamin. Level in NIDWR[2]	Applied Wastewater Conc'n	Treatment Location: Muskegon, MI		San Angelo, TI		Melbourne Aust.	
			Effl. Conc'n	Removal	Effl. Conc'n	Removal	Effl. Conc'n	Removal
	($mg \cdot L^{-1}$)	($mg \cdot L^{-1}$)	($mg \cdot L^{-1}$)	(%)	($mg \cdot L^{-1}$)	(%)	($mg \cdot L^{-1}$)	(%)
Cd	0.01	0.004-0.14	<0.002	90	<0.004	-	0.002	80
Cr	0.05	0.02-0.7	0.004	90	<0.005	>98	0.03	90
Cu	1.0	0.02-3.4	0.002	90	0.014	85	0.02	95
Pb	0.05	0.05-1.3	<0.050	>40	<0.005	-	0.01	95
Mn	0.05	0.11-0.14	0.26	15	-	-	-	-
Hg	0.002	0.002-0.05	<0.002	-	-	-	0.0004	85
Zn	5.0	0.03-83	0.033	95	0.102	25	0.04	95

[1] From Uiga and Crites, 1980.
[2] National Interim Drinking Water Regulations.

Regardless of the trace metal content of influents, concentrations in effluents were consistently below maximum contamination levels permitted by the National Interim Primary Drinking Water Regulations. Since the removal efficiencies in land treatment (80-95%) exceeded the fraction of total trace metal normally expected to be present in wastewater in suspended solid form, mechanisms other than straining and filtration must also have been active in attenuating trace metals. During wastewater application, no crop was adversely affected by trace metals. There was relatively little increase in trace metal uptake by plants receiving wastewater over those irrigated with a water of non-wastewater origin (Table 3). The accumulation of trace metals in soils receiving wastewater is directly proportional to inputs. Because amounts of trace metals added into soils through slow rate wastewater application in relation to total volume of soils is small, trace metal concentrations in affected soils may not be significantly elevated for some time. Given adequate time, accumulation of trace metals in soils will become evident

Table 3. A comparison of chemical constituents in the plant tissue (an unspecified grass) from sites irrigated and not irrigated with wastewater effluent for 17 years at Dickinson, North Dakota (derived from data in Benham-Blair and Associates, Inc. and Engineering Enterprises, Inc. 1979a).

Constituent	Control	Wastewater Irrigation
	- - - - - - $\mu g\ g^{-1}$ dry wt.	- - - - - - - -
N	19	15
P	2.7	2.7
K	5.1	4.3
Na	1.2*	3.1
S	1.2	1.1
(NO_2+NO_3)-N	0.7	1.0
As	<0.001	<0.001
B	0.14	0.20
Cd	<0.005	<0.005
Cr	0.002*	0.005*
Cu	0.007	0.007
Pb	<0.005	<0.005
Zn	0.022*	0.037*

*Control and wastewater treated are statistically different at the 0.05 level.

(Bouwer and Chaney, 1974). However, almost all trace metals introduced through wastewater applications are deposited in the surface 20 cm (Hossner et al., 1979). Obviously, soil's ability to retain trace metals is not unlimited. The capacity for metal removal in soils is expected to be equal to or better than retention of phosphorus. For a wastewater with a typical trace metal content, it will take several hundred years for a slow rate land treatment operation to exhaust the cation exchange and specific adsorption capacity of a typical soil.

Rapid-Infiltration Treatment

In order for a land treatment site to receive large amounts of wastewater, highly permeable soils and a fast draining aquifer are necessities. Large volumes of water draining rapidly through the soil profile will also influence treatment performance. Soils with acceptable hydraulic properties usually do not possess high adsorption capacities. Because of the high hydraulic loading, kinetics of chemical reactions in soils may be critical in the degree of trace metal attenuation achievable. Characterisically, rapid infiltration systems are not suitable to handle wastewater with a heavy pollutant load.

Results obtained from studying rapid-infiltration systems in the U.S demonstrated that effluents of satisfactory quality may be produced with either primary effluents or secondary effluents as input water. Most short duration operations (<5 years) showed effective immobilization of trace metals by the soil matrix (Smith et al., 1979; Vaccaro et al., 1979). However, when trace metal contents of renovated water at several long-term rapid-infiltration systems (>10 years) were compared with those from control wells, concentrations of several trace metals in renovated water were significantly elevated (Table 4). Elevated levels of iron and manganese originating from indigeneous Fe and Mn oxides in soil were reduced under conditions created by wetting cycles. Higher concentrations of other metals, such as As, Cd, Ni, and Zn, were also found in the recharged groundwater. Generally, the metal concentrations of the renovated water did not appear to be significantly different from the water applied (Benham-Blair and Associates,Inc. and Engineering Enterprises, Inc. et al., 1979b; Koerner and Haws, 1979b; Leach et al., 1980; Pound et al., 1978). Similar results also have been reported by other investigators (Bouwer et al., 1974; Aulenbach, 1979). Although their extremely low concentrations in the water are not likely to interfere with the beneficial use of the groundwater aquifer, the available data, nevertheless, demonstrates that trace metals may be transported into groundwater aquifers through long-term rapid infiltration operations. Data from rapid infiltration systems also show significant accumulation of trace metals near the surface of the affected soil profile. At Whittier Narrows (Los Angeles County, CA), secondary effluents have been used for artificial groundwater recharge since 1963 at an annual average rate of 83 m yr^{-1}. Sampling the recharge spreading basin in 1980 indicated significant amounts of trace metals were deposited in the surface 60 cm of the soil profile (Table 5). The average concentration of 4N HNO_3 extractable metals in the surface 60 cm of soil, normalized to account for different depth increments by assuming a uniform bulk density for soil of 1.5 g cm^{-3}, increases from 0.1, 38, 51, 32, 5.3, and 73 to 3.1, 128, 116, 112, 44, and 241 μg g^{-1}, for Cd, Cr, Cu, Ni, Pb, and Zn respectively. The removal of trace metals from the applied wastewater in the upper 60 cm soil layer was likely due to the result of filtration of suspended solids and compounds precipitated during water application. More importantly, the accumulation of trace metals in the soil profile has elevated the concentration of soluble trace metals in the surface of the affected soils (Table 6), enhancing the leaching of trace metals to deeper soil strata.

Overland Flow System

Overland flow treatment systems have long been used to treat food processing wastes. However, the effectiveness of these systems in treating domestic wastewater was not demonstrated until recently (Thomas et al., 1974; Overcash, 1978; Hall et al., 1979; Hinrichs et al., 1980). Despite the popularity of land treatment in the past 10 years, full scale municipal treatment facilities using overland flow are not extensive.

Table 4. Comparison of trace metal content in groundwater of long-term rapid-infiltration systems with their corresponding controls.[1]

Parameter[2]	Hollister, CA		Lake George, NY		Vineland, NJ		Milton, WI	
	Control	Renovated	Control	Renovated	Control	Renovated	Control	Renovated
Preapp. Treatment	Primary		Secondary		Primary		Secondary	
Operation Period(yr)	30		38		50		40	
Applic. Area (ha)	8.8		2.2		36		0.2	
Hydraulic Load.($m \cdot yr^{-1}$)	15.4		44.7		11-21.4		110-224	
Daily Flow ($L \cdot s^{-1}$)	43.8		30.8		215		14.5	
Depth of Groundwater(m)	6.8-9.2		1.3-7.0		1.0-3.5		2.0-3.0	
pH	7.8	7.7	6.6-7.8	6.5-7.4	5.2	6.6	7.7	7.5
EC ($dS \cdot m^{-1}$)	1.148	1.828	-	-	0.064	0.468	0.677	1.379
BOD	3	13	1.2	0.8-3.0	1.1	65	18.0	36
As	<0.01	0.01	-	-	0.006	0.01	0.005	0.033
Fe	0.03	0.51	2.24	0.43-8.25	1.6	12.6	0.50	2.0
Hg	<0.001	0.001	-	-	0.001	0.001	0.001	0.001
Mn	0.01	0.62	-	-	0.11	0.13	0.10	0.29
Ni	0.039	0.16	-	-	0.10	<0.10	0.10	0.10
Pb	0.012	0.08	-	-	0.10	0.10	0.20	0.20
Se	<0.001	0.001	-	-	0.01	0.01	0.05	0.005
Zn	0.05	0.090	-	-	0.013	0.024	0.628	0.254
Ba	<0.07	0.15	-	-	-	-	-	-
Cd	<0.003	0.050	-	-	0.02	0.02	0.02	0.02
Co	<0.006	0.013	-	-	0.05	0.05	0.05	0.05
Cr	0.032	0.017	-	-	0.03	0.02	0.05	0.05
Cu	0.025	0.036	-	-	0.02	0.023	0.10	0.10
Al	-	-	-	-	1.8	0.50	1.0	1.0

[1] From Leach et al. 1980.
[2] Expressed in $mg \cdot L^{-1}$ except pH or otherwise indicated.

Table 5. Accumulation of trace elements in soils used for groundwater recharge (Whittier Narrows, Los Angeles, CA).[1]

Depth (cm)	Concentration of Element in Soil ($\mu g\ g^{-1}$)					
	Cd	Cr	Cu	Ni	Pb	Zn
			RECHARGE AREA[2]			
0-5	2.3	88	69	71	61	176
5-15	5.6	215	128	145	79	322
15-30	3.4	111	117	128	34	209
30-60	2.2	95	86	84	21	136
60-90	0.9	46	42	44	6	73
90-120	0.3	27	31	37	5	62
120-180	0.2	31	37	45	6	75
			CONTROL AREA[3]			
0-5	0.2	32	41	26	9	83
5-15	0.1	33	45	27	5	83
15-30	0.1	27	41	24	5	65
30-60	0.1	40	51	333	3	83
60-90	<0.1	14	19	13	1	40
90-120	<0.1	20	27	60	1	61
120-180	0.1	25	45	31	-	86

Table 6. Concentrations of trace elements in soil saturation extracts of soils used for artificial groundwater recharge (Whittier Narrows, Los Angeles, CA).[1]

Depth (cm)	Concentration of Element in Soil Saturation Extract ($\mu g\ L^{-1}$)				
	Cd	Cr	Cu	Ni	Zn
			RECHARGE AREA[2]		
0-5	1.4	11	222	115	34
5-15	1.2	22	210	88	26
15-30	0.2	24	119	56	20
30-60	<0.1	33	121	65	16
60-90	<0.1	18	93	56	17
90-120	<0.1	5	80	53	16
120-180	<0.1	5	69	40	20
			CONTROL AREA[3]		
0-5	1.4	2	<20	194	79
5-15	0.2	2	<20	181	73
15-30	0.2	2	<20	181	70
30-60	0.2	<2	<20	<25	70
60-90	0.2	<2	<20	<25	70
90-120	0.4	<2	<20	<25	68
120-180	0.1	<2	<20	<25	76

[1] Derived from Chang and Page 1983.
[2] Mean of 4 samples collected from recharge basins which had received secondary wastewater from 1963 through 1980.
[3] Results from one sampling location adjacent to the recharge basins.

For domestic wastewater, overland flow may be used to polish secondary effluents or achieve secondary level treatment. In terms of treatment performance, the overland flow land treatment systems used for the latter case generally have out-performed conventional secondary treatment in publicly owned treatment works (Hall et al., 1979). In either case, substantial nutrients and trace metals removals in addition to suspended solids and BOD reductions are achieved.

Conceptually, process mechanisms of overland flow treatment systems resemble a conventional attached growth biological treatment system minus sludge withdrawal. Although performances of overland flow systems measured in terms of BOD and suspended solids removal, may be influenced by water distribution methods, soil slope, soil temperatures, etc., and are far from consistent, it has exhibited effective retention of trace metals. The mass balance data from a pilot system at Utica, Mississippi show, for example, that less than 10% of the applied trace metals appeared in the effluent (Table 7). As wastewater flows across the soil surface, trace metals are adsorbed by organic solids deposited in the decomposing layer and plant roots concentrating trace metals toward the upper end of the soil slope. Although there is little operation data to substantiate the author's claim, it is logical to expect that trace metal deposited with organic solids at the soil surface are not as tightly bound as those immobilized in the soil profile. Unless surface deposits of overland flow systems are routinely removed, their capacities for trace metal retention will sooner or later be exceeded.

Table 7. Distribution of trace metals in an overland flow treatment system, Utica, Mississippi.[1]

	Percent of Total Applied		
Element	Effluent	Vegetative Cover	Soil
Cd	1.2	7.6	91.8
Cu	4.0	14.5	81.5
Ni	1.4	4.4	93.9
Zn	7.6	12.1	80.3

[1] From Peters et al. 1980.

CONCLUSIONS

1. Trace metals introduced into soils through land application may be retained in the soil matrix by straining and filtration (for suspended and colloidal solid forms) or by ion exchange and specific adsorption mechanisms (for dissolved forms). Trace metals in dissolved forms may also precipitate out of solution and subsequently be removed as suspended solids.

2. Land treatment systems are customarily designed for optimum hydraulic loading or removal of suspended solids and BOD. Reviewing operational data of land treatment systems revealed that effective attenuation of trace metals present in wastewater was also achieved. To date, trace metals have not limited land treatment operations. However, trace metals are accumulating in the surface of the affected soils under all modes of land treatment.

3. For slow rate wastewater application, trace metal contents of effluents are reduced. Processes involved in trace metal attenuation include the physical mechanism of filtration and the chemical sorption by ion exchange and specific adsorption. Even after years of wastewater spreading, crops did not exhibit any significant increase in trace metal uptake. Under slow rate wastewater treatment, soils are expected to possess an exceptionally large capacity to immobilize trace metals.

4. Results obtained from long-term operations demonstrated that trace metals may be transported into groundwater aquifers under hydraulically active rapid infiltration systems. However, the adverse effect on the quality of renovated water can be minimized by pre-application treatment. In overland flow treatment, trace metals are deposited in the organic layer near the soil surface and are less tightly bound than those immobilized in the soil matrix. Unless organic deposits of the overland flow systems are removed routinely, their capacity for retaining trace metals will sooner or later be exceeded.

5. To prevent long-term trace metal problems land treatment systems source control and pre-application treatment techniques should be instituted to minimize metal concentration in wastewater applied.

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FOREST SYSTEMS

THE ROLE OF FORESTS IN SLUDGE AND WASTEWATER UTILIZATION PROGRAMS

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Forested areas are receiving increased attention as potential sites for the disposal and biological recycling of both wastewater and sludges from municipal treatment facilities. The current sewage management programs by the city of Bremerton in the state of Washington and by Clayton County in Georgia exclusively use forested sites to effectively and economically reuse sewage treatment plant by-products (Reynolds and Cole 1981, Snell et al. 1981). In both cases, the application rates are designed to maintain a high standard of drainage water quality while also enhancing the productivity of the forest crop. Similarly, municipal sewage sludge from the greater Seattle area is now destined to be applied to tree farms as a silvicultural tool to increase the productivity of forest stands (Metro 1983). It is the purpose of this article to discuss the underlying concepts that have given rise to this newly emerging technology for incorporating forestry into a sludge and wastewater utilization program.

There are a number of obvious reasons for considering forested sites as potential candidates for the disposal and reuse of sewage from a municipal treatment plant:

1. In many regions of the United States extensive acreages of forest land are potentially available for such a program. Forests occupy 40% of the landscape in the contiguous forty-eight states.

2. Forests are typically located in the better drained sites and not subject to the periodic flooding of alluvial agricultural areas.

3. Many of the forests in the United States are markedly deficient in major nutrients that are found in municipal sludge and wastewater, especially nitrogen and phosphorus.

A lack of adequate nutrition is the main factor limiting forest productivity in this country.

4. Since forests are not food chain crops, many of the public health concerns and land application regulations should not be as critical as those associated with agricultural sites.

5. Forest soils theoretically have properties well suited to receive sludge and wastewater additions, including a great deal of organic carbon which will immobilize available nitrogen, a high infiltration rate which should minimize the potential for surface runoff, and a perennial root system which should allow for year round uptake of available nutrients.

The application of these products in a forest environment has not been without its critics and problems. It has been argued that such additions are in conflict with traditional uses of the forest, especially those involving recreation. For example, concerns over heavy metal uptake by edible mushrooms and berries have been raised. It has also been suggested that sewage additions could adversely affect the forest wildlife, and through such wildlife, toxic substances might enter the human food chain.

What might be regarded as more tangible concerns have also been raised. One is that many forest sites are located on areas of excessive slopes which could allow escapement of sludge particulates or dissolved solids through surface water runoff. Another is the concern that tree species in their natural setting generally take up far fewer nutrients than agricultural crops, possibly resulting in more time for nutrient transformation or escapement. Not all tree species respond significantly to nutrient additions, and to convert a site to a more desirable species could take a number of years. Most important, it has been pointed out that there is much less experience and substantive research dealing with forest applications than with agricultural applications, and therefore the design of sludge use for a forest site is more speculative.

In order for forests to play a significant role in a land application program, such questions and concerns must be addressed. Research has been under way both in this country and elsewhere for well over a decade examining the effects of applications in forested sites. Examples of such work include Kardos and Sopper (1973) in Pennsylvania, Urie (1973, 1979) and Harris and Urie (1983) in Michigan, Bialkiewicz (1978) in Poland, Nutter et al. (1978) in Georgia, and Cole (1982) in Washington. Not all the results from these studies have been favorable. It is clear that excessive applications can result in detrimental environmental effects such as significant nitrate leaching (Sopper and Kerr 1978, Riekerk 1978, Vogt et al. 1981, Urie 1973, Brockway and Urie

1983). Growth trials point to the wide range of responses that can be expected from various tree species and stage of tree growth (Cole 1983, Smith and Evans 1977, Koterba et al. 1979), some of which are far from favorable. The fates of heavy metals and pathogens have not been as extensively explored in forest sites as in agricultural ones, although the initial results as reported by Zasoski (1981) and Edmonds (1979) would indicate this issue is of lesser concern.

A major initial problem in establishing forest application sites was the lack of adequate application equipment technology. Equipment that had been developed for agricultural fields was generally not suitable for the broken terrain of a forested site.

These and other problems have now been largely resolved, and fully operational forest sites now receive sludge and wastewater additions. The bases for such additions and the expected forest responses are outlined below. While the results for this discussion have been largely derived from our studies concerning both wastewater and sludge carried out at Pack Forest (the research and demonstration forest of the College of Forest Resources, University of Washington, located 75 miles south of the campus in the foothills of the Cascade Mountains), the experiences and results of other investigators are cited where appropriate.

WASTEWATER APPLICATION PROGRAM

In general, it is easier to design a wastewater application program than an application program for either 4% or dewatered sludge. Wastewater contains significantly less in the way of heavy metals or pathogens than sludge. Such items are largely retained by the sludge in the treatment plant separation process. Consequently the design of a forest application site need only match the application rate to the ability of a site to store and utilize the nutrients being added and to hydraulically accept the irrigation rates. To develop some design criteria for a forest wastewater application system, a series of field experiments was undertaken at our Pack Forest site. Study plots were located on well drained glacial outwash soil. In this program, both young plantations of poplar and Douglas-fir and an established Douglas-fir forest were irrigated with 5 cm/wk of wastewater from the Metro treatment facility for the city of Seattle. To provide controls and comparisons with other land application alternatives, barren (all vegetation removed) and grass sites were also included in the experimental design. Each site was subdivided so that one section received wastewater while the remainder received an equal amount of water pumped from the adjacent river. Irrigations were conducted weekly on a year-round basis over a five-year period (1974-79).

Consequently, each plot except for the 50-year-old Douglas-fir stand received a total of over 12 meters (40 feet) of irrigation and approximately 2000 kg/ha of total nitrogen. Movement of percolating waters through the soil profile to a depth of 180 cm was carefully monitored with both tension lysimeters and a large tank lysimeter system, 3.5 meters in diameter.

A number of important conclusions were derived from this study. Phosphorus contained in wastewater was quickly and completely removed from the soil solution by a sorption reaction on the soil colloidal surfaces (Breuer et al. 1979). Thus phosphorus did not leach through any of the plots, including the one maintained in a barren state. Leaching of nitrogen showed a somewhat different pattern. As long as the added nitrogen remained in ammonium form, little if any leaching was apparent. But with the conversion of ammonium to nitrate, leaching became rapid and extensive. On those plots with a vegetative cover this loss of nitrogen was minimized through plant uptake or soil storage (Table 1; Cole 1982).

Table 1. Fate of nitrogen applied as wastewater to plots at Pack Forest from 1974 to 1979 (Schiess and Cole 1981).

	Poplar	Douglas-fir	Grass	Barren	Douglas-fir Forest
	(kilograms/hectare)				
Input	2,171	1,811	2,217	2,286	1,333
Vegetation accumulation	317	272	0	0	--[2]
Harvest	930	621	627	0	0
Leaching	253	437	271	1,252	120
Soil accumulation[1]	671	481	1,319	1,034	--

[1] Residual of input minus plant uptake and leaching loss.
[2] Not determined.

Irrigation of wastewater resulted in a dramatic increase in biomass production (Table 2). Average aboveground production on the poplar plots irrigated with wastewater was 25 t/ha, approximately 7 times higher than the river water control. The wastewater irrigation resulted in an increase in aboveground production of nearly 4 times for the Douglas-fir seedlings and 3.5 times for the grass plot over the irrigation with river water.

Table 2. Aboveground biomass production (t/ha) by young Douglas-fir and poplar plantations and grass as a result of wastewater and river water irrigation (Schiess and Cole 1981).

	Wastewater			River Water		
	Tree	Grass	Total	Tree	Grass	Total
			(tonnes/hectare)			
Poplar						
1976	8.5	8.0	16.5	1.0	1.6	2.6
1977	28.2	6.0	34.2	1.2	1.2	2.4
1978	14.8	10.1	24.9	1.8	2.1	3.9
1979	18.4	6.0	24.4	3.4	1.9	5.3
Douglas-fir						
1976	2.9	8.5	11.4	0.9	1.6	2.5
1977	4.4	8.4	12.8	1.6	0.8	2.4
1978	14.0	6.7	20.7	3.8	2.2	6.0
1979	12.8	3.4	16.4	3.6	1.3	4.9
Grass						
1976			8.5			3.5
1977			9.1			0.9
1978			10.4			3.9
1979			10.2			---

From this research it was possible to calculate renovation capacities of the five plots over the five-year period of the experiment (Table 3). It is clear from these results that forests can effectively renovate wastewater at application rates normally used in agriculture. In this case, however, applications were made over the entire year and not just during the growing season, as is typical in agricultural practices. Results from the forest plots compared very favorably with results from the plot with grass cover. Nitrate-N concentrations at 180 cm depth were nearly always below 10 ppm EPA drinking water standards. Both the forested site and poplar seedlings demonstrated excellent renovation potential, with only 10% nitrogen escapement below the rooting zone during the five-year period of application. Escapement losses from grass and Douglas-fir seedlings were only slightly higher and nitrate-N nearly always remained within the 10 ppm concentration standard.

Table 3. The renovation and retention capabilities of a gravelly soil for total nitrogen under wastewater irrigation applied to four vegetation covers and a barren control[1] (Schiess and Cole 1981).

Year Applied	Vegetation Cover				
	Barren	Grass	Poplar	Douglas-fir Seedlings	D.-fir Forest
	(kilograms/hectare)				
1975	428 (68%)	428 (96%)	428 (53%)	428 (89%)	--
1976	403 (34%)	398 (76%)	392 (94%)	320 (84%)	384 (84%)
1977	456 (75%)	443 (77%)	438 (98%)	325 (74%)	402 (97%)
1978	449 (59%)	437 (96%)	440 (99%)	359 (76%)	350 (97%)
1979	550 (39%)	509 (93%)	473 (98%)	379 (54%)	253 (83%)
Average	457 (55%)	443 (88%)	434 (88%)	362 (76%)	333 (91%)

[1]Values are kg/ha nitrogen applied, with percent retained indicated in parentheses.

It could be concluded from these studies that forested sites in Washington can be used effectively for wastewater renovation. Since forest response was very favorable, such applications have the potential of providing a significant economic return to the forest owner. An economic analysis was not made of this data, however, because of the small size of plots and limited design of the experiment. But an economic analysis was made in the case of sludge-treated plots and will be discussed in the next section.

While these results cannot be directly applied to other areas, it does appear from the literature (McKim 1978, Sopper and Kardos 1973, Sopper and Kerr 1979) that forests in general are excellent sites for the effective disposal of effluents from sewage treatment plants. The most important criteria to keep in mind are: (1) tree species with a high growth and nitrogen uptake potential should be used, (2) soil should be at least moderately well drained so that saturation and poor aeration conditions do not develop, and (3) year-round irrigation can be carried out only where the climatic conditions allow. Irrigation during periods of freezing could well cause physical damage to the forest.

SLUDGE APPLICATION PROGRAM

As mentioned above, the design of a sludge application program is somewhat more difficult and perhaps uncertain than design of a wastewater program. This is partly because of the association of pathogens and heavy metals with sludge and less with the effluent component of a treatment plant discharge. In addition, the technology of applying sludge, especially when it has been dewatered, is far more difficult than that of wastewater. The biological and chemical reactions of sludge within the soil are not as well understood. This affects the predictability of potential heavy metals mobility and the rate of nitrogen release from organic matter to available nitrogen forms. Without careful management controls, sludge can be lost from an application site through surface erosion. This is especially critical in a forestry application where slopes tend to be steeper than those found in agriculture.

A sludge application program designed for forestry must take these considerations into account. To provide an informational base for developing such an application program, research was initiated at the Pack Forest site in 1973. The studies conducted at Pack Forest enabled researchers to take advantage of the diverse soil conditions this site has to offer. Study plots were located both on well drained glacial outwash soil as well as shallow residual soils originating from andisitic parent material. A detailed account of this decade of research is available in the symposium proceedings "Municipal Sludge Application to Pacific Northwest Forest Lands" (Bledsoe 1981) and the bulletin "Use of Dewatered Sludge as an Amendment for Forest Growth" (Henry and Cole 1983). This research addressed most aspects of a sludge application program in forestry, including:

1. Environmental soundness of applying sludge to forest areas. The major focus of this research has been on nitrate leaching, heavy metal uptake by the vegetative cover and leaching through the soil profile, and residence time and distribution of pathogens associated with sludge materials.

2. Long- and short-term growth response of various forest species of the Pacific Northwest to these applications. Both newly established plantations and existing forests have been studied at Pack Forest.

3. Management alternatives, engineering problems, and general economic costs and benefits of applying sludge to forest sites.

It is clear from the results of this research that sludge can be used effectively to increase forest productivity without causing

significant environmental problems. Some of the important findings relevant to the use of sludge as a silvicultural tool in forestry are discussed below.

Public Health and Environmental Considerations

The fate of heavy metals, pathogens, and nitrates associated with sludge applications to these forested areas has been systematically studied. Although it could be argued that forests are essentially removed from the human food chain and such public health considerations are unnecessary, it is also argued that people frequent such sites, harvest food products from them, and obtain much of their drinking water from drainages originating in forested areas. Consequently a number of public health issues were considered as part of this overall research effort.

The question of heavy metal uptake, toxicity, and leaching has been addressed in a number of studies at the Pack Forest site. As a worst case example, 18% dewatered sludge from the Metro treatment facility was applied to a depth of 25 cm at four different sites. The vertical migration of cadmium, copper, nickel, lead, and zinc was followed for a period of two years. This study clearly demonstrated that there was only a minimal migration of these metals into the underlying soil despite the general acidic conditions. Nickel demonstrated the greatest mobility, migrating to a depth of perhaps 50 cm at the site with the lowest pH (approximately 4.5 in B horizon). For the remaining sites and heavy metals, vertical migration was limited to the top 10-20 cm of soil (Zasoski 1983). In general, it was found that in areas treated with sludge, heavy metal accumulation by the native vegetation could be lower than for crop species growing on sites with no sludge history (CAST 1980). It should be recognized, however, that significant differences in accumulation between native species were evident (Zasoski 1981). In addition, metal uptake from sludge-amended soils by conifer and deciduous species has been evaluated in nursery beds and found to be relatively low (Bledsoe and Zasoski 1981). Based on these observations, it appears that heavy metal uptake by native vegetation and tree species is elevated by sludge additions, but such increases are small and phytotoxicities are not evident.

Residence time and mobility of sludge-related pathogens have also been followed in our forest sites. While most of these studies have utilized fecal and total coliform bacteria as indicator organisms, virus and other pathogenic organisms have also been studied to a more limited extent. Residence time of fecal coliform in a 10 cm application of sludge is less than two years following application to forest sites. This residence time is further decreased if sludge has been applied to recently harvested sites or applied during the summer months. There

appears to be little evidence of any downward migration of the fecal coliform in the soil. Soil samples taken directly beneath sites receiving sludge have a very low fecal coliform count. In addition, this elevated occurrence of fecal coliform in the soil is short-lived, disappearing within six months. There has been no evidence of fecal coliform in the groundwater table directly beneath treated areas (Edmonds 1979).

The loss of nitrogen from sludge-treated sites is far more difficult to control. Although much of the nitrogen within sludge is in organic and ammonium forms and thus not easily leached, it can be leached following mineralization and nitrification. The rate of leaching is therefore controlled by a series of processes not only including mineralization and nitrification but also denitrification, volitilization, immobilization by soil organisms, and uptake by the vegetation. In order to establish application rates that will minimize losses through leaching and at the same time promote tree growth, it is essential to have an understanding of the rates and durations of the reactions listed above. Studies have been initiated at Pack Forest to quantify these various nitrogen transformation processes (Riekerk and Zasoski 1979, Vogt et al. 1981, Cole and Henry 1983). From these studies it would appear that a single application in excess of 40 dry t/ha could result in some nitrate leaching during the first year following application.

Stability of Sludge

As discussed in the introduction, much of the forest land, especially in the West, is located on relatively steep slopes. Since the U.S. Environmental Protection Agency has defined slopes in excess of 12% as "potentially unsuitable areas", much of our forest lands could be classified as unsuited for sludge applications. Slope studies were undertaken at Pack Forest to establish whether this limitation is applicable to forest conditions or should remain only an agricultural guideline. This research involved the application of sludge to existing slopes and to a slab that could be tilted to any desired angle. Dewatered digested sludge (17% solids) was applied to these surfaces at various depths ranging from 1 to 5 cm. It was demonstrated through this research that sludge up to 5 cm deep could be safely applied to undisturbed forest floor surfaces on slopes in excess of 23%. Sludge depths of 2.5 cm could be applied to surfaces of approximately 40% with only minimal downward movement (Henry 1983). It was concluded from this research that the slope guidelines developed for agricultural conditions are not necessarily appropriate for use in forestry. A more reasonable guideline in forestry would allow sludge applications of 2.5 cm (45 ton/ha) to slopes up to 30%.

Application Alternatives

The technology developed to apply sludge to agricultural sites is not easily transferable to forest sites. A forest application vehicle must be able to negotiate sloped and broken terrain. Even more critical, the vehicle must be able to deliver sludge into the forest, necessitating a system capable of pumping sludge a distance of 30 to 50 meters through the air. To serve this purpose, a cannon-mounted sludge application vehicle was designed and built as part of the research program at Pack Forest (Nichols 1980).

Without such a vehicle, sludge applications in forests would be limited to cleared sites with the actual applications made prior to planting. This option, extensively tested under a number of site conditions, proved to be impractical (Wilbert and Archie 1981). Unless intensive efforts are made to control unwanted weed species, high seedling mortality and poor growth result. This is due primarily to the excellent response of the unwanted plants to the sludge application, resulting in excessive competition with the tree seedlings for both moisture and space. In addition, the higher nutritional environment in the sludge-treated areas can result in selective browsing by deer on tree seedlings. Sludge-treated sites can also provide excellent habitats for voles, which -- left uncontrolled -- will girdle the young trees.

Because of these many difficulties in plantation management following a sludge application, it has been concluded from our program at Pack Forest that applications in forestry should be limited to plantations on natural stands where the trees are well established. We would recommend that the trees be a minimum of 2 meters high before they receive their first application. In this way they are well above browsing activities by deer. More important, the increased productivity that sludge provides a forest site will be reflected in the tree seedlings and not the weeds.

If the sludge is pumped over the canopy of a young forest, care must be exercised not to damage the foliage. The easiest way of accomplishing this is to limit applications to the high rainfall periods and/or the dormant stage of the growth cycle. In western Washington we recommend such applications be limited to the winter months prior to the flush of growth beginning in late April.

In well-established forests, sludge can be applied beneath the canopy and thus there is no seasonal limitation. However, it is our general conclusion from both the sludge application program and fertilization with commercial chemical fertilizers, that younger stands are more responsive. For Douglas-fir in the Pacific Northwest the most desirable age for applying sludge seems to be between 10 and 30 years.

Growth Response

From the point of view of the forest manager, the purpose of applying sludge to forested sites is to achieve a growth response. In general, we have found most of our forest sites and many of our forest species responsive to sludge applications. This experience parallels our earlier results from the addition of chemical fertilizers. It should be recognized, however, that not all species and sites have demonstrated a response. This is partly caused by secondary factors such as weed competition, deer browsing, and vole girdling, as previously discussed. It also appears that some species and some sites have a better response potential than others. This variable response is evident from our program at Pack Forest, as discussed below.

Species trials. Twelve species have been tested for their growth response potential at Pack Forest. Since these seedlings have now been growing in sludge-treated areas for three to four years, growth and mortality results can be viewed with some confidence (Table 4).

In general, an excellent response was noted for all species tested. The most dramatic growth was for a hybrid cottonwood (Populus trichocarpa x P. deltoides): after four growing seasons the height now exceeds 6.5 meters. In this trial the average height response for deciduous trees exceeded that of conifers by nearly three times. The average diameter growth responses were nearly equal (Table 4).

Application over a plantation canopy. Growth response from over-the-canopy applications is limited to only one year of observations. Results, as one might expect, are highly variable in that a number of site conditions and application periods were tested (Table 5). The poorest response (a net decrease of 30% in height growth) was found on a site where sludge had already been applied. The greatest response was noted on a residual soil derived from an andesitic bedrock material. Although we believe an over-the-canopy application can be successfully carried out and has an important potential in a forest application program, additional research is clearly needed before this can be regarded as operational. Evidence so far indicates that young plantations that are not well established should be avoided. One should also avoid applying the sludge during the growing season, because foliage will be damaged or at least rendered nonfunctional for photosynthesis unless the sludge is washed off. From our initial observations, an over-the-canopy application is more suited to a well established forest, (perhaps 10-70 years old) rather than a newly established plantation.

Application to older forests. A very successful technique of applying sludge to forests is to apply it below the canopy using our sludge-application vehicle. The percent growth responses are not as

Table 4. Diameter and height growth response over control for various tree species established in sites receiving approximately 10 cm of sludge (160 dry t/ha).

Species	Current Age	Height Response (percent)	Diameter Response (percent)
Douglas-fir	4	64	150
Western hemlock	4	370	690
Sitka spruce	4	140	225
Western redcedar	4	65	115
Cottonwood	4	320	585
Hybrid cottonwood	4	590	895
Poplar	4	1,190	---
Redwood	3	575	1,035
Grand fir	3	200	1,250
Sequoia	3	290	1,035
Basswood	3	200	125
Birch	3	900	690
All conifers		270	645
All deciduous		640	560

[1] Age since establishment

Table 5. First-year growth response for Douglas-fir receiving an over-the-canopy sludge application (2 dry t/ha).

Site	Plantation Age	Soil	Height Response (percent)	Diameter Response (percent)
Val	9	Glacial	-8	9
XAA	6	Glacial	-30	-18
Hugo	6	Residual	180	330
27	10	Glacial	10	60
M-S	5	Glacial	1	-2

dramatic as those reported above in the species trial, simply due to the size and age of the stands. But excellent responses have been noted for Douglas-fir, the only species tested thus far in this way (Table 6). The best responses were found on the lower site classes. This is consistent with results from urea applications to commercial forest lands. In the case of sludge, the growth responses are substantially larger than the 25 to 30 percent response obtained from urea fertilization (RFNRP Staff 1976).

Table 6. Average diameter growth response for Douglas-fir over a four-year period at three sites located at Pack Forest.

Treatment	Diameter Growth Response	
	cm	percent
	59-year-old stand, site IV:	
Control	.13	
Sludge	.24	85
Control, thinned	.40	
Sludge, thinned	.60	50
	53-year-old stand, site II:	
Control, thinned	.54	
Sludge, thinned	.62	15
	25 to 50-year-old stand, site V:	
Control, thinned	.67	
Sludge, thinned	1.17	75

Recommended application rates. The research on sludge applications to forests has clearly demonstrated the potential of a substantial growth response, as documented in the previous section. This research has also shown that the application of sludge can result in nitrate leaching if the uptake and storage potentials of the system are exceeded. Following the format used for determining sludge applications in agriculture (Sommers 1980), we have calculated application rates for forest conditions (Table 7). It is clear from these calculations that an application in excess of 2 cm can potentially result in a nitrate-N leaching loss exceeding 10 ppm as measured below the rooting zone. This

Table 7. Nitrogen transformations and utilization for various application rates--year 1 (conceptual model, not actual values) (Cole and Henry 1983).

Dry t/ha	cm	Organic Nitrogen (kg/ha)	Ammonia Nitrogen (kg/ha)	Mineralized Nitrogen (kg/ha)	Volatilized Nitrogen (kg/ha)	Soil Storage		Uptake (kg/ha)	Leached NO^- -N (kg/ha)	(ppm)
						Temporary (kg/ha)	Long-term (kg/ha)			
23	1.3	784	224	196	168	140	--	112	0	0
47	2.5	1,568	448	392	420	224	--	112	84	15
93	5.1	3,136	896	784	907	448	--	112	213	42
140	7.6	4,704	1,344	1,176	1,445	560	--	112	403	79
187	10.2	6,272	1,792	1,568	1,982	672	--	112	594	117
280	15.2	9,408	2,688	2,352	3,058	896	--	112	974	192
467	25.4	15,680	4,480	3,920	5,208	1,344	--	112	1,736	341

Assumptions in Table 7:

1. 18.4% dewatered sludge; 3.4% organic N, 0.9% ammonia N.
2. Organic nitrogen mineralized 20% in year 1, 3% thereafter.
3. Ammonia volatilization = 50% of available nitrogen.
4. Temporary soil storage is available nitrogen in the sludge/soil layers.
5. Long-term soil storage is clearly present but not known at present and conservatively assumed = 0.
6. First-year uptake estimated at 2/3 of 168 kg/ha/yr rate.
7. Average nitrogen leaching concentration based on 114 cm total precipitation and 64 cm evapotranspiration.

potential for loss has been verified in our research at Pack Forest (Cole and Henry 1983). Limiting forest applications to less than 40 dry metric tons/ha, and thus minimizing the potential for nitrate leaching, could at the same time limit the growth response or duration of response the forester might expect from a sludge application. It is clear that more research is needed to meet the two critical objectives of limiting nitrate losses and maximizing growth responses. If it is decided in forestry that sites cannot receive sludge more often than perhaps every five years, in all probability higher applications, resulting in temporary leaching loss of nitrate, might be recommended in order to obtain the desired growth response. It would appear from our current experience that a 40 dry t/ha application will provide a growth response for at least a five-year period. This rate of application will have only a minimal potential of affecting nitrate concentrations in the groundwaters and soil solutions of the application site, and thus will meet current regulations.

RECOMMENDATIONS AND CONCLUSIONS

Forests of this country have been largely unused for the disposal and reuse of municipal wastewaters and sludges. Research by the University of Washington and other institutions suggests that such sites represent a logical alternative to other options that have more traditionally been used by municipal sewage treatment agencies. The major disadvantage is probably the lack of an adequate information base and understanding such as is currently available in agriculture. Because of this uncertainty, some municipalities have been reluctant to select forests for this use. This is ironic in that the application of sludge and wastewater to forests is a step removed from many public health concerns.

In order for forests to become a viable alternative for land application of sludge, the basic differences between forest and agricultural sites will have to be recognized and accepted by regulatory agencies. These include:

1. The pH of most forest soils is acid, typically below 5.5. This has the potential of mobilizing heavy metals, especially those associated with sludge. Although excessive uptake rates, phytotoxicities, and leaching in forest soils have not been found, the possibility exists and additional research is recommended. The regulatory agencies will have to recognize that forest soils by nature are acidic and cannot be effectively changed by liming.

2. It is unlikely that a sludge-application program in forestry will involve annual applications such as carried out in

agriculture. In all likelihood repeat applications every five to ten years is far more probable. Consequently, these applications will be somewhat higher (probably 80+ metric tons/ha) than those usually recommended for agricultural practices. These higher applications can result in an initial flush of nitrate from the rooting zone.

3. Application rates cannot be limited to slopes of less than 8%, as is recommended for agriculture. It has been demonstrated that forest sites with slopes up to 30% can receive 80 dry tons/ha of dewatered sludge without lateral movement. If indeed a slope limitation as in agriculture is imposed on forests, very few forest sites will be available.

Society will also have to recognize that forest areas incorporating sludge cannot be used in some traditional ways because of the residence period of pathogens and the uncertainty of the heavy metal issue. For example, mushroom harvesting might have to be limited because mushrooms tend to accumulate heavy metals.

Although no food chain problems have been found that originate in a forest application site, unresolved issues remain regarding browsing by game animals.

It is our conclusion, after ten years of research applying sludge to forest sites, that greater use should be made of forests in land application programs. Not only should it be possible to manage such programs in a sound and environmentally safe way, but also forests can dramatically increase in growth rates. Such advantages could especially benefit very low-site land that currently does not support commercial forestry.

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PATHOGENS

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INTRODUCTION AND PURPOSE

As part of the 1973 conference on "Recycling Municipal Sludges and Effluents on Land" (National Assoc. State Univ., 1973), a series of workshops were held to identify what was known about municipal wastewater effluent and sludge application to the land and what research was needed for the safe and successful land application of such wastes. The workshop on public health aspects addressed a number of issues and made a list of major research needs. Sufficient data were felt lacking in a number of key areas, including the epidemiology of human populations, development of capability of monitoring for pathogens, and the dispersion and survival of pathogens during land application. This report will cover advances in understanding the last three topics.

The paucity of information on health problems associated with land application of domestic wastes may reflect either the absence of a problem, lack of intensive surveillance, or the insensitivity of present epidemiological tools to detect recurrent small-scale incidents of disease. Often the low fecal carriage rates of agents of infectious disease and the low background of enteric disease in the United States has been cited as further evidence that the potential of public health hazard as a result of direct or indirect reuse of wastewater is minimal. It should be realized, however, that levels of enteric disease in the U.S. are low primarily because of good sanitation, personal hygiene, and a network of sanitary engineering works. As a result of this low exposure to pathogens, the population at large may have become highly susceptible to even small numbers of pathogens.

Waterborne outbreaks of disease are no longer on the decline in this country. A total of 50 waterborne outbreaks occurred during 1980, increasing the annual average of outbreaks to 39 for the five-year period from 1976-1980. This represents more than 50% increase over the 1971-1975 average of 24. The five-year averages have steadily increased from an annual average of ten during 1951-1955. Before that period, the trend

was declining (Lippy, 1981). It should be emphasized that waterborne disease outbreaks, particularly in individual systems, are notoriously underreported.

Given the difficulty of performing epidemiologic studies and inadequacy of waterborne disease reporting, evidence for potential health problems related with land application of wastewater will often rest on our ability to assess the fate of human pathogens in the environment.

OCCURRENCE OF PATHOGENS IN RAW AND TREATED SEWAGE EFFLUENTS

A. Bacteria

Pathogenic microorganisms such as bacteria, viruses, protozoa, and parasitic worms are almost always present in domestic sewage. The number and types of organisms present in sewage, however, varies from community to community depending upon urbanization, population density, sanitary habits, season of the year, and rates of disease in the contributing community (Hoadley and Goyal, 1976).

The most common bacterial pathogens associated with sewage are *Salmonella*, *Shigella*, *Vibrio*, and *Campylobacter* (Table 1). *Salmonella* occur frequently in sewage and are responsible for infecting up to 2 million people every year. Bacillary dysentery is caused by *Shigella* spp. Although *Escherichia coli* is considered a non-pathogenic organism, certain enteropathogenic strains may cause diarrhea, particularly in travelers. *Campylobacter jejuni* is a recently-recognized cause of acute gastroenteritis with diarrhea and is throught to be as prevalent as *Salmonella* and *Shigella*. The first waterborne disease outbreak of Campylobacterosis occurred in 1978 in Vermont and involved 3000 cases (Vogt et al., 1982). Waterborne transmission of *Yersina enterocolitica* resulting in yersiniosis, an acute gastroenteritis, has also been demonstrated during the last decade (Eden et al., 1977).

The United States was free from *Vibrio cholerae*, the causative agent of cholera, from 1911 to 1972 (Kowal, 1982). Since 1973, however, 31 cases of cholera have been documented along the Gulf Coast. The strains from all these cases appear essentially identical, suggesting that the toxigenic *V. cholerae* O1 has persisted in that region for at least eight years. Extra efforts should, therefore, be made to keep tract of this potential problem.

The concentration of pathogenic bacteria will vary with the incidence of disease within a community. Concentration of *Salmonella typhi* in raw wastewater range from 600 to 10^6/100 ml (Crites and Uiga, 1979).

Table 1. Bacteria and parasites in sewage and sludge

Group	Pathogen	Disease Caused
Bacteria	Salmonella (1700 types)	Typhoid, paratyphoid, salmonellosis
	Shigella (4 spp.)	Bacillary dysentery
	Enteropathogenic Escherichia coli	Gastroenteritis
	Yersinia enterocolitica	Gastroenteritis
	Campylobacter jejuni	Gastroenteritis
	Vibrio cholerae	Cholera
	Leptospira	Weil's disease
Protozoa	Entamoeba histolytica	Amebic dysentery, liver abcess, colonid ulceration
	Giardia Lamblia	Diarrhea, malabsorption
	Balantidium coli	Mild diarrhea, colonic ulceration
Helminths	Ascaris lumbricoides (Round worm)	Ascariasis
	Ancyclostoma duodenale (Hookworm)	Anemia
	Necator americanus (Hookworm)	Anemia
	Taenia saginata (Tapeworm)	Taeniasis

Little is currently known of the concentration of pathogens of emerging significance such as Yersinia enterocolitica and Campylobacter jejuni.

B. Viruses

More than 110 different virus types may be present in raw sewage (Table 2). The list of pathogenic human enteric viruses has continued to grow during the last decade. Rotaviruses are now recognized as a major cause of childhood gastroenteritis, sometimes resulting in dehydration and death of infants. Rotaviruses also cause diarrhea in adults and three waterborne outbreaks have now been reported (Zamotin et al., 1981; Sutmoller et al., 1982; Lycke et al., 1978).

The Norwalk viral agent has been demonstrated to be the cause of numerous water- and food-borne outbreaks of epidemic gastroenteritis (Kaplan et al., 1982; Gunn et al., 1982). Astroviruses, caliciviruses, cornavirus-like particles, and new serotypes of Adenoviruses have been found to be associated with human gastroenteritis. Other viral agents capable of causing gastroentertis have been recognized but are less well characterized. Laboratory methods are currently not available to study many of these agents and they await further characterization.

The concentration of enteric viruses in raw sewage is thought to vary widely depending on the incidence of disease within the community, time of year, level of hygiene in the community, etc. Thus, the concentration of viruses reported in sewage varies widely (Sattar, 1978). Concentrations reported in the literature range from 0-463,500/liter. The highest concentration reported in the U.S. was 42,000/liter (Ridinger et al., 1982). It is generally felt that concentrations of enteric viruses and other pathogens are lower in the United States than many developing nations. Unfortunately, most of the previously published information is concerned with the detection of enteroviruses and in some cases reoviruses. We know little on the concentration of other viruses in domestic wastes. Akin and Hoff (1978) have estimated that the average concentration of enteric viruses in raw sewage in the United States varies between 1,000 to 10,000 virus units/liter.

C. Protozoa and helminths

Little attention has been given to the presence of parasites in wastewater because of the popular impression that the prevalence of parasite infection in the U.S. is low (Larkin et al., 1978). However, because of the increasing recognition of parasite infections in the U.S., the return of military personnel from countries with a high parasite disease prevalence, and the existence of resistant stages of the organisms, a consideration of parasites is warranted (Kowal, 1982).

Table 2. Human enteric viruses in sewage

Virus	Number of Types	Diseases Caused
Enteroviruses:		
Poliovirus	3	Meningitis, paralysis, fever
Echovirus	31	Meningitis, diarrhea, rash, fever, respiratory disease
Coxsackievirus	23	Meningitis, herpangina, fever, respirator disease
Coxsackievirus	6	Myocraditis, congential heart anomalies, pleurodynia, respiratory disease, fever, rash, meningitis
New enteroviruses (Types 68-71)	4	Meningitis, encaphalitis, acute hemorrhagic conjunctivities, fever, respiratory disease
Hepatitis Type A (enterovirus 72?)	1	Infectious hepatitis
Norwalk virus	1	Diarrhea, vomiting, fever
Calicivirus	1	Gastroenteritis
Astrovirus	1	Gastroenteritis
Reovirus	3	Not clearly established
Rotavirus	2	Diarrhea, vomiting
Adenovirus	40	Respiratory disease, eye infections

Of the common protozoa which may be found in wastewater only three speices are of major significance for transmission of disease to humans through wastewater: Entamoeba histolytica, Giarda lambia, and Balantidium coli (Table 1). All of these organisms are known to cause mild to severe diarrhea. There has been a steady increase in the number of waterborne outbreaks of giardiasis in the United States since 1971 (Craun, 1979). Growing recognition of this disease has been responsible for increased surveillance, investigation, and reporting by U.S. public health authorities. Investigations of disease outbreaks suggest that coliforms may not be adequate indicators of these organisms in drinking waters which receive only chlorination before distribution (Lippy, 1978). Disinfection studies have shown that Giardia cysts are considerably more resistant to chlorine than indicator bacteria or even enteric viruses (Jarrol et al., 1980). Levels of Giardia cysts in feces from infected persons may be as high as 1-5 x 10^6 cysts per gram and the cyst level in domestic raw sewage has been estimated at about 10^4 cysts per liter.

Waterborne outbreaks of E. histolytica and B. coli are known to occur, but there is no recent evidence that waterborne transmission has occurred. The last reported waterborne outbreak of E. histolytica in the U.S. was in 1953.

Although helminth infections are still prevalent in the U.S. population, the occurrence of disease due to these agents in the U.S. has been extremely low during the last few decades. The presence and levels in wastewater of any helminth eggs depend on the levels of disease in the population. Foster and Engelbrecht (1973) suggested a value of 66 helminth ova per liter of raw wastewater, and Larkin et al. (1978) cited values of 15-27 Ascaris eggs/liter and 6-2 helminth eggs/liter in primary effluent.

PATHOGEN REMOVAL DURING SEWAGE AND SLUDGE TREATMENT

A. Sewage treatment

Feachem et al. (1980) reviewed the literature on pathogen removal by various sewage treatment processes. Table 3 has been adapted from his results. In interpreting this table, Feachem et al. (1980) warn that "it is not necessary to dwell on trivial differences, as between 92.3% removal and 97.8% removal, but to look at orders of magnitude". They further stated that to talk of percent removal is misleading because a 99% removal of pathogens from raw sewage containing 10^5 pathogens per liter, will produce an effluent which still contains 10^3 pathogens per liter. This level may still be of great public health concern, depending on how the effluent is going to be used.

Table 3. Percent removal of pathogens by various sewage treatment processes*

Treatment	Enteric viruses	Bacteria	Protozoan cysts	Helminth eggs
1. Primary Sedimentation	0-30	50-90	10-50	30-90
2. Trickling † Filter	90-95	90-95	50-90	50-95
3. Activated † Sludge	90-99	90-99	50	50-99
4. Oxidation † Ditch	90-99	90-99	50	50-99
5. Waste stabilization ponds. Three cells; with ≥ 25 days retention	≥99.99	≥99.99	100	100
6. Septic Tanks	50	50-90	0	50-90

* Adapted from Feachem et al. (1981)

† With sedimentation, sludge digestion, and sludge drying

As efficient as it may be, sewage treatment processes cannot be expected to remove/inactivate all of the pathogens present. Disinfection of treated wastewater is, therefore, practiced to insure further inactivation of microorganisms. In the United States, chlorination is practically the only process used for disinfection of wastewater. Unfortunately, however, there is a great variability in resistance to chlorine among different microorganisms. It is generally agreed that bacteria are much more susceptible to chlorine than are viruses and protozoan cysts. Also, chlorine may be very effective against microorganisms cultivated in the laboratory under artifical conditions, but it may not be as effective on naturally occurring strains of bacteria and viruses.

Approximately 90-95% of the enteric bacteria in sewage are reported to be removed by activated sludge process (Table 3). Coagulation with alum or lime is considered to be generally efficient for virus removal. In laboratory studies, 3-4 log reduction of viruses are common following lime treatment at pH 11. In field studies, however, viruses were isolated from lime sludge and lime-treated effluent (Sattar and Ramia, 1978). Other tertiary treatments such as ferric chloride-polyelectrolyte flocculation, sand or granular filtration, reverse osmosis, and carbon adsorption have been found to significantly reduce the level of pathogens.

Studies indicate that bacteria and viruses are not removed effectively from wastewaters during primary treatment (Gerba, 1981); removal of viruses during secondary treatment (active sludge) is dependent largely upon virus adsorption to solids. Since rotavirus adsorbs poorly to activated sludge flocs it can be speculated that wastewater treatment processes that are highly effective in the removal of enteroviruses may not be as effective in removing rotaviruses and reoviruses. Even within the enterovirus group, virus adsorption to activated sludge was found to be both type- and strain-dependent (Gerba et al., 1978). It stands to reason, therefore, that different viruses will have different removal characteristics during activated sludge treatment.

We still know little about the effectiveness of these processes for the removal of Hepatitis A and Norwalk agent or the bacterial pathogens, *Campylobacter jejuni* and *Yersinia enterocolitica*.

Reported removals of protozoan cysts are usually less than those of bacteria and viruses (Table 3). Wastewater stabilization ponds have been reported to have high removals, but this may be due to their concentration in sludge rather than complete inactivation (Kowal, 1982).

Since helminth eggs are denser than water, conventional primary treatment (sedimentation) is fairly efficient in their removal (Kowal, 1982). Again, wastewater stabilization ponds accomplish excellent degrees of helminth egg removal. Stabilization ponds at a land treatment site in San Angelo, TX, resulted in complete removal of helminths (Kowal, 1982).

Again, sludge or pond sediments resulting from these processes will have high densities of viable helminth eggs, and will require proper treatment before utilization.

B. Sludge treatment

Development of new analytical methods for the detection of enteric viruses in sludges has provided new information on the concentration of viruses in sludges and the effectiveness of sludge treatment. Reported concentration of enteric viruses (essentially enteroviruses and reoviruses) is shown in Table 4. It is now clear that naturally occurring enteric viruses can survive and be detected in aerobic and anaerobically digested sludges, although removals may be significant during these processes (Table 5). Detailed studies of the mechanisms of virus inactivation during sludge treatment (Ward and Ashley, 1977a, 1978) have shown ammonia and detergents play a significant role. Drying and loss of moisture from sludge can result in significant inactivation of viruses (Ward and Ashley, 1977b). Virus decline is low when the sludge solids is raised from 5 to 40%. However, there is a 4 to 5 $\log_{10}$ decrease in virus numbers when the final sludge solid percentage is above 90. Dewatering causes the release of viral RNA and inactivation appears to be due to the dewatering process itself.

In sludge composting, sludge is mixed with other organic materials such as wood chips or leaves and allowed to decompose for a period ranging from 3-4 weeks. Aerobic conditions are maintained either by forcing or pulling air into the compost pile or by regularly turning the pile. Composting, being a thermophilic process with temperatures ranging from 60°C to 70°C under ideal conditions generally results in inactivation of pathogenic microorganisms. Protozoan cysts, helminth eggs, pathogenic bacteria are effectively inactivated during this process. Experiments with model viruses (bacterial virus f2) have revealed that a properly operated windrow compost system may result in total virus inactivation in approximately 50 days (Bitton, 1980). Enteric virus monitoring of a windrow compost system has revealed their presence during the windrow phase but none after the curing period (Bitton, 1980). Viruses and other pathogens are inactivated much more rapidly by forced-aeration pile methods. A monitoring procedure has been developed for the Beltsville Aerated-Pile method based upon achieving 55°C in the lowest temperature portions of the pile for a period of 2.5 days to insure pathogen destruction (Burge et al., 1981). This procedure along with monitoring for salmonellae has been adopted by the health departments of several states and is being considered by others as the criteria to be met by composting operations for unrestricted release from a pathogen hazard standpoint (Personal communication, W.D. Burge, 1983).

Recent studies have also been conducted on the concentration of helminths in domestic sludges in the U.S. (Table 6). Significant numbers of

Table 4. Reported concentration of enteric viruses in sludges

Type of Sludge	Concentration	Location	Reference
raw	5-145 pfu/ml	Cincinnati	Brasher and Ward (1982)
mixed liquor suspended solids	5 $TCID_{50}$/g	Florida	Farrah et al. (1981a)
aerobically digested	1.7 to 5.2 $TCID_{50}$/g	"	"
lagooned sludge applied to land	0.02 to 4.6/g	"	"
raw	6.9 to 215 pfu/g	"	Turk et al. (1980)
digested	0.2 to 17 pfu/g	"	"
lagooned	1.2 pfu/g	"	"
raw	17.9 $TCID_{50}$/100 ml	Hungary	Palfi (1973)
anaerobic-mesophilic	0.85 $TCID_{50}$/100 ml	"	"
raw	40 to 1,419 pfu/g	England	Goddard et al. (1981)
anaerobic-mesophilic digestion	6 to 210 pfu/g	"	"
aerobic-thermophilic digestion	10 to 65 pfu/g	"	"
anaerobic high rate digestion	1.1 to 17 pfu/g	Texas	Sagik et al. (1980)
anaerobically digested lagoon	1.2 pfu/g	"	"
anaerobic digestion	5.0-6.7 pfu/g	"	"
raw	141-1,060 pfu/100 ml	"	Kabrick et al. (1979)
anaerobic-mesophilic	4-100 pfu/100 ml	"	"
aerobic-thermophilic	0-14 pfu/100 ml	"	"
anaerobic-mesophilic	0.8 pfu/ml	Wisconsin	Cliver (1975)
aerobic digestion	14 to 260 $TCID_{50}$/g	Florida	Farrah et al. (1981b)
anaerobic digestion	2 to 7 $TCID_{50}$/g	"	"

Table 5. Report on removals of naturally occurring enteric viruses during sludge treatment

Treatment	Reduction (%)	Reference
Anaerobic-mesophilic	88 - 98.6	Kabrick et al. (1979)
Thermophilic-aerobic	>99.7	
Anaerobic-mesophilic	28 - 85	Goddard et al. (1981)
Aerobic-thermophilic	77%	
anaerobic-mesophilic	95%	Palfi (1973)

Table 6. Parasite concentration in primary and secondary sludge as compared to treated sludge

Parasite	Type of Sludge[1]	Number of Viable and Non-viable Eggs/kg Dry Weight of Sample	
		Average[2]	Percent Viable Eggs
Ascaris spp. (human and pig roundworm)	Primary & Secondary	9,700	45
	Treated	9,600	69
Trichuris trichiura (human whipworm)	Primary & Secondary	800	50
	Treated	2,600	48
Trichuris vulpis (dog whipworm)	Primary & Secondary	600	90
	Treated	700	64
Toxocara spp. (dog and cat roundworm)	Primary & Secondary	1,200	88
	Treated	700	52

[1] Primary and secondary sludges include sludges from primary clarification, Imhoff digestion, activated sludge, contact stabilization, and extended aeration. Treated sludges include sludges from mesophilic aerobic and anaerobic digestion, vacuum filtration, centrifugation, lagoons and drying beds.

[2] Numbers rounded off to nearest 100.

Modified from Reimers et al. (1981).

these parasites may survive both aerobic and anaerobic sludge treatment. As in the case of viruses, sludge drying (i.e., loss of moisture) has a great influence on the inactivation of parasites in sludges (Reimers et al., 1981).

Sludge liming has been studied as a method of reducing pathogenic microorganisms, but has been found to be unreliable in the complete inactivation of viruses (Sattar and Westwood, 1979) and parasites (Reimers et al., 1981).

Radiation processing of sludge by exposure to high-energy electrons produced by an electron accelerator or radiation sources appears to be highly effective against pathogenic microorganisms and helminths (Epp and Metz, 1980).

MONITORING CAPABILITIES FOR ASSESSING PATHOGEN FATE AFTER LAND DISPOSAL

A great deal of progress was made in the last decade in the development of methodology for detecting viruses in the environment. Reliable methods are now available for the detection of enteroviruses in water, sludge, soil, and plant material (Gerba and Goyal, 1982). We now have the ability to judge the fate of many types of enteric viruses during the land application of municipal wastewater and sludge. Further research is needed on the development of methods for the isolation of more fastidious viruses such as rotaviruses, Norwalk agents, hepatitis, etc. from the environment. Many of these agents are difficult to grow or cannot grow in conventionally used tissue culture systems. But, it would appear that once methods are available to grow these agents in the laboratory only minor modification of current strategies for isolating of viruses would be necessary.

Methods for isolation and quantitation of bacterial indicators such as coliform and fecal coliform bacteria are used for assessing their fate after land disposal of domestic wastes. There is a need for the development of methods for the direct isolation of bacteria pathogens of emerging significance such as *Yersinia enterocolitica* and *Campylobacter jejuni*.

PATHOGEN FATE DURING LAND APPLICATION

A. Soil and groundwater

The potential for the transmission of disease by contaminated groundwater in the United States is well recognized (Lippy, 1981). A total of 673 waterborne outbreaks affecting 150,268 persons was reported in the

United States from 1946 to 1980 (Craum, 1979; Lippy, 1981). Of these, 295 (44%) outbreaks involving 65,173 cases were attributed to contamination of groundwater. But thus far, no outbreaks have been attributed to wastewater land application systems.

Currently, 20% of the total water consumed in the U.S. is drawn from groundwater sources and it is estimated that this usage will increase to 33% in the year 2000 (Freeze and Cherry, 1979). According to Duboise et al. (1979) over 60 million people in the U.S. are served by public water supplies using groundwater, and about 54% of rural population and 2% of the urban population obtain their water from individual wells. Since groundwater is often used for human consumption without any treatment, it is imperative to understand the fate of pathogenic microorganisms during the land application of wastewater and sludge.

The fate of pathogenic bacteria and viruses in the subsurface will be determined by their survival and their retention by soil particles. Both survival and retention are now believed to be largely determined by climate, nature of the soil, and nature of the microorganism. Climate will control two important factors in determining viral and bacterial survival: temperature and rainfall. The survival of microorganisms is greatly prolonged at low temperature; below 4°C they can survive for months or even years (Gerba et al., 1975). At higher temperatures inactivation or die-off is fairly rapid. In the case of bacteria, and probably viruses, the die-off rate is approximately doubled with each 10°C rise in temperature between 5°C and 30°C (Reddy et al., 1981). Above 30°C, temperature is probably the dominant factor determining virus survival time. Rainfall mobilizes previously retained bacteria and viruses and greatly promotes their transport in groundwater. Several studies have shown that the greatest degree of drinking water well contamination occurs after periods of heavy rainfall (Dewalle et al., 1980; Lewis et al., 1980; Barrell and Rowland, 1979).

The nature of the soil will also play a major role in determining survival and retention. Soil properties influence moisture-holding capacity, pH and organic matter - all of which will control the survival of bacteria and virus in the soil. Other soil properties such as particle size, cation exchange capacity, and clay content will influence retention. Resistance of microorganisms to environmental factors will vary among different species as well as strains. Bacteria are believed to be removed largely by filtration processes while adsorption is the major factor controlling virus retention (Gerba et al., 1975).

The following sections are a summary of the recent state of knowledge on factors currently believed to influence microbial persistence and transport in the subsurface.

Table 7. Factors that influence the survival of bacteria and viruses in soil

Factor	Bacteria	Viruses
Temperature	Longer survival at low temperature; longer survival in winter than in summer.	
pH	Shorter survival time in acid soils (pH 3-5) than in alkaline soils.	May indirectly affect virus survival by controlling their adsorption to soils.
Cations		May also indirectly influence virus survival by increasing their adsorption to soil (viruses appear to survive better in the sorbed state).
Dessication and soil moisture	Greater survival time in moist soils and during times of high rainfall. Survival time is less in sandy soils with lower waterholding capacity.	One of the most proven detrimental factors. Increased virus reduction in drying soils.
Sunlight	May be detrimental at the soil surface.	
Antagonism from soil microflora	Increased survival time in sterile soil.	No clear trend with regard to the effect of soil microflora on viruses.
Organic matter	Increased survival and possible regrowth when sufficient amounts of organic matter are present.	Unknown

Modified from Gerba and Goyal (1983).

Table 8. Die-off rate constants ($\log_{10}/day^{-1}$) for enteric microorganisms in soil.*

Microorganisms	Avg.	Max.	Min.	No. of Observations
Escherichia coli	0.92	6.39	0.15	26
Fecal coliforms	1.53	9.10	0.07	46
Fecal streptococci	0.37	3.87	0.05	34
Salmonella sp.	1.33	6.93	0.21	16
Shigella sp.	0.68	0.62	0.74	3
Enteroviruses	0.10	0.16	0.04	4

* Modified from Reddy et al. (1981).

Table 9. Reported die-off rate constants (day^{-1}) for viruses and bacteria in groundwater

Microorganism	Die-off rate (log_{10}/day^{-1})
Poliovirus 1	0.046 0.21 0.77
Coxsackievirus B3	0.19
Rotavirus SA-11	0.36
Coliphage T7	0.15
Coliphage f2	1.42 0.39
Escherichia coli	0.32 0.36 0.16
Fecal streptococci	0.23 0.24 0.03
Salmonella typhimurium	0.13 0.22

From Bitton and Gerba (1983)

1. Bacteria: At the turn of the century, it was found that the eating of raw vegetables grown on soil fertilized with raw sewage resulted in outbreaks of typhoid fever. As a result, the survival of enteric bacteria in soil systems has been extensively studied. There are several major reviews on the survival of enteric bacteria in soil (Rudolfs et al. 1950; Sepp, 1971; Foster and Engelbrecht, 1973). Most enteric bacterial pathogens die-off very rapidly outside of the human gut, whereas indicator bacteria such as Escherichia coli will persist for longer periods of time. Survival times among different types of bacteria and viruses vary greatly and are difficult to assess without studying each type individually. In most cases, it appears that 2-3 months is sufficient for reduction of pathogenic bacteria to negligible numbers once they have been applied to the soil, although survival times as long as 5 years have been reported (Rudolfs et al., 1950). Factors known to influence bacterial and viral survival in the soil are listed in Table 7.

A simple conceptual model based on the current state of knowledge on indicator and pathogen die-off has been described by Reddy et al. (1981). Microbial die-off was described by assuming first order kinetics. First order die-off rate constants (k) were calculated from the literature for enteric microbial die-off in soil-water systems. Correction factors were presented to adjust constants for changes in temperature, moisture and pH of the soil. Average die-off rate constants for selected microorganisms are shown in Table 8. In the article by Reddy et al. (1981), data on die-off of viruses during anaerobic digestion were used. Only data on virus die-off in soil systems is shown in Table 9. These values were obtained from various studies and represent an average value of several soil and environmental variables. Such an approach could prove useful for estimating microbial survival in soil-water systems, but a greater data base is needed especially for viruses and other pathogenic bacteria. Also, most of our data base on microbial survival is in soil-water systems and not in groundwater.

The straining or filtration of bacteria at the soil surface is the major limitation in their travel through soils, and they are largely removed within about 50 cm of the soil surface. Movement is greater in coarse sandy or gravelly soils (Gerba et al., 1975). Adsorption of bacteria also plays a role being increased by the presence of clay-sized particles, high cation concentration, and low pH. This adsorption is reversible, and bacteria can be released and moved down the soil profile by rainfall events which lower the conductivity of the soil water (Lewis et al., 1980).

Kowal (1982) has recently summarized the information on bacterial penetration of soil and concluded overland flow and slow rate systems pose little threat to groundwater. The exception being the presence of shallow groundwater tables, which should be avoided. Current information suggest that bacteria may penetrate 10 m vertically and up to 450 m laterally. These distances are highly site specific and travel distances are generally less than the maximum reported.

Land disposal of digested sludges have indicated little impact of bacterial contamination of groundwater, provided that the groundwater table is not too high and the soil is well-drained (Zenz et al. 1976; Liu, 1982).

2. Viruses: Virus survival in soil is not only important when decisions must be made on how long a period of time must be allowed after last application before permitting access to people and/or animals and crop harvesting, but the longer viruses survive at the surface the greater the opportunity they have for being desorbed and moving down toward the groundwater. Factors affecting virus survival are summarized in Table 7. It is believed that most virus inactivation occurs in the top few centimeters of soil where drying and radiation forces are maximal. Hurst et al. (1980) concluded that temperature and adsorption to soil appear to be the most important factors affecting virus survival. The persistence of viruses and bacteria which survive surface forces and enter groundwater is not well studied. Limited information is available on the die-off of enteric bacteria and viruses in groundwater. Table 9 summarizes the current state of knowledge on die-off rates in groundwater. Enteric bacteria and viruses persist longer in groundwater than surface waters (Keswick et al., 1982). Kowal (1982) concluded that approximately one hundred days appears to be the maximum survival time of enteric viruses in soil, unless subject to very low temperatures (4°C), which prolong survival beyond this time.

Only limited information still exists on viruses in sludge soil mixtures. Damgaard-Larsen et al. (1977) reported that it took 23 weeks during a normal Danish winter to inactivate 10^6 $TCID_{50}$/g of coxsackievirus B3. In warmer and more humid Florida, Farrah et al. (1981a) reported a two $\log_{10}$ drop in the titer of indigenous viruses in sludge amended soils within 9 days. In controlled experiments using added virus and columns with cores of soil they found that 35 days were required for a 4-$\log_{10}$ reduction of poliovirus 1 during the hot and wet summers of northern Florida. It appears that virus survival in sludge-amended soils is controlled primarily by desiccation and soil temperature (Bitton et al., 1981; Hurst and Gerba, 1979).

Adsorption is believed to be the major mechanism by which viral removal occurs during the land application of wastewater effluents. Factors which may influence virus movement into groundwater are listed in Table 10. Since adsorption is a surface phenomenon, soils with a high surface area, i.e., those with a high clay content, would be expected to have the greatest removal capabilities. Although the exact physical-chemical reasons for virus adsorption are still incompletely understood, it appears that adsorption is increased by high exchange capacity, surface area, low pH, and increased cation concentration (Gerba and Lance, 1980). As the flow rate, or application rate, increases, virus removal declines (Wang et al., 1981). Certain organics, such as humic and fulvic acids (Bitton et al., 1976; Bixby and O'Brien, 1979) may compete with

Table 10. Factors that may influence virus movement to groundwater*

Factor	Comments
Soil type	Fine-textured soils retain viruses more effectively than light-textured soils. Iron oxides increase the adsorptive capacity of soils. Muck soils are generally poor adsorbents.
pH	Generally, adsorption increases when pH decreases. However, the reported trends are not clear-cut due to complicating factors.
Cations	Adsorption increases in the presence of cations (cations help reduce replusive forces on both virus and soil particles). Rainwater may desorb viruses from soil due to its low conductivity.
Soluble organics	Generally compete with viruses for adsorption sites. No significant competition at concentrations found in wastewater effluents. Humic and fulvic acid reduce virus adsorption to soils.
Virus type	Adsorption to soils varies with virus type and strain. Viruses may have different isoelectric points.
Flow rate	The higher the flow rate, the lower virus adsorption to soils.
Saturated vs. unsaturated flow	Virus movement is less under unsaturated flow conditions.

*From Bitton and Gerba (1983)

viruses for adsorption sites on soil particles, resulting in decreased virus adsorption or even elution of already adsorbed viruses. Little differences have been found in virus removal between secondary or primary effluent (Gerba and Lance, 1978). These results suggest that adsorption of virus and movement through soil is not affected by the higher organic content of primary effluent.

The degree of virus adsorption to soils has been found to differ greatly among virus types, strains, and soils (Goyal and Gerba, 1979). Differences in adsorption among strains of the same virus type may be due to differences in configuration of proteins in the outer capsid of the virus, which affects the net charge on the virus. This affects the electrostatic potential between virus and soil, which, in turn affects the degree of interaction between the two particles. Thus, it appears that no one virus can be used as the sole model for determining the adsorptive behavior of viruses to soils and that no single virus can be used as the model for determining viral adsorptive capacity of all soil types.

Moore et al. (1982) recently demonstrated that an inverse correlation between virus adsorption and substrate capacity for cationic polyelectrolyte adsorption exists. Thus, adsorption of polyelectrolyte may be useful in predicting the fate of viruses during land application of sewage effluents and sludges.

Even though there have been no reports of disease outbreaks associated with land treatment of sewage wastes, there are a growing number of studies concerning the detection of viruses in groundwater after wastewater application to land or direct groundwater recharge. These studies are summarized in Table 11.

Wellings et al. (1974) first demonstrated viruses in the subsurface after spray irrigation of secondary sewage effluent. The wastewater was applied onto an Imolokee sand containing little or no silt or clay. Of particular interest in this study was that viruses survived chlorination, sunlight, spraying, and percolation through 3-6 m of sandy soil; furthermore, after a period of heavy rains, a burst of viruses was detected in samples that had previously been negative. These studies demonstrated that soil type, rainfall, and other factors can affect viral movement into groundwater, and that viruses are capable of surviving long periods which, when combined with the ability to move long distances laterally, could lead to wide dispersal through an aquifer.

Vaughn and Landry (1977) and Vaughn et al. (1978) reported isolations of viruses from four groundwater recharge sites, from a stormwater recharge basin, and from groundwater under a sanitary landfill in New York. These sites have soils of coarse sand, fine gravel, and 1-2% silt. At

Table 11. Isolation of viruses beneath land treatment sites.

Site Location	Type of Site[a]	Virus Types	Maximum distance of virus migration (m)	
			Depth	Horizontal
St. Petersburg, FL	S	Polio 1; coxsackie B4; echo 7	5	-
Gainesville, FL	S	Coxsackie B4; Polio 1, 2	3	7
Lubbock, TX	S	Coxsackie B3	30.5	-
Kerrville, TX	S	U[b]	1.4	-
Muskegon, MI	S	U	10	-
San Angelo, TX	S	U	27.5	-
East Meadow, NY	R	Echo 12; U	11.4	3
Holbrook, NY	R	Echo 6, 21, 24, 25; U	6.1	45.7
Sayville, NY	R	U	2.4	3
Twelve Pines, NY	R	Polio 2; U	6.4	-
North Masaqequa, NY	R	Echo 11, 23: Coxsackie A16	9.1	-
Babylon, NY	R	Coxsackie B3; U	22.8	408
Ft. Dvens, MA	R	U	28.9	183
Vineland, NJ	R	Polio; coxsackie B3; echo	16.8	250
Lake George, NY	R	Phage	45.7	400
Phoenix, AZ	R	Coxsackie B3	18.3	3
Dan Region, Israel	R	Polio 1, 2, 3	31-67	60-270

[a]R = rapid infiltration; S = slow rate infiltration

[b]U = unidentified

From Bitton and Gerba (1983)

the groundwater recharge sites, viruses were recovered at depths up to 11.4 m and at distances up to 45.7 m from the injection point of secondary or tertiary chlorinated effluent. As much as 22-33% of the 100-gal. samples at the four sites were positive for viruses, with concentrations of 1.3-10.6 PFU/gal. In order to reach the groundwater, viruses moved through 5.5-9 m of the overlying soil.

Moreover, at the Twelve Pines site, viruses were discovered in groundwater under basins where effluent seeded with viruses was applied at rates of 6-100 cm/h. The slower infiltration rates were more effective in removing the viruses, suggesting that site management is important.

Schaub and Sorber (1977) reported on a study of viruses in groundwater under rapid infiltration cells at Ft. Devens, Mass. The soil consisted of silty sand and gravel underlain by bedrock. The groundwater contained viruses at depths of 29m and lateral distances of 183 m, with concentrations of about 8.3% of the applied effluent.

At the Vineland, N.J., rapid infiltration site (Koerner and Haws, 1979) primary effluent was applied to Cohansey sand and coarse gravel. Viruses were found at 16.8 m depths and up to 250 m lateral distances in 19 or 40 samples. Total coliforms and fecal coliforms were consistently at depths up to 6 m beneath the recharge basins. Total coliforms also occasionally occurred at greater depths and downstrean. In contrast, no fecal coliforms were found in samples taken below 9.1 m and coliforms occurred only once in a shallow well 50 m down-gradient. Thus, viruses penetrated deeper into the ground and moved longer distances than did the coliforms.

Viruses in groundwater at other recharge sites have been studied with varying success. At the Flushing Meadows site near Phoenix, AZ (Gilbert et al., 1976) it was found that a fine loamy sand over coarse sand and gravel effectively removed viruses. Laboratory studies confirmed that this soil was an excellent adsorber. No viruses were detected in any of the samples of renovated water, even though levels of 158-475 PFU/mL were detected in the effluent applied. However, coliform organism were detected in the renovated water, suggesting that the removal mechanisms must have been different for viruses and bacteria, and that viruses may have been present but below the detection limit. Since this site is no longer in existence, these findings cannot be confirmed. However, since then, virus has been detected in a sample from an 18.3 m-deep well at a nearby land application site. The isolate was identified as coxsackievirus B3.

At two land treatment sites where sewage is used to irrigate cropland, both positive and negative virus isolations have been made (Goyal et al., 1982; Koerner and Haws, 1979). At the Lubbock, TX, site, coxsackievirus

B3 was isolated from a depth of 30.5 m; at Roswell, NM, no virus isolates were detected in samples taken from 3-30 m depths. In the latter case, irrigation is seasonal and intermittent, whereas application at the Lubbock site is continuous.

At an operational land application site in Kerrville, TX (Moore et al., 1981) no viruses were detected in the monitoring wells at depths of 10.7-19.8 m even though viruses could be detected in 1.4 m-deep lysimeteters.

In one often-cited report (England et al., 1965) on the Santee project, no viruses were detected in renovated water. This is not surprising, since the detection methods available at that time were not quantitative. These negative results must therefore be considered highly questionable, as should the results obtained at the Whittier Narrows, CA (McMichael and McKee, 1966) projects, which did not employ techniques sensitive enough to detect low levels of virus. This reiterates the need for careful evaluation of methods used in any report before negative conclusions are accepted.

In summary, human pathogenic viruses have been isolated from beneath municipal wastewater land application sites practicing both rapid and slow rate application and this should be taken into consideration when locating such sites.

Studies with sludge amended soil indicate that viruses are not easily eluted even after rainfall events, and pose little threat to groundwater contamination (Damgaard-Larsen et al., 1977; Farrah et al., 1981). The effectiveness of virus retention on soils during sludge application is probably due to the capacity of the sludge solids to bind viruses in the top of the soil profile (Bitton et al., 1978). Although Bitton et al. (1978) pointed out that liquid sludges may contain unbound virus which could penetrate further into the soil during application.

3. Parasites: Protozoan cysts are highly sensitive to drying and are expected to only survive a few days in most soils. Entamoeba histolytica has been reported to survive 18-24 hours in dry soil and 42-72 hours in moist soil (Kowal, 1982). Helminth eggs and larve, in contrast to protozoan cysts, live for long periods of time when applied to land, probably because soil is the transmission medium for which they have evolved toward water transmission (Kowal, 1982). Thus, under favorable conditions of moisture, temperature, and sunlight, Ascaris, Trichuris, and Toxocara can remain viable and infective for several years (Little, 1980). Hookworms can survive up to 6 months (Feachem et al., 1980), and Taenia a few days to seven months (Babayeva, 1966); other helminths survive for shorter periods.

Because of their large size protozoan and helminth post no threat as groundwater contaminants.

B. Crops

1. Bacteria: The survival of bacteria on plants, particularly crops, is especially important since these may be eaten raw by animals or humans, may contaminate hands of workers touching them, or may contaminate equipment contacting them. Such ingestion or contact would probably not result in an infective dose of a bacterial pathogen, but if contaminated crops are brought into the kitchen in an unprocessed state they could result in the regrowth of pathogenic bacteria, e.g., Salmonella, in a food material affording suitable moisture, nutrients, and temperature (Bryan, 1977).

Pathogens do not penetrate into vegetables or fruits unless their skin is broken (Bryan, 1977; Rudolfs et al., 1950), and many of the same factors affect bacterial survival on plants as those in soil particularly sunlight and desiccation. The survival times of bacteria on subsurface crops, e.g., potatoes and beets, would be similar to those in soil.

The survival of enteric bacteria on crops has been extensively studied and reviewed (Bryan, 1977; Kowal, 1982). Reported survival times for common bacterial pathogens range from less than one day to six weeks (Kowal, 1982).

On the basis of field experiments with lettuce and radish irrigated with municipal wastewater, Larkin et al. (1978) concluded that leafy vegetables cannot be considered safe from Salmonella contamination until the soil can be shown to be free of Salmonella. They also noted that, because of regrowth in soil and on leaf crops, total coliforms and fecal streptococci bore no relationship to Salmonella levels, and are unacceptable indicators of fecal contamination; they recommended using fecal coliforms or Salmonella itself.

Thus, the consumption of subsurface and low-growing food crops, e.g., leafy vegetables and strawberries, harvested from an irrigated site within about six months of last application, is likely to increase the risk of disease transmission, because of contamination with soil and bacterial survival in cracks, leaf folds, leaf axils, etc. Possible approaches to avoid this problem are (1) use of the subsurface or covered drip irrigation method for aerial crops, (2) growth of crops the harvested portion of which does not contact the soil, e.g., grains and orchard crops, or (3) growth of crops used for animal feed only, e.g., corn (maize), soybeans, or alfalfa. The last alternative is probably the most common and most economic. In the situation where the harvested portion does not contact the soil nor is within splash distance, stopping wastewater

application a month prior to harvest would be prudent.

2. Viruses: On the surface of aerial crops virus survival would be expected to be shorter than in soil because of the exposure to deleterious environmental effects, especially sunlight, high temperature, drying, and washing off by rainfall (Kowal, 1982). Reported survival time of viruses on crops is similar to those of bacteria, and likewise appears to support a one-month waiting period after last wastewater application before harvest.

The absorption of enteric viruses by plants is a theoretical possibility. Murphy and Syverton (1958) found enterovirus to be adsorbed by tomato plant roots grown in hydroponic culture under some conditions, and in some cases to be translocated to the aerial parts. However, the rapid adsorption of virus by soil particles under natural conditions may make them unavailable for plant absorption, thereby indicating that plants or plant fruits would be unlikely reservoirs or carriers of viral pathogens. Using coliphage f2 Ward and Mahler (1982) found little virus was translocated to distal plants of corn and bean plants unless the roots were cut. The intact surfaces of vegetables are probably impenetrable for viruses.

Because of the possible contamination of subsurface and low-growing crops with soil, in which viruses have a longer survival time, about one hundred days would probably be a safe waiting period.

3. Parasites: Because of their exposure to the air, desiccation, and sunlight, protozoan cysts and helminth eggs deposited on plant surfaces would also be expected to die off rapidly. The fact that cysts can survive long enough to get into the human food supply under poor management conditions is confirmed by the recent isolation of high levels of *Entamoeba histolytica*, *E. coli*, *Endolimas nana*, and *Giardia lamblia* on the wastewater-irrigated fruits and vegetables in Mexico City's marketplaces (Kowal, 1982). Rudolfs et al. (1950) found contaminated tomatoes and lettuce to be free from viable *Entamoeba* cysts within 3 days. They concluded that field-grown crops "...consumed raw and subject to contamination with cysts of *E. histolytica* are considered safe in the temperate zone one week after contamination has stopped and after two weeks in wetter tropical regions".

Because of desiccation and exposure to sunlight, helminth eggs deposited on plant surfaces die off more rapidly. Thus, Rudolfs et al. (1950) found *Ascaris* eggs, the longest-lived helminth egg, sprayed on tomatoes and lettuce, to be completely degenerated after 27-35 days.

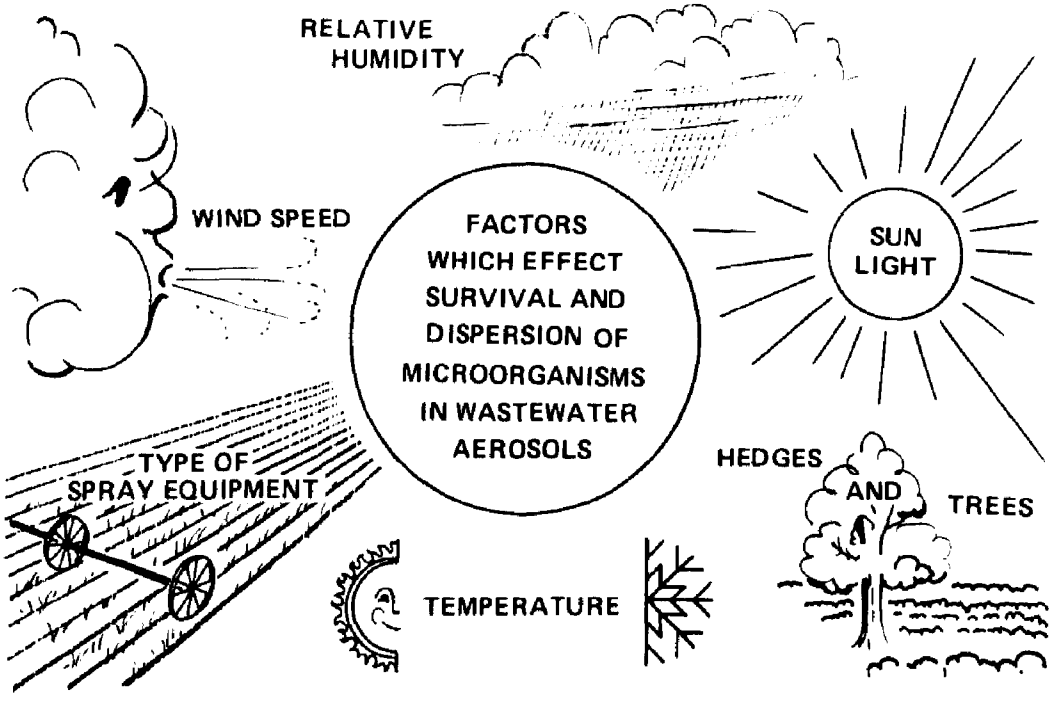
C. Aerosols

1. Bacteria: Microbial aerosols are readily created when wastewater is applied to the land by spray equipment. Their content of microorganisms depends upon the concentration in the waste water and the aerosolization efficiency of the spray process, a function of nozzle size, pressure, angle of spray trajectory, angle of spray entry to the wind, impact devices, etc. (Schaub et al., 1978). Aerosols represent a means by which pathogens contaminate fomites such as clothing and equipment but also a means of direct human infection through the respiratior route, i.e., by inhalation.

When aerosols are generated, bacteria are subject to an immediate "aerosol shock" or "impact factor" which may reduce their level by one log within seconds (Schaub et al., 1978). Their survival is subsequently determined primarily by relative humidity and solar radiation (Teltsch and Katzenelson, 1978). At low relative humidities rapid desiccation occurs, resulting in rapid die-off (Sorber and Guter, 1975), although concentration of protective materials within the droplet may occur (Schaub et al., 1978). Teltsch and Katzenelson (1978) have found bacterial survival at night up to ten times that during day time in Israel. High temperature is another factor decreasing bacterial survival. While biological aerosol decay is occurring, the rate of physical aersol decay, or deposition, simultaneously affects the distance of dissemination of the bacteria. This is influenced by wind speed, air turbulence, and local topography, e.g., a windbreak of trees. Factors which effect survival and dispersion of microorganisms in wastewater aerosols are shown in Figure 1.

Because of low density of aerosolized bacteria normally emanating from land treatment sites, high-volume samplers, e.g., 1 m^3/min electrostatic precipitators, are often necessary for aersol analysis. Likewise, because of the normally low denisty of pathogenic bacteria compared with nonpathogens, most measurements of aerosol bacteria have utilized traditional indicator bacteria, e.g., standard plate count, total coliforms, and fecal coliforms (Kowal, 1982). The measurements of Johnson et al. (1978) have shown little correlation between densities of these indicator bacteria and densities of the pathogens which they are intended to indicate. This results in "extreme underestimation of pathogen levels," since the pathogens which they studied, i.e., Pseudomonas, Streptococcus, and Clostridium perfringens, survived the aerosolization process much better than did the indicator bacteria. They suggest that fecal streptococci might be a more appropriate indicator organism because of its similar hardiness upon impact and viability to those of pathogens. Similarly, Teltsch et al. (1980) measured densities of coliforms, Salmonella, and the enteroviruses in aerosols and wastewater at an Israeli land treatment site, and from "...the ratios of salmonellae to coliforms and enteroviruses to coliforms in the air, as compared to these ratios in the wastewater, it was concluded that the suitability of coliforms as an indication of airborne contamination caused by spray irrigation is questionable."

Figure 1. Factors which effect survival and dispersion of microorganisms in wastewater aerosols.



The results of field studies suggest that the aerosol bacteria are usually detected at a maximum distance that is less than 400 m from the spray site. Bausum et al. (1978) found that at 30 m 75% of the aerosolized particles fell in the respirable range.

The human exposure to aerosol bacteria at land treatment sites can be roughly estimated from the data at Kibbutz Tzora, Israel where raw wastewater is sprayed (Kowal, 1982). An adult male, engaged in light work breathing at a rate of 1.2 m^3/hr and exposed to 34 coliforms/m^3 (the Kibbutz Tzora average) at 100 m downwind from a sprinkler, would inhale approximately 41 coliforms per hour. Since the ratio of aerosolized Salmonella to coliforms is 1:10^5(Grunnet and Tramsen, 1974) the rate of inhalation of Salmonella would be about 10^5-fold less, an extremely low rate of bacterial exposure. More recent data from Kibbutz Tzora allows a more accurate estimate of human exposure (Teltsch et al., 1980). During a period of time in 1977-78, when the wastewater total coliforms were 2.4 x 10^6 to 1.4 x 10^7/100 ml and Salmonella was 0-60/100 ml, the density of aerosol Salmonella at 40 m, the maximum distance found, was 0-0.054/m^3, with a mean of 0.014/m^3. This would result in an inhalation rate of 0.017/hr at 40 m, higher than the previous estimate, but still an extremely low rate of bacterial exposure.

2. Virus: Aerosols have been of concern as a potential route of transmission of disease caused by enteric viruses because, as with bacteria, once they are inhaled they may be carried from the respiratory tract by cilia into the oropharynx, and then swallowed into the gastrointestinal tract (Kowal, 1982).

The initial aerosol shock during the process of aerosolization may result in a half log loss of virus level (Sorber, 1976). The subsequent die-off, estimated to be about one log every 40 seconds (Sorber, 1976) is determined primarily by solar radiation, temperature, and relative humidity. Sorber (1976) has estimated that, under the least desirable meteorological conditions studied, less than 200 meters would be required to provide a reduction of three logs in aerosolized virus concentrations.

Very few measurements of aerosol viruses from the spraying of wastewater have been reported in the literature. The spraying in Israel of 3-5 day detention time oxidation pond effluent, having a coliform density of about 10^6/100 ml, resulted in the detection of poliovirus, coxsackievirus, and echovirus up to 100 m downwind (Shuval, 1978). To obtain quantitative measurements of aerosol virus concentrations in air may require heroic efforts. Johnson et al (1977) operated ten high-volume samplers (1 m^3/min electrostatic precipitators) for three-hour sample periods, at 50 m downwind from the source, to measure the aerosol enteroviruses produced by the spraying of unchlorinated aerated-effluent in Pleasanton, California. Likewise, Teltsch et al. (1980) used a large-volume scrubber-cyclone sampler to extract 27 ± 11 m^3 of air downwind from an irrigation line spraying raw wastewater at Kabbutz Tzora, Israel.

The results obtained from these two studies are highly variable, but it appears reasonable to make use of the Pleasanton aerosol virus density, i.e., $0.014/m^3$, to make human exposure estimates, since (1) the Pleasanton wastewater virus level is similar to that in U.S. wastewaters in general, (2) the high Israeli wastewater virus levels are not typical of those found in the U.S., and, in any case, a wastewater stabilization pond would decrease these levels, and (3) the $0.14/m^3$ value found at 50 m in Israel is based on only one sample, and does not appear to be representative of the other values (Kowal, 1982).

From these data it can be calculated that an adult male, engaged in light work, breathing at a rate of 1.2 m^3/hour and exposed to 0.014 PFU/m^3 at 50 m downwind from a sprayer, would inhale approximately 0.13 PFU of enterovirus during an 8-hour work day. This is probably an insignificant level of exposure. However, since the recovery of enteric viruses from environmental samples is not perfectly efficient, isolation of viruses cannot yet be isolated on cell cultures, the actual exposure to enteric viruses may be as much as ten to a hundred times the reported level (Teltsch et al.,1980). Thus it might be prudent to recommend a 100 m or 200 m minimum exposure distance of the general public to a land treatment spray source (Kowal, 1982).

3. Parasites: Because of the large size of protozoan cysts and helminth eggs compared with bacteria and viruses, it is extremely unlikely that they will find their way into aerosols.

CONCLUSIONS

Kowal (1982) recently reviewed in detail microbial aspects of the land treatment of wastewater. His conclusions and recommendations are worth repeating here.

He concluded that "although untreated wastewater should never be used for irrigation, the level of preapplication treatment required for the protection of public health may be as little as properly-designed sedimentation at land treatment sites with limited public access, where crops are protected by appropriate hydrolgical studies and selection of application rate....Because of potential contamination of crops and infection of animals, slow-rate irrigation and overland-flow systems should have complete removal of helminth eggs."

"In situations with greater public access (e.g., water disposal on golf courses), shorter waiting periods before grazing or harvest of crops ...or threat of groundwater contamination (e.g., shallow water table), more extensive preapplication treatment may be required. This treatment may consist of wastewater stabilization ponds, conventional treatment unit processes, or even disinfection. The exact degree of pretreatment

required for these situation is site-specific, and recommendations should be determined separately for each system."

Because of potential exposure to aerosolized viruses and bacteria at land treatment sites where spray irrigation is practiced, it would be prudent to limit public access to 100-200 m from the spray source. Kowal recommended "suppression of aerosol formation by the use of downward directed, low-pressure nozzles, ridge-and-furrow irrigation, or drip irrigation is recommended where these application techniques are feasible ....Aerial crops with little chance for contact with soil should not be harvested for human consumption for at least one month after the last wastewater application; subsurface and low-growing crops for human consumption should not be grown at a land treatment site for at least six months after last application. These waiting periods need not apply to the growth of crops for animal feed, however."

Kowal (1982) concluded that "properly designed slow-rate land treatment systems pose little threat of bacterial and viral contamination of groundwater." Recent findings (Table 11) of viruses at considerable depths indicate that further studies are needed before this conclusion can be validated.

The threat of bacterial and viral contamination at rapid-infiltration sites does exist, especially if high groundwater tables are present, and appropriate preapplication treatment or management techniques should be instituted, e.g., intermittent application of wasterwater," but little threat to groundwater contamination by microorganisms appears to exist from sludge disposal.

RECOMMENDED RESEARCH

1. Development of models based upon virus and site characteristics for predicting survival and transport of viruses in the soil that will take into account as much phenomena as is presently known including long-range movement as evidenced by the findings of viruses at considerable depths and distances from wastewater land-application sites.

2. Studies of virus survival and movement in the soil designed to test the validity of the above models.

3. Development of improved virus detection methodology since only one-tenth to one-hundredth of the viruses in wastewater can at present be detected.

4. Studies of the survival of viruses and protozoan cysts in storage- and waste-stabilization ponds.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

Gerald Berg, University of Cincinnati: Viruses are now being recovered from deep wells under land application sites. Would you comment on the efficiency of virus recovery techniques that have accomplished these recoveries and upon the effectiveness of these techniques in recovering hepatitis A virus, rotoviruses and Norwalk-type viruses from wastewater effluents, groundwater, sludges and soils?

Charles P. Gerba: I think that's a good point, Jerry. One of the things to remember when you show these data, the methods we have for assessing viruses in water and various environmental samples are probably no more than 30 to 50% efficient based largely on data obtained for the enteroviruses. Methods are not currently available for the detection of Norwalk agent and hepatitis A in water. These agents are a major cause of waterborne viral illness. We also have no idea of how these viruses behave in the subsurface to contaminate groundwater. We have a lot of unanswered questions about these viruses.

Lee Sommers, Purdue University: Please address the fate and movement of viruses and other pathogens following land application of municipal sewage sludges.

Charles P. Gerba: I'm sorry I moved so fast I neglected to mention that topic in my presentation. We covered that area in detail in the manuscript. It would appear, based on the current evidence, the impact on groundwater would be marginal based on the limited number of studies that have been done. Viruses, at least, appear to be very effectively immobilized on sludge solids. Bacteria and parasites being larger also have little impact on the groundwater. Both laboratory studies and field studies indicate that penetration of microorganisms in marginal land application sites. In terms of survival time, at least on the data I've seen, indicate that it depends a lot on where you are disposing the sludge. Temperature and climate may play a rather significant role -- at least in the survival of the organism at the soil surface. Sufficient waiting would be necessary to ensure inactivation of pathogen microorganisms. Certainly, I wouldn't recommend disposing raw sludge on crops.

Merilyn B. Reeves, League of Women Voters: Yesterday I asked Mr. Pound how much pretreatment is necessary for land treatment, particularly in regard to the need for nutrient removal. In the paper he presented it was noted "In general, the removal eficiency of overland flow systems for pathogenic organisms is comparable to that of conventional secondary treatment systems". However, on page 20 of the paper you just presented [Norman] Kowal is quoted as stating "although untreated wastewater should never be used for irrigation, the level of preapplication treatment

required for the protection of public health may be as little as properly designed sedimentation".... "In situations with greater public access ...or threat of groundwater contamination... more extensive preapplication treatment may be required".

Since the level of pretreatment affects the cost and thus the acceptability of a land treatment facility, how much pretreatment do you recommend for control of virus, pathogens, and other micoorganisms? Do you believe land treatment should be considered as 'advanced' treatment, an add-on for conventional treatment processes?

Charles P. Gerba: I think it is a very site specific problem. I think I would have to agree with Kowal that the amount of treatment would depend upon the actual intended reuse. It seems to me that you should take advantage-- I think one of the problems in terms of pathogens--people really (when they begin an evaluation of the problems of pathogens) they look at land treatment as a disposal method rather than a treatment method. Based on studies that I have reviewed, I think in some situations all that is needed is primary sedimentation before land disposal because you can get very high removal of the viruses from primary wastewater during land disposal of wastewaster. I think any time that a site is designed, we should try to take advantage of the site characteristics and further reduce the pathogen load on the environment. Removal during the infiltration of wastewater, at least in terms of viruses, can be very effective and more effective than chlorine under the proper conditions. I believe it is very site specific. We need to establish on a more formal basis a method for assessing the extent of pathogen removal at each site under consideration. In many cases we may find that only primary treatment is needed. I don't think we should chlorinate unless we really have to. We should take advantage of the removal that we can achieve with land application. Now, if you are going to use sewage for crop irrigating, the pretreatment requirements become much greater because there is a greater chance of potential health risks. So, there, we have to make an adjustment again based on site specific characteristics. Specific to overland flow, at least in my literature review, I couldn't find any data--the amount of data is so limited on removal of pathogenic microorganisms--that I don't think an assessment can be made at the current time of the efficiency of pathogen removal by overland flow--at least from the published literature. I hope I answered all your questions.

Cecil Lue-Hing, Research and Development, Metro Sanitary District of Greater Chicago: Relative to the degree of treatment necessary or desirable for land application of sewage sludge, I would like to share my knowledged of the Paris, France sewage farm with the lady from the League of Women Voters. First the lady has asked the bottom-line question. How much treatment is necessary? Is this option getting too expensive? The Paris sewage treatment facility is perhaps the second largest facility of its king in the world - second only to Chicago's West-Southwest facility.

Since about 1897, the sewage from the Paris facility has been used to fertilize and irrigate vegetables such as cabbage, leek, celery, and beans for sale in Paris. The only treatment given the sewage is primary treatment (settling/sedimentation). The procedure used on the land is flood irrigation.

Since 1974, the Paris authorities discovered that the nitrogen content of the sewage was too low, so they added approximately 203% anaerobically digested sludge to increase the nitrogen content.

With regard to public health, I am not aware of any adverse difference in the health of the people of Paris (attributable to their consumption of sewage-grown vegetables) compared to the health of the people of Chicago. More recently, Dr. Coin of The Ministry of Health of Paris has stated that after many years of search he has been unable to detect any microorganisms on the sewage-grown vegetables attributable to sewage irrigation.

One final comment (in my role as researcher and contributor to government policy) regarding research needs. In this decade, in this century, we can always develop an impressive list of research needs; similarly, fifty decades from now and in the next century. However, if as researchers you cause the desire for additional information to incapacitate the making of public policy, along with land treatment, you too will price yourselves out of the market.

Charles P. Gerba: I find additional comments like yours not generally useful to me in making an assessment. Your comments are based on personal observations and not controlled scientific studies, which is necessary for an adequate assessment. I think we can list a long series of personal experiences but we need to back those up by scientific evidence. We have to be shown in carefully controlled studies that there really are no differences. Such studies are very costly and will rarely provide an absolute measure of risk. What we need to do is establish a reasonable basis for making these assessments which can be modified in the future. That's a challenge. There is always going to be a need for more research. What we need to do as researchers is to provide the basis on which we can make rational decisions at a reasonable cost.

Betty H. Olson, University of California, Irvine: Can the relative sorptive capacity of soil be overwhelmed with continuous application of wastewater or sludge on land, and therefore, allow more penetration?

Charles P. Gerba: Dr. [Clarence] Lance and I conducted studies to determine if the ability of soils during continuous flooding with sewage would be overwhelmed and virus penetration increased (*Appl. Environ. Microbiol. 32:520-526, 1976*). In these experiments the capacity of the soil to adsorb virus did not change. We though that this occurred

because of inactivation of adsorbed virus allowing for the continual replenishment of adsorption on the soil.

Betty H. Olson: Cannot chlorination just be applied to groundwater and eliminate viruses?

Charles P. Gerba: The proper concentrations of chlorine can very effectively inactivate any viruses present in groundwater. Although, it should be recognized that viruses are more resistant to chlorination than enteric bacterial pathogens.

Betty Olson: Bacterial detection techniques have recently improved. Can one be sure that bacteria are not present or that they are just not being detected?

Charles P. Gerba: It is possible that not all the enteric bacteria present in groundwater are not detected. Being stressed, they may not be detected on selective media commonly used for their isolation.

Betty Olson: Certainly volume and dilution can affect numbers of bacteria detectable by traditional testing methods. How are volumes tested for both viruses and bacteria in groundwater movement? What is the rationale for determining volumes for bacterial sampling?

Charles P. Gerba: I do not know the rationale for determining volumes for bacterial sampling, which are usually reported as numbers per 100 ml volume. For viruses volume sized ranging from 10-1000 liters have been suggested. The large volume used for virus detection is based on their low infective dose.

Harvey Luce, University of Connecticut: Is the movement of viruses confined to soils with large pore systems (channels, chambers, joint places, coarse textured soils, etc.) or does movement also occur in soils with small pore systems?

Charles P. Gerba: I think the greatest movement of all bacteria and viruses occurs in sandy soils. There have been a number of epidemiological studies not associated with land treatment of sewage. The greatest incidence of diarrheal disease, for example, always occur during the periods of heavy rainfall. Thus, movement of microorganisms through the subsurface is greatly influenced by rainfall events. This movement occurs as a slug or concentrated band of viruses because of the elution

of previously adsorbed virus. We actually know less about the bacteria in those slug-type movements than we do about virus. I think that the area is an area that needs further investigation.

Virus removal by soil type varies greatly. Most of the studies have been with sandy soils because it has been seen as being the greater problem although some agricultural soils have been evaluated. Movement of viruses is less in agricultural type soils than sandy soils.

Another thing that should be looked at is the findings of Mark Sobsey that shows the groundwater pH changes directly affected the movement of viruses in subsurfaces. I am not sure if those are related to rainfall tests or not. Maybe Mark could answer that question. Generally the higher the pH, the greater the chance of isolating the virus in subsurface is going to be.

Mark Sobsey, University of North Carolina: The comment I would like to make actually follows on the comments that were just made. Also related to some of the other comments that have been made already and that is, it is probably unreasonable to expect land disposal and land treatment site for wastewater sludge to either be the recipients of materials that have already almost completely or completely disinfected or in some other way made to void a pathogenic microorganism and it's probably also unreasonable to expect such sites to be totally effective in removing pathogenic microorganisms. So, we have to bear in mind that it is likely that some relatively low level of movement of pathogens may occur at a land disposal site. And, therefore, it behooves us to consider the possible beneficial uses of that site and the other reasons for which it is being developed. We can't think of it strictly in terms of the impact of pathogenic microorganisms. Clearly what we would like to do is keep the risk of pathogen context to the public to an acceptable level and therefore we have to consider whether or not the water underlying that site is going to be withdrawn for some beneficial purpose such as water supply subsequently, and if everybody considers that possibility as water marginal levels of water treatment that is going to be utilized, are we going to be growing crops on such sites that are going to be consumed by people and animals and therefore, we have to tailor our desires, I suppose, for pathogen reduction in these sites to other aspects. We must have a management strategy that considers all of the possible uses and intentions for such sites. The other thing that kind of concerns me is that we can demonstrate and have demonstrated lower levels of pathogens in groundwaters we can demonstrate the presence of pathogens and sludges applied to application sites or in the wastewater that is applied. But we don't have standards for such pathogens and we generally rely on coliform bacteria or some other indicators of people pollution to be the standards for drinking water or for other source of materials that might be consumed by humans and we have very little information on the relationships between whether the pathogen is an indicator bacteria or other indicators nor are we fully aware of what those relationships even ought to be. This again brings us back to the lack of information on the

epidemiology of these viruses and what contribution to the circulation of the viruses and other pathogens in the population from something like the land disposal sites we are willing to accept.

B. L. Carlile, Texas A & M University: One factor not discussed in your position paper is the relationship between loading rates (hydraulic and treatment efficiencies of land application systems. Slow rate land treatment systems for agricultural utilization appears to be quite different from high rate disposal systems.

Charles P. Gerba: Laboratory studies with soil columns showed that hydraulic loading rates are directly related to virus removal efficiencies. No comparative fields have been conducted to determine if removal efficiencies of rapid infiltration and slow rate sites are different. Viruses have been detected beneath both types of land treatment sites.

William Pounds, Pennsylvania Department of Environmental Resources: Is there a reasonable, reliable test that can be conducted on a sludge to determine the degree of treatment or pathogen removal, on a day-to-day basis, before sludge is applied to the land?

Charles P. Gerba: Dr. [Wylie] Burge has suggested the use of f2 bacteriophage as an indicator of pathogens in composted sludge (*J. Water Pollut. Control Fed. 53:1683-1690, 1981*).

Ebba Lunda, Royal Veterinary and Agricultural University, Copenhagen, Denmark: The percentage removals in various treatments as presented in Table 3 seems difficult to interpret. The important questions is not how much you remove in percentage, but how much remains and if this residual is a public health hazard. The load influences the percentage removal and various systems function differently under different climatic and management conditions. Without qualifications such a table is misleading and should be deleted. That primary sedimentation removes 0 percent of viruses cannot be correct when the concentration of viruses in primary sludges is higher than in raw sewage, 10-100 fold higher in my experience. Stabilization ponds can be very efficient, but to give the general impression that they are better than activated sludge treatment seems misleading.

George J. Jackson, Food and Drug Administration: Many methods for the recovery of microorganisms from sewage sludge, soil or food are not highly efficient. Results should be interpreted keeping "false negatives" in mind.

Land application of wastewater and sludge may not only add pathogens to the soil but, possibly, serve to select pathogenic strains of opportunistic parasites ("free living" protozoa like Naegleria already present in the soil.

Flora Mae Wellings, Epidemiology Research Center, State of Florida: In all of the presentations today, one is led to believe that pathogens are of no concern. This bothers me but I am even more concerned over the last speaker's statement that landspreading results in complete cleansing. This may be true for some of the constituents in sewage and sludge but it is not true of viruses. I can assure you that even secondary effluents applied to soils composed mainly of sand results in viruses percolating into the groundwater. This has been shown over and over again. This is a real concern for Public Health and it must be addressed. I can assure you that federal regulations permitting this type of landspreading of primary or secondary effluents will not be acceptable to Florida.

Y. A. Demirjian, Muskegon County, Michigan: Who sampled and tested for viruses at the Muskegon County wastewater facility for viruses?

Charles P. Gerba: Our laboratory conducted the study at Muskegon to which I have referred, too. This information will appear in an article in the Journal of Water Research within the next year.

Scott Clark, University of Cincinnati Medical Center: My comment is in response to Dr. Y. A. Demirjian's question on the apparent discrepancy between Dr. Gerba's detection of virus at a depth of 10 m below the surface at the Muskegon land application site and our failure to detect animal viruses in either air samples downwind of the pretreatment aeration lagoons or in wastewater at the pump station just prior to irrigation. Detecting viruses in air samples is notoriously difficult. The lack of detectable viruses in the wastewater prior to irrigation was most likely due to differences in sample preparation methodology but also may been due to differences in time of sample collection.

Gerald Stern, EPA, Cincinnati: In reference to using percent of log reduction for pathogen inactivation, I wish to share some experiences with you. We collected data on the pathogenic bacteria and viruses reductions for about 3 years on mesophilic and thermophilic anaerobic digestion from one plant. In trying to evaluate the data on a pathogen density level, we found a large amount of variations. It wasn't until we paired the data that consistent relationships of 2 log reduction of indicators could be related to 1 log reduction of salmonella and viruses. Trying to evaluate data of this type is difficult.

Barbara E. Moore, University of Texas: Directed to the previous question from the floor: How do you explain not detecting virus in irrigation water, but finding them in subsurface water?

Viruses may be present in water at low concentrations. As pointed out by Dr. Gerba, they are retained (adsorbed) within the soil profile. Such retention is not synonomous with inactivation and a "concentration" of viruses or bacteria may occur in the soil depending on numerous environmental factors affecting organism viability. An elution event, such as heavy rainfall can mobilize previously adsorbed viruses. If, as an intermittant observer, one happends to be monitoring on such an occasion, a large number of viruses could be recovered from subsurface water.

Another point which deserves consideration is that our ability to detect viruses in water is not 100 percent efficient. Viral recovery from other matrices found in land treatment systems such as sludge and soil is less efficient, however. One cannot assume that sampling a few soil samples with negative results indicates a lack of viruses within the soil profile.

Flora Mae Welling: I want to emphasize the statement made by Barbara Moore. The adsorption of viruses is not synonmous with virus inactivation. Viruses may adsorb and desorb over and over again without the virus losing its infectivity. Thus, even though small numbers of virions are applied they become concentrated in the upper few inches of the soil by the adsorption process. When the pH is altered, perhaps by heavy rainfall, then viruses are desorbed en mass and are forced to move downward with the receding waters. If the groundwater is near the surface, the viruses would quickly be entrained in the groundwater. If the groundwater is at a very deep level, repeated adsorption and desorption cycles may occur before the viruses reach the groundwater. Once entrained in groundwaters we do not know how far they might travel. Secondly, I think Bob Kadlac from Michigan did some overland flow studies and I believe viruses were demonstrated in the lake into which the renovated water drained.

As for viruses in sludge, my laboratory demonstrated some 24 PFU of virus in a few grams of "dried" sludge from a package treatment plant. The sludge had been on the drying bed for 14 days under the hot Florida sun.

And last, at a sludge disposal site no virus was observed over a seventeen month period but on the 18th month viruses were demonstrated at a depth of 28 feet. Investigation of the area showed that a burrow pit had been dug to the north-northwest of the site which apparently changed the flow of the groundwater. Months later viruses were demonstrated in water from the 58 foot deep companion well. This instance is an example of the difficulties involved in monitoring groundwater. We still have a long way to go before we can devise the ideal monitoring system for groundwater.

SPECIFIC ORGANIC COMPOUNDS

LAND TREATMENT OF MUNICIPAL EFFLUENT AND SLUDGE: SPECIFIC ORGANIC COMPOUNDS

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ABSTRACT

The pathways and rates by which specific organic compounds are assimilated in a municipal sludge or effluent land treatment system have a reasonable, but not complete, data base. Use of these data is essential to compare the generation of such chemicals to the soil capacity for treating such organics. It appears that the rate of certain specific organic priority pollutant applications to land is nearly the same with agricultural sludge versus effluent land applications. The mechanisms which are evaluated to assure satisfactory assimilation can include 1) volatilization, 2) photodecomposition, 3) decomposition (chemical and microbial), 4) adsorption, leaching and runoff, 5) plant response. Review of these pathways is presented as generic mechanisms followed by illustrative data, since the number of chemicals of concern (over 100) and the magnitude of treatment mechanisms are very large. Summarizing these discussions, it was concluded that 1) decomposition is prevalent for most chemicals and at rates which appear to indicate that satisfactory assimilation is occurring; 2) crop uptake occurs at varying levels, but such data remain imprecise without some health-related quantification; 3) for traditionally designed municipal sludge or slow rate effluent application systems, leaching does not appear significant as an environmental pathway for organics of land treatment.

INTRODUCTION

The historical perspective regarding the specific organic constituents in municipal effluents or sludges is somewhat varied. Using the Proceedings of the Joint Conference on Recycling Municipal Sludges and Effluents on Land (U.S. E.P.A., 1972) as a benchmark of the municipal land treatment field, one would conclude that there had

been minimal attention on these waste constituents. However, one must review the situation in 1972 for an interpretation of this lack of attention.

In 1972 there was no focused list of organic compounds that had been proposed to be of environmental concern. Thus only certain organics were identified as possible concerns, namely the Aroclor mixtures referred to broadly as polychlorinated biphenyls (PCB). These were mentioned in the chapters on municipal effluent and sludge characterization and in the chapter on monitoring (Blakeslee, 1972). Actual quantification of Aroclor mixture concentrations were given for a number of municipal effluents and sludges in Michigan. Citations in this benchmark Proceedings represented the level of focused attention on specific organics in regard to municipal land application.

A different perspective of the concern and level of understanding for organic compounds is obtained by evaluating the broader field of the behavior of organic compounds in the terrestrial system (even at the 1972 period). This broader field encompasses research interests in pesticides and residues, other agricultural chemicals, soil sterilization, response of chemical spills, organic fertilizers, and fundamental investigations of soil-plant behavior. With this broader perspective, one concludes that there is a very large data base and a substantial delineation of mechanisms operative for specific organic compounds in soil-plant systems. This information began accumulating in early 1900 and the article by Buddin (1914) represents (even at that early date) a sound understanding of germicidal effects of organic chemicals on soils (some of which are now considered priority pollutants). Thus the research into the behavior of toxic and nontoxic organic compounds when applied to land has been continued as a significant research area today. This contrasts with the low level of attention given this area in relation to land treatment, circa 1972.

The magnitude and diversity of the organic compounds that have been evaluated in the context of a terrestrial system was estimated recently and put in catalogue form, called TERRETOX (Overcash and Sims, 1981). From an article-by-article search of 240 appropriate journals for the period 1976-1981, over 4,000 organic compounds were found (many non-pesticides). These citations contained one or more evaluations of the following classes of terrestrial behavior of specific organics: volatilization, photolysis, decomposition, adsorption, plant uptake, phytotoxicity. Thus there exists a substantial (and to many, surprisingly large) data base from which to begin evaluation of specific organic compounds as these might impact land application of municipal effluents and sludges.

In 1976 the definition of organic compounds of environmental concern changed from an amorphous collection containing a few highly visible chemicals to a specific list of organics - the Consent Decree Priority Pollutants. Prior to this date, attention on organics in municipal wastes was restricted by the lack of knowledge about what compounds were present or might be of concern. Thus attention or research priority was limited in scope. With the priority pollutant list, whether complete or not, a program was undertaken to measure these specific organics in municipal wastes, effluents and sludges (Feiler, 1980). These results are discussed in a later section. For the organic chemical class, the increased awareness of municipal effluent and sludge concentrations of organic constituents has stimulated recognition of the larger body of research information pertinent to understanding this facet of land application. Thus the merging of information from the terrestrial behavior of organic compounds to land application of municipal sludge and effluents is beginning. Given the extent of the former, it appears that there will be significant transferrable information to allow estimation of behavior, relative assimilation capacities, and critical levels for specific organics. However, specific research on the organic priority pollutants in the presence of the municipal sludge or effluent will also be necessary as the basic understanding of the individual chemicals is obtained (experimentally or via the literature).

In the inorganic field, such as heavy metals, it is possible to discuss most of the species and pathways in the context of a review article. The organic constituents, even if restricted to the priority pollutants (probably not a suffucient list) represent about 100 compounds as compared to 5-10 metal elements of concern. In addition, these organic compounds vary widely in the pathways that are significant to describe chemical fate and actually include more mechanistic routes for assimilation than metal elements. This magnitude and complexity of organic chemical information can be presented as a complete description of the terrestrial behavior of a single group of closely related compounds. Such reviews are lengthy and examples are available (Overcash et al, 1982; Pal et al, 1980; Overcash, 1981a; Sims and Overcash, 1983). Since this area of municipal waste concern is emerging, a broad overview is not possible in which all information and details are provided. This posed a dilemma for a state-of-the-art review of organic compounds, and so an alternate approach became necessary. This approach is to describe, in a generic fashion, the assimilative pathways for organic compounds in terrestrial systems. Illustrations of information are given. Then the author provides a general estimate about the significance of pathways relative to municipal wastes. The reader will thus have to pursue in some detail specific compounds by use of published literature. This is typical of the scattered yet substantial information on organic chemicals in the terrestrial systems.

MUNICIPAL WASTE CHARACTERISTICS

There has only recently been large scale sampling and analysis for specific organic compounds in municipal raw waste, effluents, and sludges. Concern with trihalomethanes, with the priority pollutants, and with public health effects have encouraged such analyses. In general the impact (in terms of mass and type of chemical) of industrial discharges has not been separated from the typical domestic or small town characterization and so domestic characteristics remain an important information need. It remains difficult to predict the concentrations of specific organics in municipal wastes.

Several studies of municipal waste characterization for specific organics (not just priority pollutants) do exist primarily as basic data with little interpretation. A comprehensive study was made on the priority pollutants (U.S. E.P.A., 1982) and covered 60 separate plants (publicly owned treatment works). These data are without much interpretation except as percent removal although the basic data were published for the reader. Majeti (1981) published a review of literature information prior to the detailed EPA work and included non-priority pollutant organics. Recent information from Michigan (Jacobs and Zabik, 1983) further expands the characterization of the specific organics present in municipal sludge.

The concentrations of organic compounds in municipal sludges and effluents are at least as variable as those reported for metals (Sommers, 1977). From the perspective of municipal effluent or sludge land treatment this variation argues presently for direct analysis of each particular waste. The reduced cost (30%-50% of 1980 prices) has brought priority pollutant analyses within a more affordable range. Table 1 is derived from the lengthy characterization by U.S. E.P.A. in 1982.

The issue of PCB concentration in municipal wastes remains undefined. In the characterization by the U.S. E.P.A. (1982), the most prevalent chlorinated biphenyl mixture was 1242, with an occurrence frequency 1% for secondary effluent and less than 1% for primary sludges. PCB-1254 occurred less than 1% for both effluent and primary sludge. This contrasts with the Michigan data of Blakeslee (1972) in which PCB-1242 occurred 3% and PCB-1254 occurred 52% of the time in municipal effluent. Certainly year of sampling and other factors are involved. If the U.S. E.P.A. (1982) data describe the typical national occurrence, it may be that as an organic compound group, PCB is less important than a large number of other organic chemicals of concern.

Table 1. Summary of Municipal Secondary Effluent and Primary Sludge Composition of Organic Priority Pollutants

Chemical Group	Two Compounds with highest concentrations (ppb on as is basis) (average of 19 plants)		Range of concentrations reported (ppb on as is basis)
	SECONDARY EFFLUENT		
Halogenated Aliphatics			0.16 - 48
	Methylene Chloride	(48)	
	Methyl Chloride	(12)	
Substituted Aromatics			0.21 - 18
	Toluene	(18)	
	Pentachlorophenol	(2.1)	
Phthalate Acid Esters			0.21 - 21
	Bis (2-ethylhexyl) phthalate	(21)	
	Di n-butyl phthalate	(3.6)	
Polynuclear Aromatics			0.16 - 0.68
	Phenanthrene	(0.68)	
	Anthracene	(0.68)	
	PRIMARY SLUDGE		
Halogenated Aliphatics			37 - 3,600
	Vinyl chloride	(3,600)	
	Chloroethane	(1,500)	
Substituted Aromatics			29 - 1,900
	Toluene	(1,900)	
	Phenol	(320)	
Phthalate Acid Esters			280 - 3,900
	Bis (2-ethylhexyl) phthalate	(3,900)	
	Butyl benzyl phthalate	(3,900)	
Polynuclear Aromatics			89 - 410
	Phenanthrene	(410)	
	Anthracene	(410)	

In the context of land treatment for municipal effluent and primary sludge, the impact of organic priority pollutants are illustrated in Table 2. For the highest organic compounds in each waste type, the typical mass loadings are similar (1.4 kg/ha/yr for bis (2 ethyhexyl) phthalate as sludge versus 0.63 kg/ha/yr for methylene chloride as secondary effluent). These two compounds have very different behavior pathways in terrestrial systems, illustrating the difficulty in generalized statements. This similarity of organic compound loading rates, with substantially different concentrations reflects the complexity of other controlling parameters in sludge or effluent applications to land.

ORGANIC COMPOUND ASSIMILATIVE PATHWAYS IN MUNICIPAL LAND TREATMENT

A specific organic compound undergoes a variety of chemical and biological processes when applied to a soil or soil-vegetation system. Although a lot of research has not been conducted to verify preliminary observations, it is felt that the presence of the overall organic and inorganic fractions of a waste merely attenuates the various pathways by which an organic species is treated in the terrestrial system. Thus at this stage of development it remains important to understand and quantify the assimilative pathways (from whatever system was used to generate such data) and to project the probable acceptable rates of land application for each organic constituent of interest.

Volatilization

An organic compound irrigated in a wastewater, surface spread with a sludge, or injected into the soil will partition between the gas and liquid phases to exert a vapor pressure. This vapor may be rapidly lost as might occur with irrigation, may be re-adsorbed if present beneath the soil surface, or may exhibit any behavior in between these two extremes. The conditions of soil and application technique land application as well as the inherent organic compound volatility are important factors in quantifying how an organic compound might be lost through volatilization. It must also be remembered that volatile losses occur when municipal effluents are treated and discharged to streams and when sludges are put in landfills as well as when these wastes are managed by land treatment.

Volatilization literature exists for chemicals applied to soil systems. The following represents a partial summary for the information on certain aromatics (Overcash and Pal, 1979).

Table 2. Annual organic chemical loadings, kg of compound/ha/yr associated with traditional sludge and effluent application rates.

	For Primary Sludge at 500kg of N/ha/yr	For Secondary Effluent at 1 inch/week
Methylene Chloride	0.016	0.63
Bis (2 Ethylhexyl phthalate	1.4	0.28
Toluene	0.66	0.24
Phenol	0.11	0.02
Anthracene	0.14	0.009
Di-n-Butyl Phthalate	0.1	0.047
o-Dichlorobenzene	0.0035	0.01

"Many aromatics such as benzene, toluene, cyclohexane, benzoic acid and other substituted benzene derivatives are volatile. A considerable amount of these volatile aromatics can be lost from the soil-plant system by evaporation at ambient temperatures, depending on the wind speed and method of application in the soil. Chlorinated benzene derivatives are much less volatile and the vapor pressure of chemicals like hexachlorobenzene (1.2×10^{-4} mm Hg at $25^{o}C$) is so low that these could be categorized as nonvolatile (Smelt 1976). When applied in high doses, even the nonvolatiles are lost by volatilization, which is encouraged by high temperature, high wind velocities and surface application. In a laboratory experiment, about 80% of added quintozene was lost from a treated soil over a 10-month period when air flow was continuously directed over a 1-cm soil layer. Of this loss, 62% was accountable by volatilization. Increasing the organic matter level of a soil tends to decrease the volatilization of the hydrophobic nonpolar aromatics. Incorporation of wastes into soil is a management option available to further reduce volatilization losses."

The level of compound vapor pressure at which volatile losses are known to be significant (Weber, 1982) is usually taken as 5×10^{-6} mm Hg at $25^{o}C$. If one tries to identify vapor pressure characteristics as a measure of whether volatile losses should be evaluated, this threshold is probably a good first estimate. However, the adsorption to organic matter and soils could greatly reduce such initial concerns. For example, the substantially volatile toluene is the second most prevalent organic priority in municipal sludge even after opportunities for volatile loss during aerobic treatment.

Photodecomposition

A small percentage of a surface applied waste remains available to exposure for solar radiation. Under these conditions the organic compound can be degraded resulting in treatment of the parent organic chemical via photolytic mechanisms. Phenolics and polynuclear aromatics are two groups that readily undergo such reactions. Pathways for such losses have been delineated, but the significance of such losses in municipal land treatment has not been evaluated as a separate pathway. Usually photodecomposition is measured as a part of the total decomposition by biologic and abiotic routes.

Decomposition

This is a rather broad category of terrestrial pathways which basically reflect the conversion of specific organic compounds (the parent species) into metabolites, chemical intermediates, biological material, and small molecules (primarily CO_2). Microbiological as well as chemical reactions are usually acting simultaneously and, when measured as loss of parent compound, are usually undifferentiated. The decomposition pathway is probably the most important facet of the organic constituent behavior in municipal waste land treatment and differentiates this research most clearly from that of metals or other inorganics.

Chemical reactions (abiotic routes) are a part of the overall measure of compound decomposition. Two typical reactions are hydrolysis and neutralization of the parent organic species. While these have been measured, it is not typical that these be differentiated from the overall decomposition process, probably since these chemical reactions leave the bulk of the parent structure still intact. Thus the separate quantification of abiotic reaction rates is uncommon.

Microbial processes are the major focus in decomposition pathways for organic compounds. An initial consideration is whether a particular organic compcound reduces microbial number or affects enzymatic activity. The following description illustrates these effects.

> "Phenols and quinones inhibit enzyme activity in soils, especially urease (Douglas and Bremmer, 1971, and Haselhoff, 1932) and catalase (Dolgova, 1975). As the phenol concentration in polluted soils decreased from 19.6 to 10.3 ppm, the catalase activity increased from 0.5 to 6.6 ml oxygen consumed in a 5-minute exposure (Dolgova, 1975). In a control soil sample from a nearby botanical garden, the catalase activity was measured as 18.5 ml O_2 in a 5-minute exposure. The Chernozem sand and clay soils from the least polluted parts of an industrial area were fumigated with phenol in a hermetically sealed chamber using 160 mg/m^3 of phenol for 5 days. Phenol content of the soil was determined after the fumigation, whereas the catalase activity of the Chernozem, clay, and sand soils were measured before and after the fumigation. Catalase activity, expressed as ml O_2 after 5-minute exposure, for Chernozem, clay, and sand were 17.3, 9.5 and 0.3, respectively, prior to fumigation with phenol and declined to 13.3, 5.5 and 0.0 with fumigation, at soil levels of 104, 96 and 6.2 mg/kg phenol, respectively. Dolgova (1975) also observed a decrease

in activities of dehydrogenase, protease and urease enzymes in the same soils receiving 20-50 kg/ha of phenols as a result of coke industry contamination.

The effectiveness of phenols and their derivatives as germicidal agents is well known. The phenol index has been used historically by pharmacists as a means of comparison of germicidal quality (Coffman and Woodbridge, 1974). Despite the sterilizing influence, there are certain microorganisms that can withstand relatively high concentrations of phenolics.

Phenols exert an effect on soil microbial numbers that is dependent on the soil concentration or amount added. At low doses (0.01-0.1% of soil wt), the phenol serves as an available substrate and there is a dramatic increase in microbial values. This response is similar to that found with the addition to soil of other hydroxy compounds. As the dose level is increased (0.1-1.0% of soil wt), an increasingly strong inhibitory or sterilizing effect is noted. At these levels, a partial sterilization occurs in which there is a depression in microbial numbers but not a complete die-off. After a period of time, microbes adapt or phenol is lost through sorptive inactivation or volatilization and a regrowth of population occurs. The milieu of available dead biomass is such that mineralization and excess nitrogen for microbial growth occur. The nonvolatile quinones and hydroquinones exert a prolonged microbial effect at certain soil levels, similar to the response described for phenols."

The microbial die-off effects of organic chemicals are quantified by:

1) The soil concentration of an organic species at which significant inhibition occurs, and

2) The length of time to re-establish average levels of populations.

In general, the levels at which organic compounds exert adverse microbial effects are much greater than one would expect to find with municipal effluent or sludge land treatment. Exceptions certainly exist, but these have not been identified at this time. Even if application led to inhibitory microbial conditions, the net effect would only be a lag period after which decomposition or other losses would still be expected. For the case in which such a specific organic was the controlling waste loading factor, this lag would be a finite delay in successive waste applications rather than a constraint which leads to no allowable application.

Decomposition of specific organics is measured by four common methods:

1. CO_2 evolution
2. O_2 consumption
3. dynamics of intermediates or final products
4. loss of parent compound

Such experiments are not measuring the same phenomena in precisely the same manner and so results do not always agree. The intricacies of alternative techniques are not discussed here, but the reader must note that differences exist.

From the course of extensive literature searches concerning the decomposition of specific organics in the terrestrial environment, one concludes that very few organic compounds can be said to be non-degradable. This large decomposition potential of soil systems is not generally understood, but is significant in the national program to treat organic priority pollutants. Conventional aerobic treatment reviews list numerous organics as non-degradable. However, this categorization most often should be interpreted as applicable for the relatively short residence times found in conventional processes (6 hours to 30 days). Given very long time periods typical in soil systems, it appears that two classes of compounds could be regarded as nondegradable based on present terrestrial research information:

1) synthetic polymers manufactured for stability;

2) very insoluble large molecules, e.g., 5-10 chlorinated biphenyls.

In both of these cases, the soil microorganisms appear to have the potential to decompose the compounds (deamination, dehalogenation, etc.) but simply cannot achieve the molecular level conditions necessary for decomposition.

Decomposition rates (g/ha/d or ppm/day) are dependent on the concentration of the organic species applied to the soil. This is illustrated in Figure 1 for polynuclear and heterocyclic aromatics. Such relationships are first order and the measurement of the decomposition rate or half life then becomes the essential data for design. However, some compounds exhibit zero order decomposition rates (anthraquinone dyes, Tilchin 1983) under the conditions studied. Thus several reaction models and a large number of decomposition reaction rate constants can be used to assess loading criteria and behavior of specific organic compounds. The decomposition rate for organics have been measured for so many classes and specific compounds that a complete listing or rate quantification is difficult. Table 3

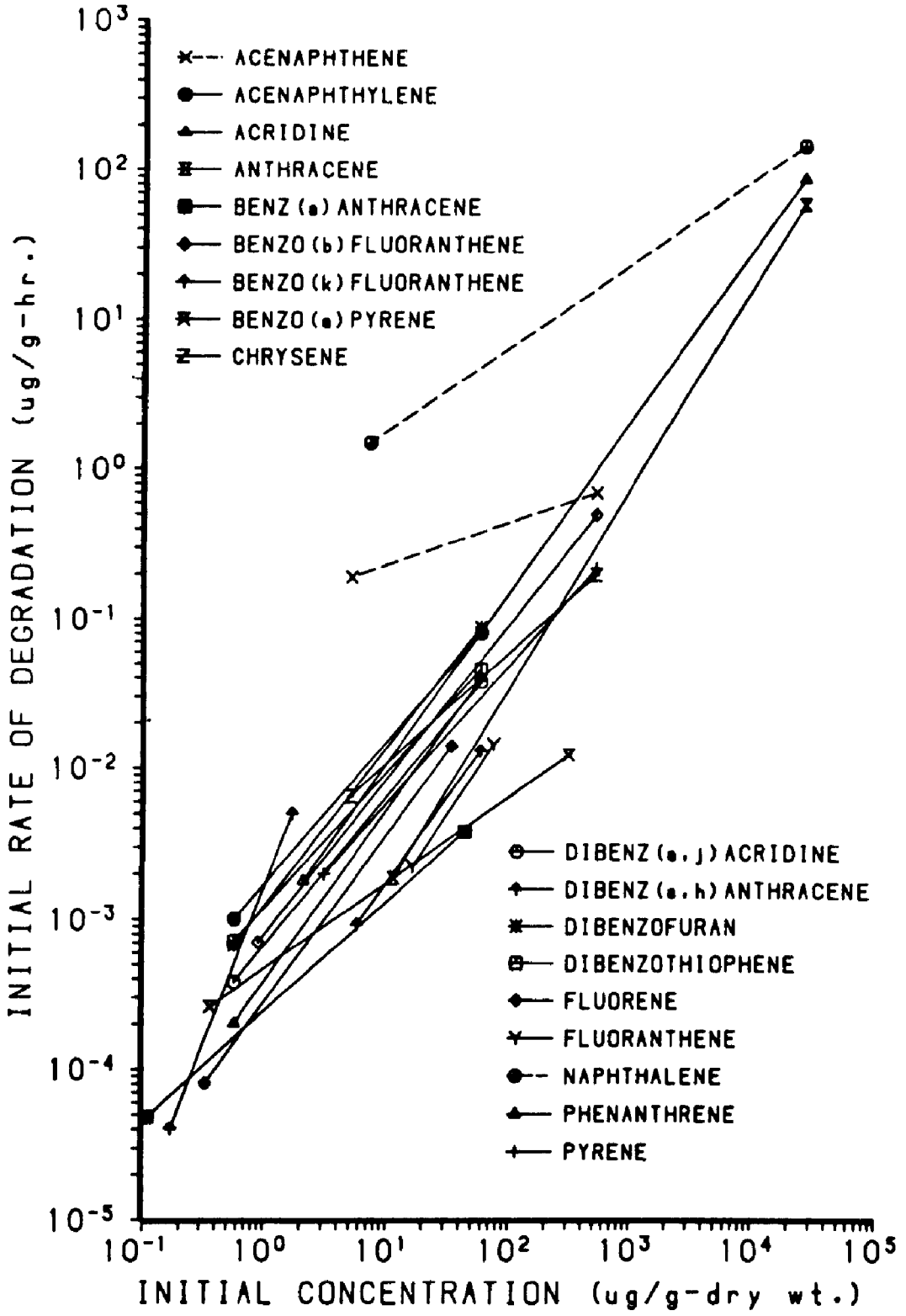

Figure 1. Decomposition of Polynuclear and Heterocyclic Aromatics (Sims and Overcash 1983)

Table 3. Illustrative Range of Decomposition Half Life for Organic Compounds

Compound	Approximate Half-Life
Aminoanthroquinone dyes	100-2,200d
Anthracene	110-180d
Benzo(a)pyrene	60-420d
Di n-butylphthalate ester	80-180d
Nonionic surfactants	300-600d
2,4 methyaniline	1.5d
n-Nitrosodiethylamine	40d
Phenol	1.3d
Pyrocatechin	12 hrs
Cellulose	35d
Acetic Acid	5-8d
Hydroquinone	12 hr

contains a range organic decomposition half life values to illustrate the assimilative capacity data for specific organics. At this time verification of decomposition rates of specific compounds in the presence of other municipal waste constituents is not widely available.

Adsorption, Leaching, and Runoff

Among the physical processes governing behavior of specific organics in a municipal effluent or sludge land treatment system the adsorption/leaching phenomena represent a balanced relationship. The attraction charge behavior between the organic species and either the mineral or the organic soil phases results in a partitioning into adsorbed and solution states. This partitioning may occur immediately upon application to the soil or may be delayed until separation from the waste medium occurs. Competing decomposition reactions can vary the actual amount of a given organic compound that resides on the soil/waste phase or in the soil-water solution.

The adsorption or partitioning processes occur within very short distances from the waste application location (molecular or soil pore size distances). In contrast, the occurrence of a soil-water matrix hydraulic potential (for example, from a rain event), migration of soil-water along appropriate gradients can cause transport of organics in the soil solution phase. This leaching can be viewed as theoretically capable of moving a dissolved organic (i.e., all organics) to any location receiving that soil water. That is, at least one molecule could be leached. As a practical matter, leaching of organics is insignificant if 1) the municipal effluent or sludge land treatment occurs at normal application rates, 2) a reasonable drainage and cyclic establishment of sustained aerobic soil conditions occur, and 3) groundwater remains deeper than 1-2 feet from the soil surface. That is, for most usual municipal land application situations, leaching or soil migration of specific organics does not appear to be significant or detectable. Rapid infiltration systems (with requisite sandy or gravel conditions) and overland flow systems are possible exceptions to this limited leaching behavior.

Organic compounds present at or near the soil surface (whether adsorbed or in the soil-water solution) can also be transported in rainfall runoff. The mechanisms and the algorithms to estimate the magnitude in runoff of organic compounds are available in existing non-point source models and research related to pesticides (Donigian, 1976; Frere, 1975; Overcash, 1983). Some modification to account for the organic sludge phase would be necessary to establish the runoff impact of organic priority pollutants. The reader should utilize this extensive non-point source information to assess the runoff pathway.

In a practical sense, the runoff of specific organics as a part of municipal waste land treatment would need to be evaluated to see if any critical levels could occur. Then the control measures would be to use the best management practices approach that has been established to manage non-point source pollution from all land uses. These practices are usually adopted anyway as a means of controlling erosion, as recommended agricultural practices, or in the course of permit approval for such land treatment systems.

The broad category of organic species behavior encompassing adsorption, leaching, and runoff, as a consideration in municipal waste land treatment design, is generally a secondary facet. The considerations of decomposition and plant response are generally more important. Observations of the various transport pathways under normal conditions of land treatment design have also not identified leaching or runoff as major considerations, or at least not unmanageable with sound design and operation.

Plant Response

The final pathway describing the terrestrial behavior of specific organic compound centers on the interaction with vegetation. This vegetation may be continuously present (e.g., forest, grasses) or may be planted and harvested in cycles (e.g., corn, soybeans). An organic compound impacts vegetation in two broad ways:

1) At low concentrations, the compound partitioned into the soil water solution is available for uptake by the vegetation root system;

2) At high concentrations, a phytotoxic response occurs.

The magnitude of these plant responses is dependent upon both the organic chemical and the vegetative species. As with decomposition rates, there are substantial direct and indirect information to evaluate the plant response to numerous organic compounds.

Phytotoxicity can occur when the foliar portion of vegetation is exposed to an organic compound or when the root system contacts an organic compound. The applied concentrations (in municipal effluent or sludge) at which phytotoxicity occurs is typically different for foliar versus root contact. In the latter pathway the competing adsorption and decomposition phenomena allow greater tolerance to elevated concentrations of organic compounds. As an example, Figure 2 compares the critical levels of the organic priority pollutant 2,4 dinitro phenol to produce growth reduction (Overcash et al, 1982).

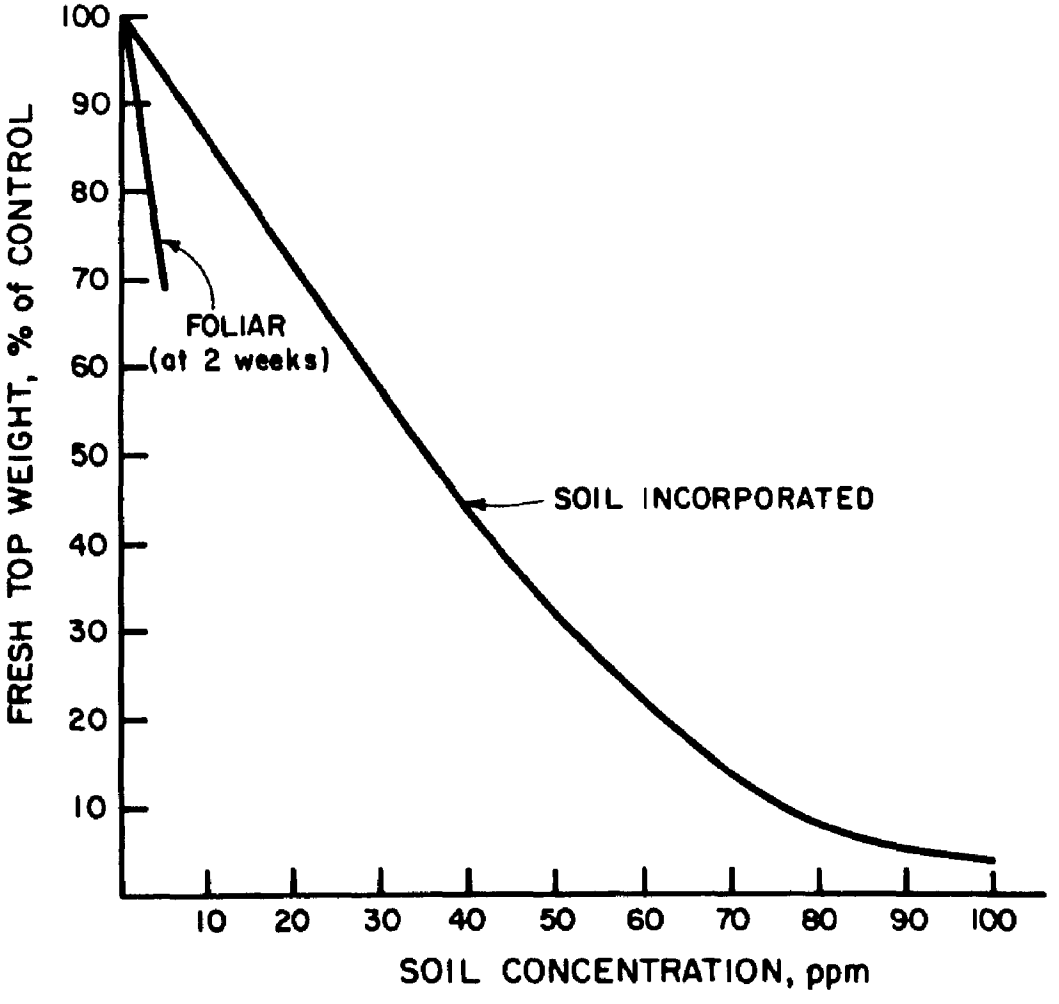

FIGURE 2. Foliar versus soil impact of 2,4 dinitro phenol on corn in Lakeland sand. (Overcash et al, 1982)

As a converse to the phytotoxicity caused by organic compounds, many of these same compounds actually stimulate plant growth when present at low concentrations in the soil. That is, the chemical acts like a herbicide, insecticide or growth hormone when applied in small amounts. An example organic compound group that illustrates this stimulatory effect at low soil concentrations is nonionic surfactants, Figure 3. At high levels, the phytotoxic response is seen, Table 4. Figure 4 depicts the generalized plant growth behavior to applied organic species. Depending on the compound, there are different degrees of low level stimulation and varying critical soil concentration at which phytotoxicity occurs.

In municipal wastewater and sludge land application, the plant response to organics present appears to be below the critical phytotoxic regime. No reported municipal cases of organic chemical impact on crop yields were found. Furthermore, preliminary calculations of anticipated soil loadings compared to critical soil concentration of some specific organic compounds (known from other research studies) indicates that plant phytotoxicity is not likely.

The second broad impact on vegetation from specific organic compounds is the uptake from the soil-water solution. This is a complex process that can involve transfer into the roots, translocation to upper portion of the vegetation, decomposition intraplant, and even migration back to the roots and soil (as growth conditions in the vegetation change). Most frequently, data are available on the measured concentrations of a specific organic in the vegetation (sometimes differentiated by parts) as related to the concentration found in the soil. Examples of these data for two compounds are given in Table 5. In a review of a number of organic compounds and the uptake by various vegetative species, there does not appear to be any precise predictive mechanism for determining plant levels at this time. With very low detection limits, most organics can be taken into plant roots or tissues; however, with simultaneous decomposition and adsorption, the issue of uptake remains quantitatively unclear. In the absence of other soil processes (e.g., in hydroponic culture), there does appear to be an uptake preference for cationic>nonionic>anionic. Since mineral soil adsorption operates with a similar preference (cationic>nonionic>anionic), the actual response of vegetation concentration is quite soil- and chemical-specific.

Once in the vegetation, an organic compound can be degraded as illustrated by the following discussion for nitrosamines.

> "A detailed study of the plant concentration with time (Dressel 1973, 1976) was conducted. The two nitrosamines studied were found to accumulate to a

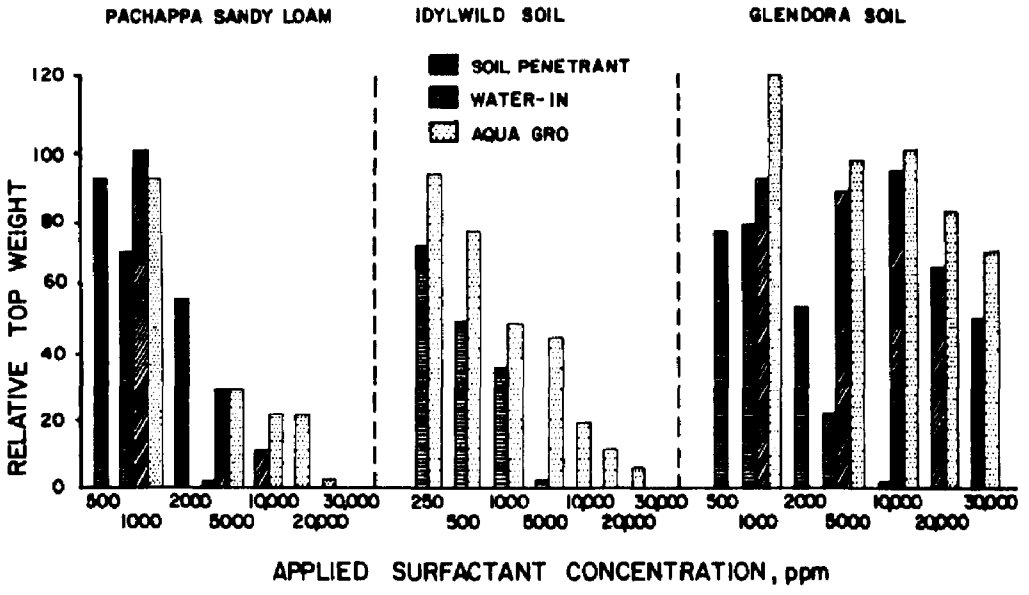

FIGURE 3. Relative top weight of barley grown in soils receiving various applications of nonionic surfactants (Valoras et al, 1976)

Table 4. Summary of critical phytotoxic levels for surfactants (Overcash and Pal, 1979)

Surfactant	Critical Soil Level (kg/ha) Yield <80% of Control		
Three cationics	>10		
Water-In (nonionic)	50	<>250	Pachappa sl
	500	<>1,000	Glendora
Aqua Gro (nonionic)	50	<>250	Pachappa sl
	15	<>25	Idylwild
	1,000	<>3,000	Glendora
Soil Penetrant (nonionic)	25	<>50	Pachappa sl
	~15		Idylwild
	~50		Glendora
Three anionics	<300		

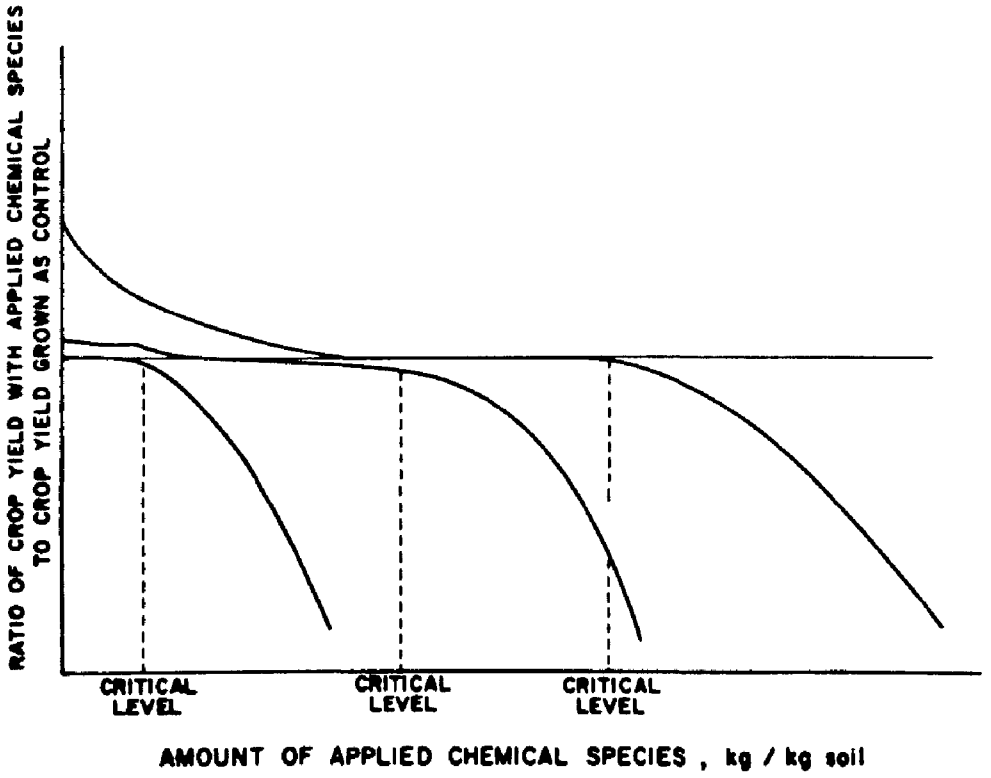

FIGURE 4. Illustrative stimulation and phytotoxic response of organic chemicals applied to soils.

Table 5. Soil and Crop Levels of Benzene Hexachloride and Quintozene

Crop	Soil Level (ppm)		Crop:Soil Ratio (Average Range)
Benzene Hexachloride			
Radish	0.05-5		(0.09-2.9)[1]
Parsley	0.02-2		2.6 (0.5-4.6)[2]
Watercress	0.5-50		0.12 (0.04-0.18)[2]
Potato	0.03-0.4		0.72 (0.5-1.2)[3]
Carrot	0.05-5.0		1.9 (1.0-2.8)[1]
Sugar beet	0.02-0.44	Roots	0.39 (0.23-0.48)[3]
	0.02-0.41	(leaves)	0.24 (0.05-0.37)[3]
	0.02-0.41	(heads)	0.087 (0.03-0.15)[3]
Head lettuce	0.5-1.9		0.23 (0.16-0.3)[4]
	0.02-0.2	(at 67 days)	9.9 (5.1-16.0)[5]
	0.02-0.2	(at 81 days)	3.6 (1.8-5.8)[5]
Grass	0.07-0.14	(roots)	5.2 (2.7-9.0)[3]
	Same	(0-5 cm tops)	1.7 (1.4-2.4)
	Same	(>5cm tops)	0.28 (0.23-0.31)
	0.001-0.03	(roots 1-yr-old)	26 (23-29)[3]
	Same	(>5 cm tops)	0.09
Quintozene			
Potato	1.5-3.2		0.11 (0.03-0.25)[3]
Carrot	0.02-0.16		19 (10-27)[3]
Sugar beet	0.12-0.34	(whole plant)	<0.03 (0.01-0.07)[3]
	0.44-6.3	(67 days)	11.0 (5.3-21)[5]
Head lettuce	0.44-6.3	(81 days)	4.4 (1.4-6.0)[5]
	20-69		0.44 (0.34-0.54)[4]
	0.43-2.0	(roots)	3.8 (1.5-5.5)[3]
Grass	Same	(0-5 cm tops)	2.2 (0.2-1.2)
	Same	(>5 cm tops)	0.03 (0.01-0.03)

References:

[1] Wallnofer et al. (1975)
[2] Hafner (1975)
[3] Smelt (1976)
[4] Casanova and Dubroca (1972)
[5] Dejonckheere et al. (1975)

maximum value at 4-5 days after application and then to decrease to background value within 12 days (Figure 5). The author postulated that an evapotranspiration pathway was responsible for the loss from plants, but the experiment could not discern the relative magnitude of evapotranspiration and metabolic losses. Sander et al (1975) also found that plant-absorbed nitrosamine disappeared with time for Lepidium sativum (garden pepper cress). A final corroboration of the uptake and disappearance of nitrosamines from lettuce was performed with dimethylnitrosamine (Dean-Raymond and Alexander, 1976). A peak concentration appeared to occur at 2 days after application with 150-fold reduction after 15 days.

A total of 21 organic compounds have been found to undergo such losses from vegetation. The extent to which organic compounds of environmental concern (e.g., priority pollutants) exhibit this dynamic intraplant behavior cannot be determined at this time.

IMPLICATIONS RELATED TO THE PRESENT UNDERSTANDING OF ORGANIC COMPOUNDS IN MUNICIPAL WASTE LAND TREATMENT SYSTEMS

The detailed research and field investigations necessary to document the land treatment behavior of specific organics has only just begun. However, it is important to build upon extensive existing information regarding organic species in the terrestrial system. It is also possible to look ahead to establish what some of the more critical research needs might be as well as to put in perspective the relative importance of this area of municipal land treatment research. This perspective, as presented here, is that of the author and was not developed as a concensus of others concerned with municipal effluent or sludge land application systems.

One should compare the situation of specific organic compounds with that of heavy metals in municipal waste land treatment. Metals accumulate in the soil system after undergoing a series of reactions (adsorption, precipitation, chelation) that reduce the availability of metals to vegetation. Since very little of a metal, which is in the soil, is translocated into a plant, exceeding acceptable soil levels would require a very long time to reclaim such a site. That is, exceeding the levels at which vegetation parts are above food metal limits or at which phytotoxicity occurs leads to a less reversible terrestrial condition than found with organics.

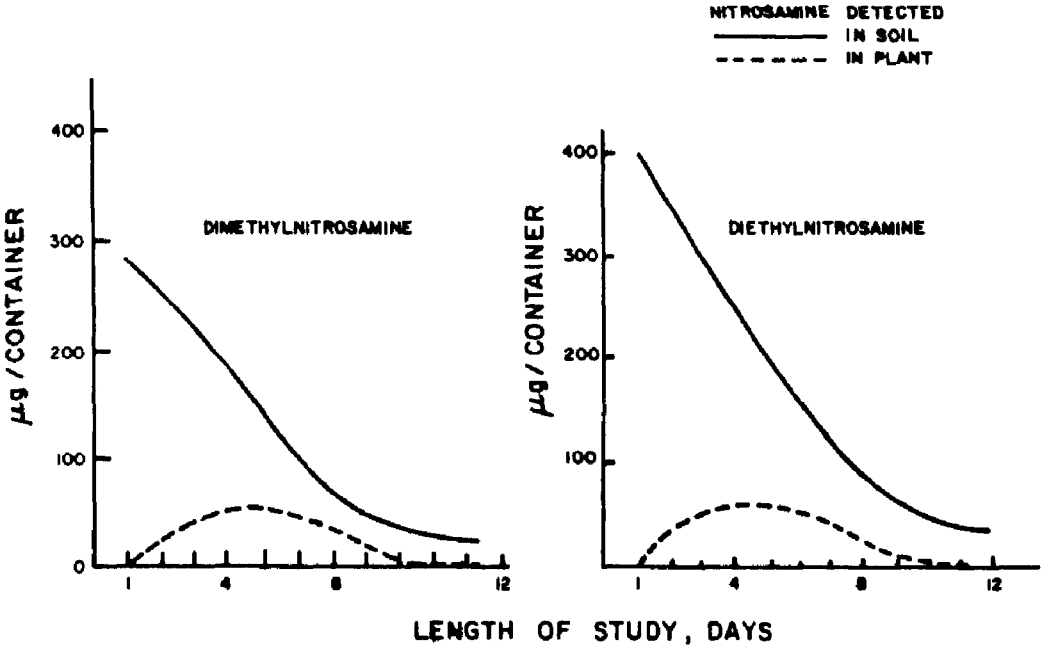

FIGURE 5. Intraplant loss of nitrosamines (Overcash and Pal, 1979).

Organic compounds ("toxic" or other species) are characterized as decomposing, i.e., the amount of a chemical is reduced with time after an application to the terrestrial system. Thus the relationship to vegetation uptake, runoff, microbial effects is constantly changing toward a lower impact as time after application increases. In a sense, any excess, whether as a single dose or accumulated from repeated applications, would be self-correcting in a relatively short period after closure. This conclusion presupposes the typical conditions of municipal sludge and effluent land application. The decomposition half life for most organic compounds is less than one year. Other organics (described previously) with extremely long life are relatively inert or insoluble, factors which lead to a low environmental impact. Thus excess metal loading can be much more difficult to correct than a similar excess with organic compounds.

The observations of vegetation uptake of organics must also be put into perspective. First, there is very little information on vegetation levels at which impact on the human diet would be of concern. Thus the absolute concentrations in vegetation uptake cannot be interpreted as to whether any changes in design criteria are needed. Second, even if organic compound concentrations were of concern in vegetation it would be a short-term consideration. That is, if the vegetation is not harvested but is incorporated into the soil, then the organic compound is subject to further decomposition as the plant material is mineralized. Thus the soil levels would approach background conditions after one or two crop cycles and the organic compound would be successfully treated.

Municipal effluent land application is an alternative to stream discharge. This is an environmental trade-off between the stream receiver and the terrestrial system. On a general basis, the bioaccumulation ratio for an organic compound in soil is low in comparison to that found in an aqueous system. The microbial density and time of contact are very much greater for an organic placed in the terrestrial environment. Thus on a heuristic basis (since the level of organics is fixed by the waste type, municipal effluent) the land treatment of organics appears to have certain inherent advantages.

For a municipal sludge, an alternative to land application is often a landfill. These storage facilites are known to have conditions that cause leaching of substances, including organic constituents. In contrast, migration of organics to groundwater from municipal sludge land application systems has not been reported for reasonably designed systems. Thus since municipal sludge must be managed with some technique, land application has certain inherent advantages. These technology comparisons for land application of both municipal sludge and effluent are developed not as all inclusive, but to emphasize that:

1) municipal wastes must be managed and so one or more alternatives must be selected;

2) use of the terrestrial system for treatment has a number of important advantages for environmental protection.

Discussion of specific organic compounds in relation to municipal land application is often centered on "toxic" organics. Caution is needed with this concept. Often the connotation of toxic has been developed from the response of aquatic systems. The toxicity is very much reduced, of a different character, or non-existent when the receiver system is instead a terrestrial system. In addition, since the somewhat more direct impact (through stream-based drinking water) of stream discharge has a greater human health potential, the terrestrial approach (often with no direct human consumption of crops) may offer a reduced likelihood of toxicity for organic compounds. The direct animal consumption of municipal sludge organics is not covered herein as such effects are primarily in the area of public health.

RESEARCH NEEDS

The information most useful in improving the understanding of the impact of organic compounds in municipal effluent and sludge land treatment were developed from the perspective described above as the available data in 1983. These needs have been limited to a relatively few of higher general priority. Many other meaningful research objectives can certainly be developed as individuals explore this area of land treatment technology.

1) An established algorithm is needed to develop the relationship between the concentration of an organic compound in vegetation and the annual human intake or burden that represents. Then the level of concern from a health perspective can be established and linked to observations of plant uptake. Without these relationships, the research data on the behavior of organic constituents in municipal land treatment systems have only limited use.

2) Since the dominant terrestrial pathway for many organics is decomposition, the rates of loss from actual sludges or effluents applied to land must be measured to verify or correct available data on pure compounds.

3) A review of organic priority pollutants is needed to compare with detailed analysis of organics (by mass spectrometer techniques) found in municipal effluents

and sludges. This comparison should identify any major organic species that might be of environmental concern, but not presently included in the priority pollutant lists.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

Dr. Michael Connor, Harvard School of Public Health: Animal uptake of organic compounds can be an important route in the assessment of the risks of these compounds to humans. There have been instances where cows grazing on sludge-amended lands have developed levels of PCBs in their milk above FDA action levels. I hope that the final position paper includes a discussion of this pathway.

Michael Overcash: Unfortunately, the final paper will not address the direct animal consumption. This is an area primarily of health effects related to specific organic chemicals and not precisely answered at this time.

Stephen B. Smith, Black and Veatch, Denver, CO: I feel that an important item has been missed in the presentation and discussion of your paper and the previous paper on pathogens. This entails that most of the treatment plants have a discharge to a stream and this stream, in turn, is used for a water supply downstream, often in a very short distance. The stream provides an excellent means of transportation for the pathogens and organic compounds.

Therefore, I would submit to you that land application, as a process, is a substantially better choice than the direct discharge to a stream of these pathogens or organics.

Michael Overcash: This is an important and difficult concept since complex risk assessments are needed. I think your conclusion regarding land treatment advantages would probably remain correct.

Jack L. Cooper, National Food Processors Association: In plant uptake, what part of the plant is analyzed, the edible or non-edible? Is there a tendency for the plant to exclude the organics from the edible portions?

Michael Overcash: For some compounds this is certainly true.

Dr. Merry L. Morris, New Jersey Dept. of Environmental Protection: Can you suggest a potential surrogate for soil monitoring of organics degradation from sites at which high levels of organics may be present, such as at spill clean-up sites?

Michael Overcash: No.

Harvey Luce, University of Connecticut: Researchers at Cornell University in a recent communication recommend against all applications of municipal sewage sludges to farm land because of health risks from toxic organic chemicals. Would you like to comment on this?

Michael Overcash: I believe the total evidence will indicate that land treatment is a superior alternative for sludge since sludges must be managed by some technique. Given that choices must be made on which alternatives to use for sludge, I find that I disagree with those researchers at Cornell who oppose the prudent use of land treatment.

William Pounds, Pennsylvania Department of Environmental Resources: Rather than run a priority pollutants analysis on all sludge samples, have you looked at using an indicator parameter, for example TOX? If you exceed a certain level in this parameter, then it would be recommended that a more detailed analysis for organics could be conducted.

Michael Overcash: From our experience I can suggest no surrogate nor do I have any reason to believe that such a surrogate will be readily found.

Richard J. Bull, Toxicology and Microbiology Division, EPA, Cincinnati: Chemicals associated with the specific matrix with which we are dealing are those that we have to be concerned about in order to assess risks attributable to that matrix. The 'priority pollutant' list was drawn up primarily from a perception of important industrial pollutants not because they were the major constituents of municipal sludge. A second complication is that determination of chemicals in municipal sludge has been conditioned by and limited to those for which we have analytical techniques. For example, we have been studying the extent to which removal of mutagenic activity from municipal wastewater correlates with removal of the priority pollutants. Secondary treatment of wastewater was seen to remove priority pollutants present by more than 97%. On the other hand, removal of indirect mutagenic activity measured in the Ames assay was less than 5% in one case and less than 40% in another. This simply shows that behavior of the priority pollutants in the wastewater treatment plant is in no way related to the removal of one type of biological activity considered important in evaluation of human health hazards. It seems reasonably obvious that an effort is needed to specifically identify those chemicals that possess toxicological properties over which we have concern in sludge. Then we need to document the extent to which they can actually impact health and the extent to which they would present themselves in terms of a human exposure via the food chain, drinking water, and other routes.

Michael Overcash: I have emphasized on a number of previous occasions and in this conference paper that it is unlikely that the priority pollutants will long represent the minimum limit of our interest in organics found in wastes.

Robert B. Dean, Lun-Dean Environmental Co., Copenhagen, Denmark: The objectives of "no permanent degradation" is useful for research but does not always correspond to what society is willing to accept. One example of non-degradable organic substance that is regularly applied at loads of kilotons per hectare is asphalt which permanently degrades the soil. The only useful plants which can be raised on such land are industrial plants or factories.

Michael Overcash: Interesting observation, but in all cases we should start with the non-degradation criterion and be sure we know beforehand why we have chosen to relax this standard for terrestrial systems.

Michael Plewa, University of Illinois: The use of the Ames test can identify the presence of mutagens in sludge. However, higher organisms such as plant assays in maize and Tradescantia can detect types of genetic damage not measured in the Ames test. Also, it is important to note that plants can be used as in situ or in laboratory monitors to evaluate the mutagenicity of a sludge over time as well as the mutagenic potency of different sludges. One final note, information exists in the literature that plants can activate some promutagens [i.e., dimethylnitrosamine, benzo(a)pyrene, and 2-aminofluorene] into the ultimate mutagenic agent. Thus, some agents may be taken up from sludges by plants and converted into mutagens.

Phillip K. Hopke, University of Illinois: We have compared mutagenicity of 5 sludges using a battery of tester organisms. In the principally domestic sludge, we found low levels of activity in the extracts using Ames TA98, and TA100, but did not see effects in higher eukaryotic species. Only in the more industrialized sludges did we see mutagenetic or cytogenic effects in the higher eukaryotes. One municipal sludge and effluent were extremely toxic. This plant directly receives effluent from a major chemical manufacturing plant and clearly municipal sludges can be every bit as noxious as direct industrial treatment effluents. A wide range of possible contamination levels must be considered.

Michael Overcash: In comment to Drs. Plewa and Hopke, I share with you the complexity of interpreting mutagen tests for sludges. Our experience is that these tests are less than precise in determining the actual mass loss of mutagens and hence, exposure. Thus, caution is now needed.

George Braude, US-FDA, Washington, DC: 1. Need to consider uptake of organics by (a) animals on pasture, (b) plants [and people, and in food crops] through foliar absorption and adsorption and translocation.

Michael Overcash: We refer you to the conference question of Dr. Connor.

METALS

UTILIZATION OF MUNICIPAL WASTEWATER AND SLUDGE ON LAND -- METALS

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INTRODUCTION

The 1972 amendments to the Water Quality Act, which would spur the construction of advanced wastewater treatment facilities and greatly increase the production of municipal sewage sludge in the U.S. were one year old when university researchers, representatives of the Federal agencies and others met at the University of Illinois to develop a coordinated program for research on application of sewage sludge to land. Individual workgroups identified research needs and priorities on a number of questions, including metals. The dramatic increase in research on land application of sewage sludge in the following decade has helped to provide answers to some of these questions, while, at the same time, raising new issues and identifying new problems. In that same period, Federal, State and local agencies have attempted to formulate guidelines and regulations for the safe utilization and disposal of sewage sludges on land.

The purpose of this paper is to examine the various issues regarding the transmission of sludge-borne metals through the soil/plant/water system and its impact on livestock and human health. This paper is not intended as a comprehensive literature review of metals, a task beyond the scope of this report. It will examine, however, advancements made in the last ten years in our knowledge of the fate of sludge-borne metals when applied to land, and will also discuss the evolution of a regulatory approach to protect the food chain from toxic levels of metals. The reader who comes to this subject with little or no previous background is encouraged to consult a number of excellent reviews which have appeared in the last decade (Sommers, 1980; CAST 1976, 1980; Chaney 1980, 1983a, 1983b; Chaney and Giordano, 1977; Page, 1974; Baker and Chesnin, 1976; Leeper, 1978; Ryan et al., 1982; Walsh et al., 1976).

This paper will examine the transmission of metals from their sources, through the sludge treatment process, into the soil and through the plant and animal systems (Figure 1). Major issues listed in 1973 will be addressed and new research problems identified in 1983 will also

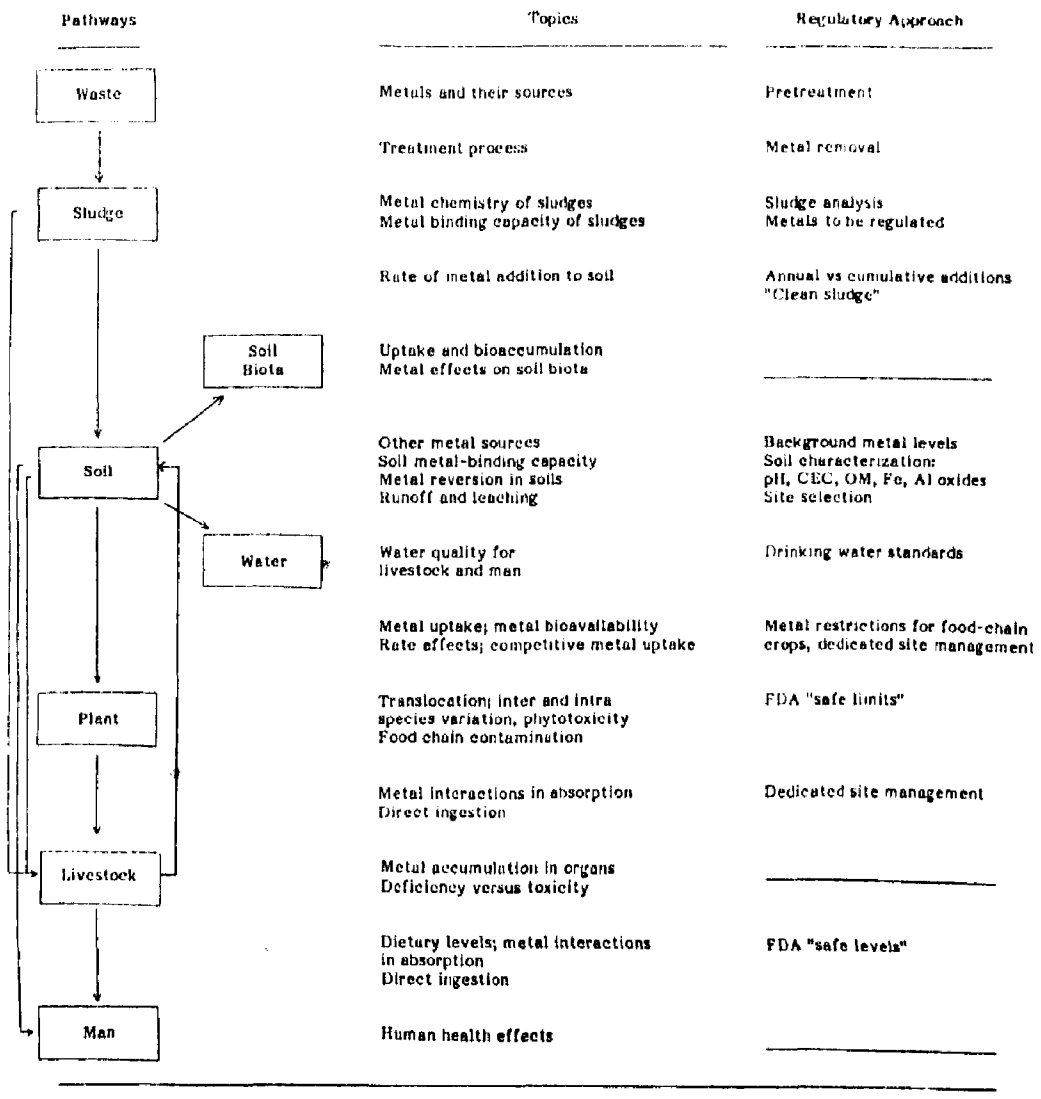

Figure 1. Schematic of the biogeochemical trace element system as discussed in this paper.

be discussed. Present and potential regulatory approaches will also be identified.

METALS IN SLUDGE

Metals of Concern

The 1973 Conference workshop group on Toxic Chemicals identified the following elements as being of primary concern because of either phytotoxicity or potential hazards to animals and man: Cd, Pb, Hg, As, Se, Zn, Cu, and Ni and indicated that, of these, Cd, Pb, Hg, As, Se and Zn should receive the highest priority because of their potential to contaminate the animal and human food chain. At the same conference, Chaney (1973) identified Cd, Cu and Zn as posing the greatest threat to animal and human health through plant accumulation.

The Council for Agricultural Science and Technology (CAST, 1976) identified Cd, Cu, Ni, Zn and Mo as the elements with the greatest potential to accumulate in plants and pose a hazard to plants, animals and humans. Lead, Hg, As, Se were not considered hazardous because "they either have low solubility in slightly acid or neutral, well-aerated soils or, as with Se, are present in such small amounts that the concentration is low in soils." In addition, B was not considered hazardous in non-irrigated regions, but could be phytotoxic to sensitive crops at hot-water soluble concentrations > 1 mg/kg soil. Molybdenum was included in the 1976 CAST report because the authors felt that "Mo may accumulate in plants at concentrations sufficient to cause molybdenosis in ruminant animals without prior warning from plant behavior." By 1977, cadmium was receiving considerable attention as a food chain contaminant, because of the extensive work by Japanese and European scientists (Ryan et al., 1982) and the growing awareness that Cd could readily accumulate in crops at concentrations which could increase the dietary intake of the metal without causing crop phytotoxicity. In 1979, the Federal government (EPA, 1979a) published interim final regulations for the control of Cd entry into the food chain via land application of sewage sludges. These regulations will be discussed in more detail later in the paper, but they dramatically increased the focus on Cd and decreased the attention being paid to potential problems from metal phytotoxicity.

By 1980, Chaney had introduced the term "Soil-Plant Barrier" in assessing the risks to the plant-animal-human system from toxic elements. Chaney (1983b) states that the "Barrier" protects the food chain when one or more of the following processes limit trace element transmission: 1) insolubility in the soil prevents uptake, 2) immobility of an element in fibrous roots prevents translocation to edible plant tissues or 3) phytotoxicity of the element occurs at concentrations of the element in edible plant tissues below that

injurious to animals. Chaney (1983b) lists Cd, Se and Mo as elements which can escape the "Soil-Plant Barrier" and adds Be and Co as metals which may also escape the "Barrier" but are primarily added to soil in industrial wastes. As discussed later, the "Soil-Plant Barrier" can be circumvented when animals directly ingest soil, sludge or vegetation on which soil or sludge particles adhere.

Other Metal Sources

Although widespread application of sewage sludges to land may provide the most ubiquitous source of metal additions to soil, other sources can pose local problems. These include trace amounts of Cd in phosphate fertilizers (Mulla et al., 1980; Williams and David 1973, 1976), contaminated soils in the vicinity of smelters (Hutchinson and Whitby, 1974; Lagerwerff and Brower 1974), mine tailings (Benson et al., 1981; Robinson et al., 1947) localized high natural concentrations of specific metals (Lund et al., 1981), and metal-containing pesticides such as lead arsenate. While these may have local significance, their impact on the total metal burden to man is probably low. In many cases, food chain crops are not grown in the vicinity of these sources, and phytotoxicity greatly reduces the economic viability of these areas for crop production.

Amounts

By 1973, results of extensive monitoring of metals in sludges were available from the U.S., Canada and Europe. Large variations in the concentrations of metals were found. The distributions were skewed towards high concentrations (Dean and Smith, 1973) and arithmetic means overestimated the average metal concentrations of most municipal sludges. Dean and Smith (1973) and, more recently, Sommers (1977) have suggested using median metal concentrations as measures of average contents of municipal sludges, and median values could represent reasonably attainable background levels in sludges after pretreatment to remove known sources. Table 1 gives median concentrations of the major metals found in sludges (Chaney, 1983b). These values are similar to values reported by Page (1974) for treatment plants in the U.S., Canada and Europe. Page (1974) also reported that metal concentrations in sludge were relatively unaffected by the sludge process used.

The 1973 Conference attendees agreed that pretreatment could reduce metal concentrations in sludges, but there was no agreement as to the level to which concentrations could be reduced. Metals from copper drains and galvanized plumbing were expected to maintain fairly high background levels of some metals, especially in soft water areas. National statistics are difficult to obtain since many treatment plants have only recently begun to routinely analyze sludge for metals (Forster

Table 1. Ranges and median concentrations of trace elements in dry digested sewage sludges (Chaney, 1983b).

Element	Reported Range		Median
	Minimum	Maximum	
	mg/kg dry sludge		
As	1.1	230	10
Cd	1	3,410	10
Co	11.3	2,490	30
Cu	84	17,000	800
Cr	10	99,000	500
F	80	33,500	260
Fe	1000	154,000	17,000
Hg	0.6	56	6
Mn	32	9,870	260
Mo	0.1	214	4
Ni	2	5,300	80
Pb	13	26,000	500
Sn	2.6	329	14
Se	1.7	17.2	5
Zn	101	49,000	1,700

et al., 1981). However, examination of individual pretreatment programs reveal that metal concentrations can be drastically reduced. A pretreatment program at the City of Defiance, Ohio sewage treatment plant greatly reduced Cd and Ni levels in its sludge (Miller and Logan, 1979). Concentrations after pretreatment were similar to median values reported by Sommers (1977) for cities in the North Central U.S. Other U.S. cities have achieved median metal sludges through pretreatment.

Forms and Bioavailability

Speakers at the 1972 Conference did not address the question of forms of metals in sewage sludges, and a search of the literature prior to that time reveals very little research on the subject. Dean and Smith (1973) in their paper from the Conference state that "...organic matter in sludge complexes heavy metals...," but this is the only reference made to the metal chemistry of sludges. Surprisingly, characterization of metal forms in sludges was not identified by the conferees as a research need. A year later, Page (1974) reported that "as far as the author is concerned, no data are available on the chemical forms of trace elements which occur in sewage sludges." As recently as 1977, Chaney and Giordano (1977) indicated that "the chemical species of the individual metals present is not known."

Since that time, however, there has been considerable progress in the characterization of metal forms in sludges. Fractionation is one method of characterization. Gould and Genetelli (1975) used elutriation and filtration to separate sludge into four fractions: particulate, supracolloidal, colloidal and soluble. More than 90% of the metals were in the particulate fraction (> 100 mm diameter). Hayes and Theis (1978) developed a fractionation scheme which employed elutriation and filtration as well as acid and EDTA extraction of the particulates to give the following fractions: soluble, precipitated (insoluble), intracellular, and extracellular (adsorbed to the sludge biomass). These two schemes are primarily physical and do not characterize the chemical forms of the metals in sludge. Stover et al. (1976) presented a fractionation scheme which partitioned metals into: exchangeable, adsorbed, organically bound, carbonate and residual. The residual fraction was uncharacterized, but would include such metal solid-phases as sulfide and phosphate. Silviera and Sommers (1977) sequentially extracted soil with 1M KNO_3, 0.005M DTPA and 1M HNO_3 and found that metal extracted was generally proportional to the total content of the metal in the sludge. Distributions between the fractions for a given metal varied between sludges. In general, nitric acid extracted 50% of the metal. Holtzclaw et al. (1978) extracted highly organic sludge-soil mixtures from sewage lagoons with 0.5N NaOH and fractionated the extracted metals into humic acid, fulvic acid and precipitable fractions. Iron, Al and Zn were generally associated with the precipitable fraction, while most of the copper was in the humic acid fraction. Cadmium and Zn were in the precipitable and fulvic acid

fractions. The authors concluded from these results that Cd, Ni and Zn would tend to be more mobile in soil than Cu.

The bioavailability of sludge-borne metals will depend on those processes operating within the sludge which control metal activity in solution and the competing processes operating in the soil. Sludges are a sink as well as a source of metal, and some sludges may retain metals much more strongly than others. As Corey et al. (1981) have pointed out, the specific metal sorption capacity of a sludge may exceed the soil's affinity for the metal, and the bioavailability of the metal will be determined by the chemical characteristics of the sludge rather than the soil. If this is so, the present regulatory approach which considers only the total amount of metal added to soil and the soil's untreated background metal-binding capacity (e.g. cation exchange capacity) may overlook the role of the sludge itself in controlling the bioavailability of sludge-applied metals.

Several important processes are believed to operate in the sludge system to control solution metal activity. These include solid-phase equilibria, adsorption-desorption, chelation and oxidation-reduction. Attempts to characterize solid phase equilibria have assumed the presence of discrete metal forms such as CdS, Pb phosphates, etc.; however, Corey et al. (1981) and Corey (1981) have indicated that many trace metals may be coprecipitated with the major metal cations in sludge (Fe, Al, Ca) during the sewage treatment process. Predicting the solubilities of minor constituents of solid solutions are difficult. Studying the exchange reactions of metal cations on sludge is limited because of the present inability to accurately predict or measure free ion activities and because of the analytical problems associated with working at the very low solution concentrations found in sludge systems. Hendrickson and Corey (1981) have shown that, at low solution activities, trace metals are specifically adsorbed with respect to cations which have much higher solution activities, such as Ca or Mg. In addition, organically complexed dissolved metals interfere with the use of solid phase equilibria models.

Understanding the complexities of the sludge-metal system will require further basic research, but this understanding is critical to explaining the changes with time in metal bioavailability which occur when sludge-borne metals are added to soil.

SLUDGE-BORNE METAL INTERACTIONS IN SOIL

Reaction Processes in Soil

Most microelements (with the exception of boron) are strongly retained by soil and their concentrations in the soil solution are very low. In the Conference Proceedings, Lindsay (1973) pointed out that

much of the work on soil-metal reactions has been empirical with very few basic studies to examine the precipitation reactions of metals in soils. Lindsay felt that, although the exact reaction products of Zn, Cd, Cu and Ni that precipitate in soils were not known, they likely involved substitution in crystalline minerals and amorphous precipitates as well as forming possible discrete silicate compounds; further, adsorption could control metal availability.

Ellis and Knezek (1972) concluded that, in addition to cation exchange, metals are specifically adsorbed to functional groups on soil mineral surfaces. Adsorption rather than precipitation was thought to be an important control on ionic activity at the low metal ion activities normally found in soils, except at pH's above neutrality. Soil organic matter was considered a strong sink for some metals such as Cu and Zn, while the formation of soluble metal-organic complexes reduced free ion activities and metal adsorption by reactive soil surfaces. Leeper (1972) drew similar conclusions after extensive review of metal reactions in soils.

Page (1974) also concluded that sorption reactions predominated for metals added to soils; however, for sludge-borne metals, the metal solid phases existing in sludges may be more stable than the reaction products in soils. He also noted that boron would not be retained by soil since it exists as undissociated boric acid at the pH's normally encountered.

Chaney and Giordano (1977) reported that the relative importance of various metal-soil reactions in reducing plant-available metal levels was not clear, and that no research existed on the effects of several metals added simultaneously. The CAST Report (1980) indicated that "adsorption is the predominant mechanism of trace metal removal from dilute solutions by clay minerals, metal oxides, and organic matter, and by whole soils." The report also indicated a strong pH-dependence of soil adsorption-desorption processes involving Cd and Zn, with pH affecting not only the metal species in solution but also the charge on the soil surface. Cadmium and Zn could exist in soils as discrete precipitates, coprecipitates with iron or aluminum oxides or alkaline earth carbonates, and species bound to soil organic matter (CAST, 1980).

Corey et al. (1981) suggest that adsorption on mineral surfaces is probably the dominant process controlling metal activities in soil solutions and that a competitive Langmuir isotherm can describe the relationship (Harter and Baker, 1977). They felt that trace metals are primarily specifically adsorbed because the selectivity of adsorption of trace metals is on the order of 10^3 to 10^6 over ions like Ca^{2+} and Mg^{2+}. Because of the very low activities of metals in soil solution compared to the activities of more abundant cations (e.g. Ca, Mg, K), metals must be specifically adsorbed if they are to occupy a large percentage of the adsorption sites.

Corey et al. (1981) also concluded that, when metals are added with sludge, solution metal activities would be controlled by adsorption to

soil mineral surfaces as long as the amount of metal added did not exceed the capacity of specific adsorption sites in the soil. In this region, metal activity should increase linearly with sludge metal addition. At high sludge application rates, metal binding sites on the sludge would control metal solubility, and further sludge additions would not increase metal activities. If this is true, the metal chemistry of a particular sludge which is applied to soil at high rates may be as important in controlling metal uptake by plants as the metal binding characteristics of the soil or the total amount of metal applied. This may be significant in the USEPA approach to regulating metal additions to soil, as discussed later. The considerations of Corey et al. (1981) also indicate that the ratio of metal to specific sorption capacity should be important in controlling metal ion activity.

Metal Reversion in Soils

Chaney (1973) in the Conference Proceedings reported that "metals revert with time to chemical forms less available to plants." He gave the example of fertilizer Zn reversion which had been shown to be quite rapid. He indicated that the processes involved were poorly understood but were most rapid in calcareous soils.

Chaney and Giordano (1977) attributed the term "reversion" to Leeper (1972) and cited several cases of reversion of Cu, Zn, Fe and Mn when added to soil as fertilizer salts or chelates. Chaney and Giordano (1977) also reported that "in nearly every study on the effect of incubation on the extractability of microelements added as sludge or refuse compost, a continuing decline in extractability is observed."

The CAST (1980) Report carefully examined the data on residual availability of Cd and Zn to crops after termination of sludge application. The authors concluded that the residual effects were indeterminate. Although there was some evidence that bioavailability of the metals may remain constant or decrease over a period of years at a constant pH, they felt that soluble metal concentrations would increase if pH decreased.

It appears that, if metal reversion in soil does occur, the effects are small and/or slow, and can be reversed by increasing soil acidity. The lack of strong reversion processes in soil, especially for Cd (Lloyd et al., 1981), implies that land used for food-chain crops which has received large doses of sludge-borne metals must be managed for the forseeable future to minimize metal uptake by crops. Further research is needed on the long-term changes of sludge-metal forms in the soil, and the effects of such factors as sludge-metal chemistry, soil chemical environment and sludge organic matter decomposition on metal availability to plants. The "time bomb" effect whereby sludge-applied metals become more available over time (at constant pH) suggested by Beckett and Davis (1979) has not been supported by research results.

Soil pH, Organic Matter Content and Cation Exchange Capacity Effects on Soil Metal Binding Capacity

Chaney (1973) identified pH, cation exchange capacity (CEC) and organic matter (OM) content as soil factors controlling metal toxicity to plants. Soil pH might be more important than the amount of metal added. Soil OM served as a source of CEC for metal cations and also formed chelates with the metals, especially Cu and Ni. Chaney pointed out that, at low soil pH's, chelation would decrease metal availability to plants, but, at least in the case of Zn, chelation increased availability at high soil pH's. He stated that CEC was important in binding all cations, and "a soil with a high CEC is inherently safer for disposal of sludge and effluent than a soil with low CEC." The Conference Workshop Group 8 (Plant Response-Toxic Chemicals) felt that research on empirical correlations of the toxicity of metals with soil parameters, including CEC and pH was needed.

Soil pH: Soil pH is the parameter most consistently identified in the literature as controlling metal availability. With the exception of Mo and Se, all microelements are more labile at low pH, due to hydrolysis of hydroxide species (Ellis and Knezek, 1972) and the solubility of other solid phase minerals such as carbonates and phosphate. Low pH also lowers the adsorption of metals to pH-dependent specific adsorption sites on mineral surfaces (H^+ is a competitive cation), and lowers the CEC of soil OM.

Sludge additions to neutral or slightly acid soil tend to decrease pH somewhat depending on $CaCO_3$ equivalent content of the sludge (hardwater; lime in dewatering). Nitrification of sludge NH_3, oxidation of organic matter and sulfur containing organic compounds in sludge, and oxidation of sulfides in anaerobic sludges all increase soil acidity, while the high buffer capacity of sludge organic matter tends to ameliorate these effects. Sludge additions tend to decrease the pH of calcareous soils somewhat (CAST, 1980). At normal rates of sludge addition (no greater than the nitrogen requirement of maize), the soil buffer capacity and soil pH will determine the pH of the soil-sludge mixture. The CAST Report (1980) concluded that the ability of calcareous soils to maintain high soil pH with sludge additions greatly reduced the risk to the food-chain from Cd and Zn uptake by crops for crops grown on these soils.

Chaney et al. (1978a) limed acid soils that had not received sludge for four years and found that Cd uptake was reduced. Liming also appears to reduce the availability of native soil metals (Sommers, 1980).

The soil pH level considered to be effective in reducing metal uptake by plants seems to be $\sim$6.5 (Chaney, 1973; CAST, 1976; CAST, 1980) and is required by USEPA (EPA, 1979a) for sludge applied to land for the

production of food-chain crops. This pH level is usually attributed to Chumbley (1971) who developed guidelines in England for the control of phytotoxic levels of Zn, Cu and Ni in soils. However, there is no indication from the work of Chumbley and others that there is any specific basis for selecting this pH other than observed effects on metal uptake (CAST, 1980).

Solid-phase equilibria of the metals in soil may provide evidence for the effect of pH on metal solubility (Lindsay, 1979) but that analysis has not been done, and is complicated by not knowing if the solid phases controlling metal solubility are sludge derived or are soil minerals. Also, activities of organometallic complexes are not as affected by pH as are free ion activities. There is no evidence from recent research that pH 6.5 is not an effective control for metal uptake by crops, and, except for acid soils strongly buffered by Al, this pH level is easily attained and maintained by periodic liming. On more acid soils, as found in unglaciated regions of the eastern U.S., raising pH to near 6.5 is difficult and expensive. Once attained, this pH can be maintained with lime additions every two to three years.

Cation exchange capacity: With the exception of As, Se, Mo and B, the elements of concern are all cations and, as such, are attracted by electrostatic forces to the predominant negative charges on soil minerals and organic matter. Ellis and Knezek (1972) pointed out that the metal cations, in addition to cation exchange, are specifically adsorbed by covalent bonds to functional groups on clay surfaces. They also reviewed early work (Bower and Truog, 1941; DeMumbrum and Jackson, 1957; Hodgson, 1963) which showed that hydrolysis of aquo groups coordinated to the metals significantly affect their adsorption by soils.

Most authors (Chaney, 1973; Haghiri, 1974; Latterell et al., 1976; CAST, 1976 and 1980) recognize cation exchange capacity (CEC) as one of the soil properties which are related to soil retention of metals. Leeper (1972) was the first to suggest that soil CEC be used as an index of the amount of metal (Cu, Ni, Zn) which could be safely added to soil without causing phytotoxicity. The North Central Regional Research Committee on Land Application of Sewage Sludges (NC-118) (EPA, 1980) proposed in 1974 that medium textured soils in the region could safely accept cumulative amounts of Cd, Ni, Cu, Zn, and Pb based on their relative phytoxicities (in the case of Ni, Cu, Zn) and risk of food chain contamination (Cd, Pb) as well as their relative uptake by crops. In 1976 the Committee expanded the guidelines on cumulative metal additions. Soils were arbitrarily divided into three groups by CEC (< 5, 5-15, > 15 meq/100g soil) with cumulative additions as given in Table 2. (The Ni levels were later raised to be the same as for Cu).

Even though no research showed a direct link between Cd uptake by plants and CEC, the Committee felt that CEC was a readily and routinely measured soil property that directly related to the ability of a soil to

Table 2. Cumulative metal additions in soil.

	Cumulative Metal Additions (lbs/acre)		
	<5 meq/100g	5-15 meq/100g	>15 meq/100g
Cd	5	10	20
Ni	50	100	200
Cu	125	250	500
Zn	250	500	1000
Pb	500	1000	2000

minimize Cd solubility. The Committee also recognized that the effects of CEC are compounded by the specific soprtion characteristics of soil organic matter, Fe, Mn and Al oxides. Organic matter binds metals by chelation as well as by cation attraction. Iron, Mn, and Al oxides can specifically adsorb metals yet do not have sufficient net negative charge to bind metals by cation exchange. Increasing pH will increase CEC of organic matter and other soil minerals with pH-dependent charge, but an increase in metal retention may also be due to decreased solubility of the metals adsorbed to solid phase minerals.

The CAST Report (1980) stated that "..CEC is best viewed as a general, but imperfect, indicator of the soil components that limit the solubility of Cd and Zn (i.e., organic matter, clays, and hydrous oxides of iron, aluminum and manganese) instead of a specific factor in the availability of these metals."

Corey et al. (1981) determined relative selectivity coefficients of Cd with Zn, Cu, Ni and Fe for EDTA, and solid phase and soluble complexes in a soil which had received 320 mt/ha of sludge. Selectivity coefficients were similar for the three types of metal complexes and, even at the relatively high metal levels in the soil, there was still considerable selectivity for the different metals. This indicates that these specific sorption sites are much different than the cation exchange sites suggested by Bittell and Miller (1974) and Cavallaro and McBride (1978). Corey et al. (1981), however, do not distinguish between sludge and soil sites in their system.

Further research is needed to clarify the role of specific versus non-specific metal absorption, and to determine the nature, persistence and relative importance of sludge versus soil metal-binding sites.

Soil organic matter: Soil OM can bind metals through cation exchange and through the formation of organometallic complexes (Mortensen, 1963; Hodgson, 1963; Schnitzer, 1969). On the other hand, soluble metal-organic complexes can reduce sorption of metals and increase their availability (Stevenson and Ardakani, 1972). Soluble chelates and complexes allow faster diffusion from soil solid phases to plant roots (Lindsay, 1974). Stevenson and Ardakani (1972) reviewed the relative stabilities of metal-organic complexes of the trace nutrients and found at pH 5 that they were in the order: Cu > Pb > Fe > Ni > Mn > Co > Zn. More recent work (Mahler et al., 1980) has shown that only a small percentage of Cd found in solutions from sludge amended soil is organically complexed.

It has only been in the last six or seven years that researchers have begun to characterize the role of sludge and soil organic matter in metal complexation and retention. Sposito and co-workers (Holtzclaw et al., 1976, 1978) extracted the humic and fulvic acid fractions of sewage sludges and found that metals differ in their distribution in these fractions. Other workers (Davis and Leckie, 1978; Newton et al.,

1976; Elliott and Huang, 1979; Chubin and Street, 1981; Elliott and Denneny, 1982) have studied the effects of organic complexing ligands on the adsorption of metals by soils and minerals. Ligands such as EDTA which form strong metal complexes reduced metal adsorption while other organic ligands had less effect. Adsorption of the ligand itself can also affect metal adsorption.

While it is clear that both sludge and soil derived organic matter can affect metal retention in soil and, therefore, metal uptake by plants, this area remains poorly understood. Microbial biomass may also bind metals (Kurek et al., 1982). Because of its highly complex nature, advances in the knowledge of metal-organic matter reactions have been slow. Recent developments in analytical methodologies should provide needed new information on the chemistry of organometallic complexes and their role in the retention and bioavailability of metals in soil.

Extraction Procedures for Measuring Bioavailability of Sludge Borne Metals

Melsted (1973) in his Conference paper stated that "from past experience, and the inability to establish uniform soil testing methods for the determination of available essential heavy metals like zinc and copper under a wide range of soil conditions, the likelihood of developing standard methods of analyses for the toxic elements are rather remote." Melsted suggested that, although methods of soil analysis for available forms of some metals had been developed on a regional basis, procedures for other metals can only be developed when their available chemical forms are known. Workshop Group No. 8 identified an immediate need to develop laboratory diagnostic techniques for measurements of available toxic metals in soils.

Cox and Kamprath (1972) discussed tests that were based on four types of nutrient pools: (1) water soluble, (2) exchangeable, (3) adsorbed, chelated or complexed and (4) minerals. Soluble concentrations of the micronutrient metals (and other metals) are very low in soils, and water extraction has been of little value as an extractant for metals. However, hot water-extractable B has long been used as an index of boron availability to crops (Wear, 1965). Exchangeable Zn and Mn have been successfully correlated with plant uptake (Cox and Kamprath, 1972), but the exchangeable levels of most metals in soil are low. Cox and Kamprath (1972) felt that metal release from soil minerals would contribute very little to plant uptake from highly weathered soils, but could be an important source in arid and semi-arid environments.

The most routinely used soil test procedure for available metals is DTPA-TEA extraction (Lindsay and Norvell, 1978) which is primarily used for Zn, Cu, Fe and Mn. In recent years it has also been used to extract other metals (Bingham et al., 1975; Kelling et al., 1977, Street et al.,

1977). Korcak and Fanning (1978) compared DTPA with double acid (0.05N HCl + 0.025N H_2SO_4) for extracting Cd, Cu, Ni and Zn applied to soil at pH 5.6 and 6.5 as inorganic salts or in sewage sludge. The two procedures were highly correlated with each other, but the DTPA procedure was more sensitive to reduced solubilities of Zn and Cu at the higher pH. Neither procedure correlated well with plant uptake when the metal was applied with sludge, although correlations were improved when the data for the two pH regimes were separated. The DTPA-TEA method is sometimes replaced by an ammonium bicarbonate-DTPA test which allows multielement analysis by plasma spectrometry (Soltanpour and Schwab, 1977).

Baker and his coworkers at Pennsylvania State University (Baker and Amacher, 1981; Baker, 1980; Baker et al., 1976) have developed a quantity/intensity method of soil testing for nutrients and trace metals. DTPA is equilibrated with soil metals and interpretations based on both the labile pool of metal able to equilibrate with DTPA (quantity) as well as the free ion activity (intensity) are made. Baker and Amacher (1981) give high, normal and low quantity and intensity levels for Al, Pb, Ni and Cd.

Haq et al. (1980) examined the ability of nine extractants to predict metal uptake by swiss chard from 46 Ontario soils with varying degrees of metal contamination. No one extractant successfully predicted uptake of Cd, Ni, Cu or Zn, but several of them, including DTPA and ammonium acetate gave satisfactory results when combined in the regression equation with other soil properties such as pH, OM and CEC.

More recently, workers have attempted to characterize Cd in soil solution by saturation extraction or extraction with 0.1M $Ca(NO_3)_2$ (Corey et al., 1981; Lampert et al., 1982). Mahler et al. (1982) found that saturation extract Cd correlated with Cd concentration in lettuce, chard, tomato and corn, but the relationships were different for calcareous and acid soils. The calcareous soils had more extractable Cd than the acid soils, but the difference was not reflected in plant uptake. This may have resulted from use of Cd SO_4-amended sludge, as other studies did not find this pH effect on Cd solubility. Corey et al. (1981) suggest that saturation extracts or electrolyte extraction may be more sensitive to changes in the soil-sludge chemical environment than more rigorous extractions, but the results of Mahler et al. (1982) indicate that factors other than solution concentration may affect metal uptake by plants.

As identified at the 1973 Conference, reliable soil test procedures for the determination of available metals in soil need to be developed. However, to date no such test or tests are available, and it is likely that no one test or tests will be applicable to all metals on all soils. Further research should emphasize regional development of soil test procedures for specific metals and crops.

Movement of Sludge Borne Metals by Runoff and Leaching

Workshop Group 8 identified as a long term research needs a trace element inventory which would include "leakage from the soil-root zone" and "erosion from surface soils into lakes and streams." Only a very small fraction of trace metals in soils is soluble. Page (1974) reported that concentrations of the metal cations in neutral soil are usually < 0.05 mg/ml, and, in the pH range common to soils (5-8.5), soluble metal concentrations do not exceed 0.25 mg/ml. Thus, metal cations do not leach readily from soil. However, under conditions of heavy applications of sludge and low soil pH, there have been reports of metal movement to depths up to several meters. Lund et al. (1976) found increased levels of Cr, Cu and Zn to a depth of 2m beneath a sludge drying bed that had been used for more than 20 years, while elevated Ni and Cd concentrations were detected at 3.5m. Kirkham (1975) found movement of Cd, Cu, Ni and Zn to a depth of 61cm below a 3-year old sludge lagoon. Hinesly et al. (1972) and Boswell (1975) found movement of metals to depths up to 45 cm with heavy applications of sludge (135-168 mt/ha) to agricultural land. Emmerich et al. (1982) studied the movement of sludge-applied metals to soils in a column study and found no movement of metals had occurrred from the 476 mt/ha sludge application after leaching with 5m of water over a 25-month period. Although soil pH's were in the range 5.2-8.0 prior to sludge application, nitrification of sludge nitrogen greatly lowered pH in the soil zone below the sludge.

At annual sludge application rate < 15 mt/ha which are recommended for efficient crop removal of sludge-nitrogen, leaching of metals should pose little threat to groundwater resources for all but those sludges with very high metal levels.

Because of the strong retention of metals by soils, loss of metals by erosion and runoff transport of sediments should pose a greater threat to water supplies than leaching. This would be particularly true in instances of heavy applications of sludge to land adjacent to water bodies. However, no research reports indicate that this occurs (Dowdy et al., 1980). Even if some erosion and transport of soil containing sludge-applied metals occurs, the low solubility of the metals will result in very low metal levels in the stream or lake water. Proper site selection and sludge application management should be adequate to protect our water resources from metal contamination from surface runoff and erosion.

METAL UPTAKE BY PLANTS

Introduction

Uptake of sludge-borne metals by plants is important for three basic reasons. First, plants require several micronutrient elements for normal growth; sludge can serve as a Zn, Cu, or Fe fertilizer. Second, if excessive amounts of some elements are absorbed by plants, reduced yield or plant death can result; sludge Zn, Cu, and Ni have caused phytotoxicity when poorly managed. Third, plant uptake of some microelements is the primary route for them to cause food-chain risk.

In 1973, little was known about plant uptake or phytotoxicity of sludge-borne metals (Chaney, 1973). Results from excessive application of microelement fertilizers or pesticides, or on soils developed over metal-rich geologic anomalies provided nearly all the field data. Poor sludge practices in the United Kingdom provided evidence that sludge metals could cause phytotoxicity (Chaney and Giordano, 1977).

Much has been learned about metal uptake by plants in the last decade. However, our knowledge remains incomplete -- especially on phytotoxicity, and on basic chemistry of metals in sludge-amended soils. Many of the examples reviewed in this section will deal with Cd, partly because of the human health implications of Cd, resulting in much research on this metal.

Sludges as Microelement Fertilizers

Chaney and Giordano (1977) summarized results from studies of using sludge to correct micronutrient deficiencies, and noted that "It seems reasonable that wastes which are otherwise safe for use on cropland could be applied to satisfy N or P requirements, and the waste could simultaneously correct any Zn, Cu, etc., deficiencies." Since that time, McClaslin and O'Connor (1982) have shown that sludge could also correct Fe deficiency of sorghum in New Mexico soils. Irradiated raw sludges were more effective than digested sludge, perhaps indicating that reduction of soil iron in inert crystalline ferric oxides to form the more plant available hydrous ferric oxide contributes to Fe availability along with sludge organic matter-bound and chelated Fe.

At this time, sludge has been experimentally shown to correct Zn, Cu, and Fe deficiencies. Boron, Mn, and Mo in sludge are seldom at high enough concentrations to serve as fertilizers in the field. Composted sludge provided all required microelements for horticultural potting media (Chaney et al., 1980).

Phytotoxicity

Excessive heavy metal absorption can kill a plant, or allow normal environmental stresses to kill the plant. Of the elements commonly present in municipal sludge, Zn, Cu, Ni, and Mn are likely to cause phytotoxicity with improper management of soil pH or cumulative metal application. In a few industrially polluted sludges, Co could also cause phytotoxicity. Sludge Mn is not generally a cause of toxicity, but soil Mn can become more available due to sludge use, and Zn plus Mn phytotoxicity can occur in strongly acidic soils (White et al., 1979b).

Excessive Cu, Ni, or Co usually cause injury of plant roots, and inhibit translocation of Fe from roots to shoots; then phytotoxicity is expressed as yellow (chlorotic) young leaves which is metal-induced Fe-deficiency. Zinc and Mn are translocated more freely to the leaves; although Zn and Mn can injure roots and induce Fe-chlorosis in some species, their toxicity is usually manifested through injury of older leaves and reduction of plant growth through interference with biological processes (Foy et al., 1978). Some refuse composts are rich enough in B to injure sensitive crops when applied at high rates (Purves and Mackenzie, 1974; Sanderson, 1980).

Potential phytotoxicity is affected strongly by soil pH, other soil properties, crop species and cultivar, sludge metal concentration, and climatic factors. Soil pH, sludge application rate, and crop strongly affect phytotoxicity (Table 3). Most vegetable crops and legumes (dicots) are relatively sensitive to high levels of sludge metals in acidic soils.

Although soil OM or CEC are important in toxicity of added metal salts (White and Chaney, 1980), there is little evidence of the importance of these properties for sludge-applied metals (CAST, 1980). Sludge metal sorption properties can dominate over soil properties in controlling metal uptake (Corey et al., 1981). This could result from sludge organic matter, hydrous Fe oxide, and/or phosphate. Webber et al. (1981b) found that sludge Cu could not cause phytotoxicity unless the sludge contained appreciably over 1000 mg Cu/kg. High metal concentrations are also implicated in the studies by Williams (1980) and Marks et al. (1980) where sludges rich in Zn, Cu, or Ni caused severe yield reductions in vegetable crops when soil pH was low.

Most commonly, sludge metal phytotoxicity is observed only with low soil pH (Marks et al., 1980; Chaney et al., 1978a). Lutrick et al. (1982) applied a low-metal liquid digested sludge to an acidic fine-textured soil in Florida. At high cumulative sludge rates (300 mt/ha; over 700 kg Zn/ha), soil pH declined substantially due to oxidation of sludge N and S, reaching pH 4.6. Soybean suffered Zn + Mn phytotoxicity; seedling leaves contained as high as 1190 mg Zn/kg and 725 mg Mn/kg. Toxicity was easily corrected by liming. In a field study with a low metal sludge the soil at pH 6.2 and 5.4, received

Table 3. Relative sensitivity of crops to sludge-applied heavy metals (Chaney and Hundemann, unpublished).*

Very Sensitive†	Sensitive‡	Tolerant§	Very Tolerant#
chard	mustard	cauliflower	corn
lettuce	kale	cucumber	sudangrass
redbeet	spinach	zucchini squash	smooth bromegrass
carrot	broccoli		'Merlin' red fescue
turnip	radish	flatpea	
peanut	tomato		
	marigold	oat	
ladino clover		orchardgrass	
alsike clover	zigzag, Red, Kura and crimson clover	Japanese bromegrass	
crownvetch	alfalfa	switchgrass	
'Arc' alfalfa	Korean lespedeza	red top	
white sweetclover	sericea lespedeza	buffelgrass	
yellow sweetclover	blue lupin	tall fescue	
	birdsfoot trefoil	red fescue	
weeping lovegrass	hairy vetch	Kentucky bluegrass	
Lehman lovegrass	soybean		
deertongue	snapbean		
	Timothy		
	colonial bentgrass		
	perennial ryegrass		
	creeping bentgrass		

* Sassafras sandy loam amended with a highly stabilized and leached digested sludge containing 5300 mg Zn, 2400 mg Cu, 320 mg Ni, 390 mg Mn, and 23 mg Cd/kg dry sludge. At 5% sludge, maximum cumulative recommended applications of Zn and Cu are made.

† Injured at 10% of a high metal sludge at pH 6.5 and at pH 5.5.

‡ Injured at 10% of a high metal sludge at pH 5.5, but not at pH 6.5.

§ Injured at 25% high metal sludge at pH 5.5, but not at pH 6.5, and not at 10% sludge at pH 5.5 or 6.5.

\# Not injured even at 25% sludge, pH 5.5.

298 kg Zn/ha at 224 mt/ha of applied sludge. Romaine lettuce contained only 67 and 225 mg Zn/kg at the respective pH levels (Chaney et al., 1982).

Crops differ remarkably in metal uptake, and in tolerance of soil metals (Table 3). Crops differ somewhat in tolerance of foliar Zn, Ni, and Cu, and substantially in tolerance of foliar Cd (Bingham, 1979). Mahler et al. (1978) and Baxter et al. (1974) noted that Zn or Cd in leaves associated with 25% yield reduction of lettuce (Cd) or chard (Cd, Zn) varied with soil pH; higher foliar metal levels were tolerated in acid soils.

Further, crops differ in relative toxicity of soil Zn, Cu, and Ni, and soil pH affects the relative toxicity coefficients for added soil metals (Mitchell et al., 1978). Beckett and coworkers have studied relative toxicity and additivity of metal toxicity (Beckett and Davis, 1978; Beckett and Davis, 1979), although they focused only on foliar metal levels. Most of their conclusions are based on nutrient solution or sand culture studies using soluble metal salts, which may underestimate tolerance of foliar metals. They concluded that Zn and Cu are largely antagonistic rather than additive. Metal additivity did not occur until 0.5 to 0.67 of the "upper critical" foliar concentration was reached. They concluded that the Zn (equivalent) equation of Chumbley (1971) seriously underestimated safe sludge loadings because it presumed total additivity. Mitchell et al. (1978) came to a similar conclusion because the equation's relative toxicity coefficients for Zn, Cu and Ni (1:2:8) were higher than observed in a metal interaction/additivity study with wheat and lettuce. They also reported that excessive metals could interact by competing for sorption sites in the soil. Added Cu or Zn increased Ni in soil solution and Ni uptake by crops and hence Ni phytotoxicity.

Knowledge of tolerance of foliar metal concentrations has progressed, although agreement among researchers is incomplete (Table 4). Some differences occur because chelated Fe in nutrient solution is sorbed differently by monocots and dicots. Davis and Beckett (1978) found barley tolerated only 200 mg/kg foliar Zn in sand culture (upper critical level), while Boawn and Rasmussen (1971) found small grains tolerated 500 mg/kg foliar Zn when grown in soil (25% yield reduction). White et al. (1979a) found tolerance to 400-600 mg/kg foliar Zn among soybean cultivars.

Common Errors in Study of Toxic Element Uptake by Plants

Researchers have noted several major errors in experimental methods (CAST, 1980). First, the source of metals added may strongly affect the result, and second, the location in which the experiment is conducted (field vs. pot studies) may affect the results. Potential problems in nutrient solution vs soil studies were noted above.

Table 4. Different estimates of foliar metal concentrations associated with appreciable yield reduction due to single element metal toxicity.

Element	Melsted (1973)	Chaney et al. (1978a)	Davis et al. (1978) Davis & Beckett (1978) Beckett & Davis, (1977)	
		25% YR*	"Critical"	10% YR
	mg/kg dry leaves			
Zn	300	500	200	290
Cu	15	20-40	20	20
Ni	3	50-100	11	26
Cd	3	varies	8	15
Mn	300	500	--	--

* Denotes yield reduction

The first error is usually called the "salt vs sludge" error. When metals are added as soluble metal salts (sulfate, chloride, nitrate), they nearly always cause greater plant uptake and toxicity than when applied in sludge. Metals in digested sludges should be much nearer to equilibrium with sludge organic matter, with sparingly soluble inorganic compounds, or occluded in $CaCO_3$ or other minerals (Corey et al., 1981). Bloomfield and McGrath (1982) recently reported that a significant difference remains even when metals are incubated with sludge after digestion. Davis and Carlton-Smith (1981) reported results from adding metals to raw sewage to subsequently obtain digested sludges with desired compositions. Uptake of Cd and Zn by radish was similar for the prepared sludges and sludges obtained from sewage treatment plants.

Sludge contains organic matter and hydrous Fe and Mn oxides, both of which increase the specific metal sorption capacity of the soil (Soon, 1981; Garcia-Miragaya and Page, 1978) and reduce plant availability of Cd (Singh, 1981). Further, metal salts lower the pH of the soil by displacing adsorbed H+; few authors have corrected soil pH to the level of the control. Alternatively, soluble metal acetates raise the soil pH as the acetate is oxidized; metal oxides or carbonates also raise pH. Adding equimolar mixtures of $NiSO_4$ and $Ni(Acetate)_2$ caused little change in soil pH even at very high Ni applications which lowered pH over 1.5 units when applied as $NiSO_4$ (Chaney and Leech, unpublished). Many authors have reported the salt-sludge difference (e.g. Dijkshoorn et al., 1981). In light of research summarized in this paper, it is inappropriate to purport that metal salt additions simulate sludge metal additions.

The second error is generally called the "greenhouse vs field" error. Greenhouse and growth chamber studies offer greater manageability and reproducibility and lower cost than field studies (and do not suffer from foraging wildlife). However, crop Zn, Cd, and Mn are increased 1.5 to 5 fold over field studies with the same soil, sludge, and crop (deVires and Tiller, 1978; deVries, 1980). Davis (1981a) also found higher uptake of Cu in outside and greenhouse pots than in the field. This appears to result from 1) use of NH_4-N fertilizers which lowers soil pH more in pots than in the field; 2) higher soluble salt levels in pots than field due to smaller soil volume for required fertilizer salts; 3) confinement of plant roots to the small volume of treated soil in pots, and 4) abnormal watering pattern and relative humidity in greenhouse pot studies. The smaller the pot, the greater the error. Another common error in pot studies is inadequate supply of fertilizer nutrients to obtain maximum growth rates (Terman, 1974).

The pH of the soil immediately adjacent to plant roots (the rhizocylinder) is especially important in plant uptake of microelements. When roots absorb NH_4-N, the pH of the rhizocylinder soil declines, and when the roots absorb NO_3-N, the pH rises, and thus, the form of N absorbed by the root has a strong influence on metal cation uptake (Barber, 1974). In the field, most crop N is absorbed from the soil as NO_3-N; this would raise rhizocylinder pH (Nye, 1981). Use of NH_4-N

fertilizers also causes the pH of the bulk soil to decline since H+ is generated when NH_4-N is oxidized to NO_3-N (Jolley and Pierre, 1977).

When metals are added as a constituent of sludge, plant uptake depends on time since application. Rapid breakdown of organic matter (especially at higher sludge rates) supplies organic ligands and NH_4-N, both of which facilitate metal diffusion and uptake (Barber, 1974; Lindsay, 1974; Wallace et al., 1977). Radish absorbed very high levels of Cu and Zn and suffered phytotoxicity when grown on freshly sludge-amended and heated (37 C) soils (Sheaffer et al., 1981); five crop years later, soil temperature had almost no effect on metal uptake, and healthy crops resulted (Chaney et al., unpublished) (Table 5). The higher soluble salts in pot studies or high rates of recent sludge additions can increase soluble metals by forming complexes (Cl) or ion pairs (SO_4) with metals, thereby increasing metal diffusion and uptake (Bingham, 1980).

Thus, although pot studies in greenhouse and growth chamber allow the control of experimental variables needed to characterize details of the soil-plant interactions of metals, most researchers agree that regulations must be based upon field research. First year results, and annual sludge rate in excess of crop N requirements cause errors even in the field. Further, the error due to study of high metal concentration sludges discussed earlier precludes obtaining results applicable to use of low metal sludges in agriculture.

Plant Differences in Metal Uptake and Translocation

Plant factors: Crop plants differ widely in uptake of an element, all other factors held constant (Chaney and Giordano, 1977; Sommers, 1980; Peterson and Alloway, 1979). Growing on the same sludge-amended soil, spinach may contain 10 times more Zn than tall fescue, orchardgrass 15 times more Ni than corn, and chard 5 times more Cu than tall fescue. A recent report by Davis and Carlton-Smith (1980) shows relative concentration of Zn, Cu, Ni, Cd, Pb in foliage and edible parts of 28 species grown in 10 kg pots of soil from two long term sludge farms. This report is the widest published comparison of metal uptake by crops from metal rich soils.

Differences in elemental concentrations in plants may result in: differential ion uptake by roots, soil-plant interactions and differences in root distribution in the soil with depth.

Factors affecting microelement translocation: After a microelement enters the root cells, its translocation to shoots is controlled by metal and plant characteristics. Root cell sap contains high levels of organic acids and amino acids which can chelate many metals. Membrane surfaces and proteins contain functional groups which

Table 5. Effect of soil temperature and time after sludge application on metal concentration in radish leaves (Morella, Chaney, Decker, and Sheaffer, unpublished; Sheaffer et al., 1981).

Sludge Rate mt/ha	First Crop Year				Sixth Crop Year			
	Soil pH	Metals in Radish Leaves			Soil pH	Metals in Radish Leaves		
		Zn	Cu	Cd		Zn	Cu	Cd
		mg/kg dry				mg/kg dry		
				Ambient (22 C)				
0	5.3	64e*	9.8ef	1.0a	6.7	35e	5.1e	0.8ab
56	6.0	295de	16.6de	1.1a	6.5	95cd	7.6cd	0.6b
112	6.2	663bc	36.1b	1.8a	6.1	152b	9.0c	1.0a
				Heated (35 C)				
0	5.6	86e	6.8f	2.1a	6.7	27e	4.3e	0.5b
56	6.0	490cd	26.9c	2.8a	6.5	51e	6.9d	0.5b
112	5.9	1070a	59.9a	2.6a	6.3	108cd	10.7b	0.8ab

*Means followed by the same letter in the same column are not significantly different from each other ($p = 0.05$).

can chelate some metals. Thus, a metal can be trapped in the roots if chelates formed in the root cells sap can not be transported into the xylem. Xylem is the system of non-living tubes in plants in which water and nutrients are translocated from roots to shoots. Most metals reaching the xylem are pumped into it by specialized cells. These cells, and chelates formed in the root cytoplasm, control whether a plant translocates a metal. Jarvis et al. (1976) noted wide variation among species in percent of absorbed Cd which was translocated to shoots; species ranged from 10-to-90% translocated. Crops often called Cd "accumulators" translocated a high percentage of absorbed Cd.

Generally, Zn, Cd, Mn, B, Se, and Mo are easily translocated because they are weakly chelated. Copper, Ni, and Co are more strongly chelated; a much smaller portion of the absorbed Cu is translocated to shoots than of Zn. Pb, Cr, and Hg are so strongly held in the root cells that very little is translocated to the shoots of crop plants. Research has characterized chelation of Fe, Ni, Cu, Co, Zn, and Cd in xylem sap, but only Fe citrate has been unequivocally identified (Tiffin, 1977; Foy et al., 1978; White et al., 1981; Cataldo et al., 1981). Amino acids control translocation of Ni and Cu in crop plants (Tiffin, 1977; Cataldo et al., 1978). Citrate probably chelates Zn and Cd in xylem sap (White et al., 1981; Chino and Baba, 1981), although Cataldo et al., (1981) concluded that plant-absorbed Cd appeared in non-citrate complexes.

Transport to storage organs: Many crops form storage or reproductive organs (edible roots or tubers; fruits; seed) which are used as food or feed rather than the whole plant shoot. Crops differ widely in botanical type of storage organ formed, and in translocation of microelements into the organ as it forms. The stored fat, protein, and starch come from sugars and amino acids via phloem from foliar photosynthesis. Some species have close control on composition of their storage organs (maize; beans; potatoes; fruits), while storage organs of other crops are readily increased in microelements when the leaves are increased (wheat, oat, rice, soybeans; root crops) (CAST, 1980). Interactions among metals may occur not only during root uptake, and translocation to shoots, but also during transport to edible tissues (Chaney et al., 1976). Although vegetable crop transplants grown in media containing sludge compost may have increased Cd levels, Cd did not increase in the edible portions of mature crops (Sterret et al., 1983).

A further source of difference among crops can be expressed as a result of food processing. When many grains are processed into "refined" flour products, the starchy endosperm is separated from the mineral and fiber rich bran. Metals in rice, wheat, and corn refined products are substantially lower than in whole grain products, although there is increasing consumption of whole grain foods over more refined products (Hinesly et al., 1979b; Chino, 1981; Kitagishi and Obata, 1981). However, oat groats contain the bulk of metals in oat grain

(Kirleis et al., 1981), and soybean cotyledons and normal soy protein products are as high in Cd as the whole grain (Braude et al., 1980).

Cultivar Differences in Metal Uptake

The potential for breeding crop cultivars with reduced metal concentrations or greater metal tolerance was noted in the 1973 Work Group 8 report. Some very promising results were reported by Hinesly et al. (1978; 1982) for differences in Cd and Zn uptake among maize inbreds and maize hybrids. Bache et al. (1981) found smaller differences in Cd in foliage or grain of maize cultivars adapted to New York. These results indicate that Cd-excluder cultivars are available for use on dedicated sludge farms. Some of the early misunderstandings about crop uptake of Cd resulted from these appreciable differences in Cd uptake among maize cultivars used by several researchers (CAST, 1980).

Soybeans, however, differ much less than corn in relative Cd and Zn uptake and tolerance (Boggess et al., 1978; White et al., 1979a). Although lettuce cultivars differ in Cd and Pb uptake, the range is small (CAST, 1980). Whether these smaller differences result from the smaller genetic base of adapted soybean and lettuce cultivars, or other differences between these crops is unknown.

Harris et al. (1981) recently reported on studies of differences in metal uptake among six potato cultivars (3 early, 3 main crop) when they were grown on an old sludge farm. The soil (pH 6.6) contained 1811 mg Zn/kg, 19.6 mg Cd/kg, 752 mg Cu/kg; these metal concentrations exceed median levels in sludges. Yet the washed, unpeeled potato tubers contained only normal Cd levels (mean = 0.28 mg/kg dry), and did not differ significantly among cultivars.

Wheat cultivars also differ in Cd transport to grain. Substantial differences were found among types of wheat on soils containing background Cd levels (Meyer et al., 1982). Durum types were much higher in grain Cd (140 mg/kg) than soft red spring, soft red winter, hard red spring, or white wheats (44 mg/kg). Durum wheats are a different species (tetraploid) than other wheats (hexaploid). Grain Cd was not significantly correlated with soil Cd, although low soil pH gave slightly higher grain Cd for the non-durum samples.

Andersson and Pettersson (1981) reported a small range for grain Cd among wheat cultivars grown in Sweden. Hinesly (T. D. Hinesly, Dept. Agronomy, Univ. Illinois, personal communication) examined Cd in grain of nine soft red winter wheat cultivars grown on pH 5.5 sludge-amended strip mine soils (34 mg Cd/kg). Cultivars did not vary significantly in grain Cd (with a range of 2.4-to 3.4 mg/kg and a mean of 3.0 mg/kg).

Role of Sludge-borne Metal Sorption Capacity

A number of alternative explanations have been suggested during the last decade for the apparently different effects of sludge-applied Cd on crop uptake of Cd. These include: 1) that repeated additions are different than single additions, also stated as annual additions of sludge Cd cause different Cd uptake than cumulative additions; 2) that sludge-borne Cd has either some or no persistent effect on crop Cd; and 3) that Cd salts, high Cd sludges and low Cd sludges have unequal effects on crop Cd. These are important differences since they would affect the regulatory strategy needed to protect humans from excessive dietary Cd (Ryan et al., 1982). Recent re-analyses of the data on which these different explanations are based have led to an underlying mechanism or model which addresses these difficult questions.

At the 1980 meeting of the W-124 Regional Research Committee, Corey introduced the concept that sludge is richer than soils not only in Cd, but also in Cd-specific sorption sites. These specific sorption sites sorb Cd on the order of 10^6 times more strongly than they bind Ca (Corey et al., 1981). He reasoned that, as sludge application rate increases, Cd activity will initially increase in the sludge-soil mixtures. The increase should be nearly linear at low sludge rates, as the soil Cd-specific sorption sites become progressively saturated. At still higher sludge rates, the soil Cd-sorption sites become saturated, and equilibrium Cd activity is now controlled by the ratio of Cd in sludge to Cd-sorption sites in the sludge. Futher increase in sludge Cd application would produce little or no further increase in Cd activity. Soil pH would affect Cd activity regardless of the source of the Cd-sorption sites. These concepts are discussed in Corey et al. (1981), and are strongly supported by field results with sewage sludge on several Wisconsin soils (Fujii, 1983).

The implications of this model include 1) Cd is adsorbed (hence labile), rather than precipitated as an inorganic compound in soils; 2) Cd activity in soil-sludge mixtures will be affected by sludge Cd concentration; 3) the concentration of Cd in crops may be a linear function of sludge-applied Cd only at low sludge additions, and 4) relative crop availability of an increment in soil Cd will depend on the history of sludge application for the soil. The CAST (1980) report concluded that these were indeed the characteristics of sludge amended soils.

Street et al. (1977) found that activity of Cd in soils was below that controlled by the least soluble inorganic Cd compounds for nearly all soils. Thus, they concluded that sorption controlled Cd activity. Further, Lloyd et al. (1981) reported that the Cd in soils of long-term sludge farms was nearly 100% labile.

Sludge applied Cd, as well as other Cd sources, remain available for plant uptake for many years in acidic soils. Studies of crops grown

on long-term sludge farms indicate crops can absorb increased amounts of Cd for at least as long as 20 years after the last sludge application (Chaney and Hornick, in CAST, 1980; Mahler and Ryan, 1982; Kirkham, 1975; Chaney et al., 1977; LaConde et al., 1979; Richardson, 1980; Chumbley and Unwin, 1982; Davis and Carlton-Smith, 1980). Cd is also persistent from phosphate fertilizers (Williams and David, 1973), from mine waste contaminated soils (Davies and Ginnever, 1979; Takijima and Katsumi, 1973), and in very old naturally Cd enriched soils (geochemical anomalies) such as the at least 50,000 year old California soils described by Burau (1980) and Lund et al. (1981), and the old New York peat bog described by Cannon (1955) and Cannon and Anderson (1971).

The annual vs cumulative effect of Cd may now be explained by the work of Mahler and Ryan (1982) who found that historic sludge applications caused a persistent increase in Cd sorption capacity, hence decrease in Cd uptake by crops. The slope of Cd concentration in chard or corn leaves vs newly added sludge or salt Cd was strongly reduced in previously sludged soil compared to the control non-sludged soils. It appears that organic (Soon, 1981) and inorganic (Singh, 1981) materials in sludge can provide the specific Cd sorption sites. This information indicates that the "time bomb" (Beckett and Davis, 1979) effect predicted to occur after sludge organic matter is stabilized in soil has no basis.

Further, Dowdy et al. (1981) deduced a new method to evaluate the separate contribution of annual and cumulative applied Cd. Since annual Cd is a part of the true cumulative Cd, these dependent variables are co-variant; by using Cd applied up to the year before the present crop (lag cumulative Cd) they could use multiple regression analysis. Their study showed that cumulative Cd dominated over the effect of annual Cd on Cd uptake to corn ear leaf or silage. Note that the presence of organic and inorganic (salts, NH_4^+) ligands in freshly sludged soils can contribute to the apparent "annual" Cd effect.

Sludge Cd concentration has a strong effect on potential Cd levels in plants grown in acid soils many years after the sludge has been applied (Chaney et al., 1982). Using orthogonal contrasts, they found that, in addition to linear Cd rate and soil pH, quadratic Cd rate also significantly affected Cd uptake by lettuce. The curves for lettuce Cd vs sludge-applied Cd were not linear, but had much lower slopes after the first sludge Cd increment. This effect was clear for sludges with 13 mg Cd/kg or 200 mg Cd/kg, but was especially important for the lower Cd sludge. Using slopes for plant Cd vs soil Cd from experiments with high Cd sludges, Cd salts, or Cd-salt amended sludges should no longer be acceptable in estimating risk from recommended practices with low Cd sludges.

Many scientists have argued that farmers should accept only low Cd sludges because the cumulative applied sludge benefit is inversely related to sludge Cd concentration. The above research indicates that equal Cd applied in low or high Cd concentration sludges will cause

unequal effects on Cd in crops. In many studies, experimental results were confounded by the factors discussed. In order to apply higher Cd rates, high Cd sludges were needed. Usually, experiments showing high Cd uptake used high Cd rates, high Cd concentration sludges, and high Cd:Zn ratio sludges.

Many experiments using low Cd, low Cd/Zn sludges have found low Cd in plants (Dowdy and Larson, 1975; Dowdy et al., 1978; Chaney et al., 1978a, 1982; Schauer et al., 1980; Harris et al., 1981; Giordano et al., 1979; Zwarich and Mills, 1982; deVries and Tiller, 1978; Sommers, 1980; Webber et al., 1981a; Chang et al., 1982; Lutrick et al., 1982; Sheaffer et al., 1981; Anderson and Nilsson, 1976; Gestring and Jarrell, 1982).

The effect of sludge metal concentration is also seen for other elements. Sludge Cu concentration greatly affected Cu uptake and phytotoxicity from equal Cu applications (hence unequal sludge applications) (Gupta and Haeni 1981). This finding supports earlier indications that potential phytotoxicity of sludge Zn, Cu, and Ni is related not only to metal application rate, but also to sludge metal concentration (Webber et al., 1981b; Marks et al., 1980). As expected, sludge-applied Zn, Cu, and Ni persist in their plant availability, at least in acid soils (Berrow and Burridge, 1981; Chaney et al., 1982; Marks et al., 1980; Williams, 1980).

Metal Interactions Affecting Cd Level in Crops

Besides phytotoxic interactions, interactions affecting Cd concentration in edible plant tissues are also very important. In 1973, Chaney suggested that limitation of Cd:Zn ratio in sludges could limit crop Cd to acceptable small increases or Zn toxicity would result. He initially based this on the parallel increase of Cd and Zn in crops as sludge rate increases or soil pH declines, and on already known interactions of Cd and Zn in animals.

For vegetable crops, this interaction now appears to satisfy the claims of Chaney (1973; 1980). Chaney and Lloyd (unpublished) amended Sassafras sl with Cd and Zn sulfates and added $CaCO_3$ as needed to maintain equal soil pH. Romaine lettuce was grown for 6 weeks, and analyzed (Table 6). Lettuce yield was substantially reduced at 400-to-500 mg Zn/kg in leaf tissue. At intermediate Zn levels, added Zn reduced crop Cd concentration, and crop Zn concentration was reduced by added Cd. However, the Cd concentration in healthy edible young lettuce leaves (those grown after transplanting) was constrained to about 4 mg/kg by either 1) Zn phytotoxicity, or 2) remaining within the recommended 1.0% Cd:Zn ratio (1.5% maximum on private cropland).

Related experiments were conducted by Hortenstine (1980) with lettuce and radish and Webber (1980) with lettuce, with similar results, although soil pH varied with treatments. Often small additions of Zn

Table 6. Soluble salts of Cd and Zn added to Sassfras sl and resultant Cd and Zn contents in lettuce grown in soil at pH 5.8 (Chaney and Lloyd, unpublished).*

Added Cd (mg/kg)	Added Soil Zn (mg/kg)					
	0	30	60	90	120	150
	mg Cd/kg dry young leaves					
0	1.15	0.60	1.16	1.16†	0.37†	1.07†
0.75	8.14	5.68	4.21	4.55	4.63†	4.95†
1.5	18.5	7.75	7.60	9.41	7.88†	8.69†
3.0	25.2	15.2	15.0	19.3	17.2	21.4†
	mg Zn/kg dry total shoots					
0	30	165	262	401†	508†	615†
0.75	31	154	240	312	425†	842†
1.5	31	128	205	320	478†	672†
3.0	37	102	185	293	406	522†

* Means of 3 replications.
† Denotes phytotoxicity, yield reduced 35% or more from maximum.

increase Cd movement to shoots, while higher Zn reduced foliar Cd (Haghiri, 1974). Chaney et al. (1976) confirmed Haghiri's findings with soybean foliage, but found that Cd movement to soybean grain was strongly inhibited by increased Zn. Grain Cd:leaf Cd ratio fell from over 1 to less than 0.1 as soil Zn increased. Unfortunately, monocots do not appear to follow this pattern (oats - Chaney and White, unpublished; wheat - Bingham et al., 1979; rice - Bingham et al., 1980). However, movement of Cd to grain of monocots is not very great under present EPA (1979a) regulations. The greater tolerance of monocots to Zn in acidic soils could theoretically allow small grains with 2-5 mg/kg Cd; however, dicots in rotation would indicate the pH mismanagement by phytotoxicity and lead to limestone application.

Although the above research was conducted with metal salts, or metal salt-amended sewage sludge, the parallel increase of Cd and Zn is apparent in several research studies in which low metal sludges were applied to acid soils. For foliar Cd to exceed a few mg/kg, plant Zn approaches phytotoxic levels, (Chaney et al., 1978a, 1978b, 1982; Zwarich and Mills, 1982; Gestring and Jarrell, 1982; Dowdy and Larson, 1975). The Cd:Zn ratio limitation was meant to add additional protection under the conditions where soil pH was poorly managed. The EPA (1979a) regulations adequately restrict metals in vegetable crops in soil at pH 6.5 or above.

METAL ACCUMULATIONS BY ANIMALS AND THEIR EFFECTS

Pathways for Transfer of Metal to the Food-chain

Sludges are applied to land in many fashions. These management options allow substantially different amounts of sludge-borne metals to enter the food-chain, by different routes. High exposure can come through ingestion of sludge-covered forage by livestock, while mixing sludge into the soil uses reactions in soils and properties of plants to largely prevent exposure.

Sludge adherence to forage crops: Spray-applied liquid sludges form a thin film covering crop foliage and, if not removed immediately by rainfall or irrigation, dry and adhere strongly (Chaney and Lloyd, 1979; Jones et al., 1979; Bertrand et al., 1981). Sludge adherence is directly related to solids content of the applied sludges, and to sludge application rate. Growth of crop biomass dilutes the sludge percentage in harvested forage.

Digested sewage sludge has adhered to all crops studied, although crops do differ in relative adherence due to differences in structure and growth habits. Adherence is easily characterized because levels of many elements (Cu, Pb, Cr, Fe) in/on sludge-contaminated forages are

much greater than levels ordinarily possible by uptake-translocation by plants. Many reports on uptake of microelements from surface-applied sludges have presumed uptake when sludge adherence fully explains the observations.

Another possible route of entry to food-chain is through farm equipment. Dalgarno and Mills (1975) reported that baled hay was enriched in Cu above levels found in/on the sampled forage crop. They presumed that the baling equipment lifted high Cu swine manure from the soil surface (or possibly the thatch layer). This route is poorly demonstrated. Silage can also contain soil, and the soil can interfere with microelement availability (Lamand, 1979).

When increased levels of microelements in forage indicate sludge adherence, all constituents present in the sludge contaminate the forage. Not only microelements, but also macroelements, pathogens (Brown et al., 1980) and toxic organic compounds (Fitzgerald, 1978) are increased.

Ingestion of sludge-amended soil or sludge from the soil surface: Grazing animals consume soil as part of the normal grazing process. This was noted initially during study of the excessive wear of teeth of sheep and cattle. Using soil Ti (not appreciably absorbed by plants) as an indicator/tracer on soil in feces, Healy et al. (1974) found that wet weather and excessive stocking rates caused forages to be trampled into the soil and increase soil adherence and ingestion. Although soil is normally only 1-2% of sheep's diet, soil could reach 24% in worst cases.

Mayland et al. (1977) reported that cattle grazing dryland-grown crested wheat grass consumed considerable quantities of soil. Because the cattle ate the plants complete with soil-laden roots, the ingested diet contained 20% soil. Fries (1982) recently reviewed soil ingestion by dairy cattle. Under good management, soil seldom exceeds 3% of dry diet.

Sewage sludge or composted sludge can be ingested from the soil surface. Decker et al. (1980) estimated that cattle grazing tall fescue pastures fertilized with sludge compost consumed 3.2% (1977) and 1.0% (1978) compost (of dry diet) based on fecal analysis. Compost did not adhere to the plant but lay on the soil surface.

Soil/sludge ingestion can also expose humans to applied microelements. Some children and adults deliberately consume soil in a practice called "pica." If the soil is rich in Pb (over 500-1000 mg/kg), individuals may absorb excessive Pb (Wedeen et al., 1978; Shellshear, et al., 1975; NRC, 1980a). Children also inadvertantly ingest soil and dust due to hand-to-mouth play activities, and mouthing of toys, etc. (Sayre et al., 1974; Baker et al., 1977; Roels et al., 1980; Hammond et al., 1980).

Soil or sludge ingestion can be an important process allowing entry of sludge-borne metals into the food-chain particularly when the element is poorly absorbed by plants (soil level exceeds plant level). For some elements (Zn, Cd, Mn, Se), plant levels often exceed soil levels and plant uptake is the more important route. However, soil ingestion is a potential route for allowing excessive exposure to Pb, Fe, Cu, F, As, Hg, Co, Mo and Se if these elements are present at high levels in surface soil (e.g. Thornton and Abrahams, 1981). Further, soil ingestion can interfere with availability to animals of microelements in feedstuffs (Suttle et al., 1975).

Research has shown that applying sludge to recently mowed or grazed fields, and waiting to allow the crop to grow and dilute the adhering sludge, can keep the sludge to 3-5% of dry forage. These practices, coupled with use of sludges low in metals and toxic organics, protects the health of livestock and safety of animal food products. Subsurface injection of sludge removes this food-chain pathway, and is clearly the best management practice for sludge application to forages/pastures.

Plant uptake of metals from soils: Some elements are easily absorbed and translocated to food-chain plant tissues (e.g. Zn, Cd, Mn, Mo, Se, B), while others are not readily transported. The poorly translocated elements are strongly precipitated or bound in the soil, or are retained in plant fibrous roots, and are not translocated to plant foliage in injurious amounts even when soils are greatly enriched (e.g. Fe, Pb, Hg, Al, Ti, Cr^{3+}, Ag, Au, Sn, Si, Zr). Even though an element may be easily or relatively easily absorbed and translocated to foliage, phytotoxicity may limit plant levels of the element to levels safe for animals (e.g. Zn, Cu, Ni, Mn, As, B).

These concepts were developed over many years by many scientists. Several reviews are very helpful (Underwood, 1977; Allaway, 1977; Reid and Horvath, 1980; Ammerman et al., 1977; Chaney, 1980, 1983b). Chaney introduced the term "Soil-Plant Barrier" to describe these concepts when considering sludge-soil-plant-animal relationships of microelements. Historically, plant absorbed residues of Se and Mo have caused localized poisoning of livestock. Direct ingestion of soil or sludge circumvents the protections of the soil-plant barrier. Many elements are so insoluble or non-toxic that animal health is not affected even if ingested soil or waste contains the element (e.g. Cr^{3+}, Zr, Ti, Si, Al, Sn). However, direct ingestion of soil or sludge rich in some elements (e.g. Cu, F, Zn, Pb, Fe, As, Co, Hg) allows risk to livestock when risk would have been insignificant if the sludge were mixed with the surface soil (0-15 cm).

Metal Interactions in Animals

During the last decade, many studies were conducted to determine whether direct ingestion of sludges or high Cu manures would allow microelements to injure livestock. Researchers fed levels of waste-borne Cu, Zn, etc., high enough to severely poison animals based on research in which single metal salts were added to diets.

Evaluation of the potential risk to animals of sludge-applied microelements is very complex. Animal species differ in tolerance. Tolerance is influenced by age, with younger animals generally more sensitive. Total and relative element concentration is affected by crop species and cultivar, plant age, soil pH, soil organic matter, soil temperature, and other factors. Sludges differ in levels of elements and ratios among elements which interact in the animal. Particular toxic elements interact with other elements in the diet, often inversely. Because these interactions are often the basis for observed toxicity, interactions are very important in assessing risk of sludge-applied microelements.

Chaney (1983b), has provided a detailed discussion of exposure, interactions which affect risk, etc., for the principal metals in municipal sewage sludge (see summary in Table 7). Interactions are especially important in allowing or preventing toxicity of ingested Cu, Mo, Fe, Zn and Cd.

"Domestic" low metal sewage sludge contains a mixture of potentially toxic elements. When sludge is ingested, increased dietary Zn is balanced by normal or high levels of Cu and Fe. Increased Cu is balanced by Fe and Zn which reduce Cu toxicity in ruminants. Although many thought high Cu swine manure would poison Cu-sensitive sheep based on "toxicology" studies, manure Cu has not poisoned sheep in feeding or grazing studies (Bremner, 1981; Poole, 1981). Sewage sludge ingestion has not even caused Cu accumulation in liver of cattle fed sludge (Kienholz, 1980; Edds et al., 1980; Smith et al., 1979; Decker et al., 1980) unless sludge Cu was high (Baxter et al., 1980; 1982). Low Cu, high Fe sludge allowed Fe-induced Cu-deficiency in cattle grazing tall fescue pastures (Decker et al., 1980).

Accumulation by Animal Organs

Animals have homeostasis mechanisms for most elements (Underwood, 1977). High dietary Zn, Cu, Mn, Ni and Fe increase intestinal and liver concentrations of the element, but have little effect on other tissues. When feeding of the enriched diet ceases, tissue levels return to normal. Lead can build up in bone and kidney in forms slowly released. Inorganic Hg can build up in liver, and organic Hg in fat deposits and nerve tissue in forms slowly released. Fluorine can build up in bone as

Table 7. Maximum tolerable levels of dietary minerals for domestic livestock in comparison with levels in forages.

Element	"Soil-Plant Barrier"	Level in Plant Foliage*		Maximum Levels Chronically Tolerated†			
		Normal	Phytotoxic	Cattle	Sheep	Swine	Chicken
		-- mg/kg dry foliage --		---------- mg/kg dry diet ----------			
As, inorganic	Yes	0.01-1	3-10	50	50	50	50
B	Yes	7-75	75	150	(150)	(150)	(150)
Cd‡	Fails	0.1-1	5-700	0.5	0.5	0.5	0.5
Cr^{3+}, oxides	Yes	0.1-1	20	(3000)	(3000)	(3000)	3000
Co	Fail?	0.01-0.3	25-100	10	10	10	10
Cu	Yes	3-20	25-40	100	25	250	300
F	Yes?	1-5	--	40	60	150	200
Fe	Yes	30-300	--	1000	500	3000	1000
Mn	?	15-150	400-2000	1000	1000	400	2000
Mo	Fails	0.1-3.0	100	10	10	20	100
Ni	Yes	0.1-5	50-100	50	(50)	(100)	(300)
Pb‡	Yes	2-5	--	30	30	30	30
Se	Fails	0.1-2	100	(2)	(2)	2	2
V	Yes?	0.1-1	10	50	50	(10)	10
Zn	Yes	15-150	500-1500	500	300	1000	1000

* Based on literature summarized in Chaney (1983b).

† Based on NRC (1980b). Continuous long-term feeding of minerals at the maximum tolerable levels may cause adverse effects. Levels in parentheses were derived by interspecific extrapolation by NRC.

‡ Maximum levels tolerated based on human food residue consideration.

fluoroapitite, a form slowly released. Cadmium can accumulate in liver and kidney in the form of metallothionein; Cd has the longest biological half-life of the elements of concern in sludge (about 20 years in humans).

Potential Effects of Sludge-borne Cadmium in the Food-chain

In 1973, relatively little was known about Cd in food and risk to humans. The occurrence of Cd-disease in Japan had just been announced. Toxicologists still thought hypertension could be caused by Cd. Food specialists thought daily Cd intake was nearly at the safe maximum level based on meager knowledge of Cd toxicology and poor analytical data.

Research on Cd risk has been especially intense because of these fears, and from this work, a much more complete understanding of Cd risk has developed.

Cd exposure and health effects: Cadmium is an unusual and difficult case for evaluation of risk to the food chain. In contrast to other elements, Cd has a long biological half-life in humans--generally considered 20 years. Absorbed Cd is bound to a low molecular weight protein to form metallothionein which is accumulated and retained in the kidney for a long period. High metallothionein-Cd in the kidney can lead to adverse health effects in the kidney.

Over a lifetime, chronic food chain Cd exposure can cause different health problems than those experienced from acute exposure. Long-lived animals (e.g., humans) are at greater risk of this chronic exposure health effect than are short-lived animals (wildlife; domestic animals). Accumulation of Cd in organ meats (liver, kidney) was the basis for suggesting a low dietary Cd tolerance in domestic animals rather than a direct health effect to the animals (NRC, 1980b).

The potential risk of excess soil Cd to humans has been clearly documented. Adverse health effects resulted from prolonged consumption of foods grown locally on Cd enriched soils (Tsuchiya, 1978; Friberg et al., 1974; Hammons et al., 1978; Kobayashi, 1978; Nogawa, 1978). A large number of Japanese farmers suffered Cd health effects after long-term ingestion of Cd-enriched rice grown in paddies polluted by Zn- and Pb-mining wastes or Zn-, Pb-, and Cu-smelter emissions in at least 7 different areas of Japan (Kobayashi, 1978; Takijima and Katsumi, 1973; Shigematsu et al., 1979; Kjellstrom et al., 1977; Kojima et al., 1979; Saito et al., 1977; Nogawa, 1978; Nogawa and Ishizaki, 1979; Nogawa et al., 1975; 1978; 1980; 1981). The farm families consumed their own rice due to cultural factors and tax policy. Rice Cd concentration and number of years exposure were both strongly related to the incidence rate of Cd health effects. A smelter enriched area in Belgium may have

caused Cd-induced renal disease (Roels et al., 1981a) although route for exposure and increased kidney Cd have not yet been demonstrated.

The name "itai-itai" disease (translated as ouch-ouch disease) came from expressions of pain by women suffering repeated bone fractures due to Cd-induced osteomalacia. Although the osteomalacia brought attention to this environmental Cd disease, severe osteomalacia does not frequently result in humans ingesting excessive Cd; osteomalacia appears to result in Cd-exposed individuals only when dietary Ca is low. Renal proximal tubular dysfunction (Fanconi syndrome) is the first health effect of excessive chronic Cd exposure. The renal disease had high incidence in areas where Cd exposure was increased, and showed a dose-response relationship with Cd exposure (expressed as "Cd level in rice-times-years ingested"). All individuals with advanced itai-itai disease had severe proteinuria, a characteristic of the kidney disease. Renal disease subsequently proceeded to mild osteomalacia in some European workers who ceased exposure when the kidney disease was identified (Kazantzis, 1979). However, this aspect of Cd disease is poorly understood. Sub-clinical osteomalacia is found in many of the Japanese farmers who experience renal disease but not "itai-itai" disease (Mukawa et al., 1980).

Renal tubular dysfunction (Fanconi syndrome) resulting from Cd ingestion is quite different from classic kidney failure. Fanconi syndrome seldom proceeds to kidney failure requiring dialysis. Kjellstrom (1978) indicated that Fanconi syndrome (low molecular weight proteinuria, glucosuria, aminoaciduria, phosphaturia) is the first Cd health effect; if Cd-exposure (rate-times-duration) is increased, kidney stones and osteomalacia/osteoporosis may result. Kjellstrom et al. (1979) found greater mortality (shorter life span) in Cd exposed workers. Nogawa et al. (1981) found significantly greater mortality risk in farmers with proteinuria. Neither hypertension nor prostate cancer incidence are increased even when proteinuria is severe (Friberg et al., 1974; Doyle, 1977; Hammons et al., 1978; Tsuchiya, 1978; Ryan et al., 1982; Commission of the European Communities (CEC), 1978; Kjellstrom and Nordberg, 1978; Kjellstrom et al., 1979; Lauwreys et al., 1980; Nogawa, 1978; Nogawa et al., 1981; Shigematsu et al., 1979). Although Kopp et al. (1982) claimed that hypertension in Cd-fed rats indicated risk to humans at current dietary exposure levels, the dose rates they fed expressed on the basis of 70 kg adults (700-1400 µg/day) are high enough to have caused Fanconi syndrome. Although laboratory studies with rats and other animals have shown that anemia, enteropathy, and teratogenesis (due to Cd-induced Zn or Cu deficiency in the fetus) can result from ingested Cd, these are very unlikely with practical diets or with crops fertilized with low Cd sludges.

A number of researchers and groups have attempted to clarify the dose-effect and dose-response relationships for Cd (CEC, 1978; Friberg et al., 1974; Kjellstrom and Nordberg, 1978; Ryan et al., 1982; Tsuchiya, 1978). The first sign of renal tubular dysfunction (increased excretion of B_2-microglobulin or retinol binding protein, a specific

proteinuria characteristic of Cd injury) is generally agreed to occur at about 200 mg Cd/kg wet kidney cortex. Some research indicates that the critical kidney cortex Cd level may be as high as 300 mg/kg (Roels et al., 1981b; Ellis et al., 1981), but 200 mg/kg is the level generally accepted for use in risk analysis. Individuals may vary about this population estimate.

Kjellstrom and Nordberg (1978) developed a sophisticated multicompartmental dose-effect model for Cd metabolism in humans: "This present model predicted that a daily intake corresponding to 440 µg at age 50 would give 200 mg Cd/kg of (wet) kidney cortex at age 45-50." These results were obtained by assuming a high, constant Cd concentration per unit calories, and that calorie (hence Cd) ingestion varied with age in the manner of the average diet of the Swedish population. The "best fit" equation calculated a 4.8% lifetime average absorption of dietary Cd, 440 µg Cd/day at age 50, and a 12 year biological half-life for Cd to achieve the 200 mg Cd/kg wet kidney cortex at age 45-50.

Other researchers have used different ways to express Cd-exposure information, thus complicating interpretation of results from these many sources. In the U.S., the Food and Drug Administration (FDA) has measured food Cd concentrations and average Cd ingestion (FDA, 1977). Food consumption was based on USDA's 1965 dietary intake survey but adjusted to a "balanced" diet; FDA, USDA, and EPA agreed to use a food consumption model based on teenage males (highest food consuming group) in a pesticide residue survey program. Thus, for the same food supply, a mean food Cd ingestion of 39 µg/day from FDA corresponds to about 23 µg/day intake at age 50 in Kjellstrom and Nordberg's (1978) model. Their model reflected 3430 kcal/day for Swedish teenage males vs. 2045 kcal/day for 50-year-old Swedish individuals (Fig. 4.32 and 4.34 in Friberg et al., 1974). Thus, the critical 440 µg Cd/day ingestion rate for 50-year-old individuals in Kjellstrom and Nordberg's (1978) model corresponds to approximately 738 µg Cd/day ingestion in U.S. teenage male diets. The present average exposure is only 5.2% of the critical exposure (23 vs 440 or 39 vs 738).

Chaney (1983b) and Ryan et al. (1982) discuss difficulties in interpreting dose-response relationships for dietary Cd. Individuals vary widely in self-selected diet and dietary Cd (Yost et al., 1980), in Cd absorption rate (Flanagan et al., 1978; McLellan et al., 1978), and in sensitivity to absorbed Cd. These phenomena are generally assumed to vary in a log-normal fashion in a population. Kjellstrom (1978) extended the 440 µg/day model "critical" level to a population by arbitrarily using a geometric standard deviation of 2.35 based on studies of Cd in autopsy tissues (see Ryan et al., 1982 for details). However, Kjellstrom's (1978) model would require greater than 100% absorption of dietary Cd by the most sensitive individuals (see Figure 1 in Chaney, 1980). The highest Cd absorption rate observed for humans is 25% reported by Flanagan et al. (1978) for a woman with mild anemia; Fe stress strongly increases Cd absorption. Several researchers (Chaney,

1980; Ryan et al., 1982) argued that it was unreasonable to extrapolate the 440 µg Cd/day "Average Human" model result to an assumed maximum sensitivity group with greater lifetime percent absorption of Cd than ever observed in humans. Further, individuals are unlikely to be in this greatest risk group for their whole lifetime.

Thorough analysis of the Cd injury of Japanese farmers conducted during the last decade appears now to support only a dietary threshold type dose-response curve. For any individual to be injured, sufficient Cd must have accumulated in the kidney to induce proteinuria. Although individuals vary in retention of dietary Cd, and in kidney Cd associated with initial low-molecular-weight proteinuria (LMW-proteinuria), much Cd must be consumed to reach this condition. Food Cd must exceed some level before the most sensitive individuals experience LMW-proteinuria.

The Japanese data indicate that, for individuals over 70 years of age, incidence of LMW-proteinuria was not significantly increased until rice Cd exceeded 0.40 mg/kg. For individuals 50-60 years of age, incidence of LMW-proteinuria was not significantly increased until rice Cd exceeded 0.60 mg/kg. Individuals are presumed to have eaten 300 g rice/day; thus acceptable exposure was 300 x 0.4 = 120 µg Cd/day for persons over 70 years old, and 300 x 0.6 = 180 µg Cd/day for 50-60 year old's. The standard Japanese is considered to weigh 53 kg (Kjellstrom, 1978), while the standard Westerner is considered to weigh 70 kg. Adjusting for this difference, the first significant observation of LMW-proteinuria in U.S. citizens should require 238 µg Cd/day for a 50 year model, or 158 µg Cd/day for a 70 year model. These analyses disregarded Cd exposure from smoking which would have increased daily intake above that from rice. Thus, the log-normal extrapolation appears inappropriate compared to the threshold model (Ryan et al., 1982).

Ryan et al. (1982) concluded that a 200 µg/day (150 µg/day after protecting smokers) threshold model (based on average lifetime daily Cd intake) of individuals was more appropriate for dose-response considerations, as did the CEC (1978) workgroup. This value corresponds to about 14.1% lifetime Cd absorption rate for the most sensitive individuals [4.8(440/150)].

Cd bioavailability: Cadmium absorption by animals is strongly influenced by other dietary factors (Fox, 1976, 1979; Fox et al., 1978, 1979; Jacobs et al., 1978a, 1978b, 1983; Flanagan et al., 1978; Welch et al., 1978; Welch and House, 1980; Kostial et al., 1979; Cousins, 1979; Kobayashi, 1978; Washko and Cousins, 1977). Iron status of the animal appears to be the most important control of percent absorption of Cd and many individuals ingest low Fe during at least part of their life. Zinc status of the animal and dietary Zn level is the next most important factor, followed by dietary Ca. Protein and fiber in the diet and age of animal also influence Cd retention. These factors should allow a greater percent absorption rate for women than men. Women as a group showed greater Cd absorption (Flanagan et al., 1978), and women's

kidney Cd exceeds men's in autopsy kidney studies, as did women's susceptibility to excessive dietary Cd in Japan.

Dietary interactions can thus influence bioavailability of Cd. Leafy and root vegetables which are enriched in Cd may also be a good dietary supply of Zn, Fe, and Ca. Leafy vegetables have been shown to provide bioavailable Fe and Zn (Welch et al., 1977, 1978; Van Campen and Welch, 1980; Wien et al., 1975). Chaney (1980) suggested that leafy and root vegetables grown on soils enriched in Cd from being fertilized by low Cd, low Cd:Zn sewage sludges comprise a separate risk scenario. In this case, consuming sufficient food Cd to pose a risk to individuals susceptible because of poor nutrition would result in increased dietary Fe, Zn, and Ca, thereby shifting the individuals to a less susceptible population group who do not suffer from poor nutrition.

Feeding studies have been conducted with sludge and with crops grown on sludge-fertilized soil. Ingestion of sludge Cd has been evaluated in ruminant and monogastric animals with most work done with cattle. When sludges with high Cd and high Cd:Zn were fed, kidney Cd was significantly increased (Kienholz, 1980; Baxter et al., 1980; 1982; Hansen and Hinesly, 1979; Hinesly et al., 1976, 1979a; Edds et al., 1980; Fitzgerald, 1980; Lisk et al., 1982; Johnson et al., 1981). However, when sludges with lower Cd and low Cd:Zn were fed, kidney Cd was not significantly increased (Decker et al., 1980; Kienholz et al., 1979; Kienholz, 1980; Baxter et al., 1980; Bertrand et al., 1981; Edds et al., 1980; Smith et al., 1979). Sludge Cd was less bioavailable to swine than equal Cd added as $CdCl_2$ (Osuna et al., 1979; Edds et al., 1980). Food products of animals are unchanged in Cd except for liver and kidney (e.g., Sharma et al., 1979). Kienholz (1980) noted that dietary interactions could avoid even this impact of sludge Cd. Thus, risk analysis for ingested sludge Cd requires evaluation of several factors other than dietary Cd concentration.

As noted above, the Cd-injured Japanese farmers were deficient in several nutrients (Ca, Zn). Rice was the source of nearly all the increased dietary Cd. Although the soils contained both Cd and Zn (at Cd:Zn about 1:100) from mining wastes, the rice was increased in Cd but not Zn. Rice grain grown under flooded culture contained only 20-25 mg Zn/kg while rice Cd rose from normal levels of 0.05 mg/kg to over 2 mg/kg (page 237 in Tsuchiya, 1978); polished rice is very low in Ca and Zn.

Similarly, risk analyses for ingestion of Cd in foods grown on Cd-enriched soils requires careful evaluation of factors other than Cd. Far too little research has been conducted to characterize bioavailability of food Cd. Further, very little of the completed research conforms with the experimental designs which Fox et al. (1978, 1979) and Fox (1976) indicated were needed to allow interpretation of data. Dietary Cd level should correspond to the range of nutritional relevance to humans. Cadmium enriched foods should be fed in the state ordinarily ingested by humans (e.g., fresh leafy vegetables).

Nutritional status of the experimental diet should be adequate for all known essential factors or varied as part of the experiment. The feeding period should be of sufficient length to allow nutritional status of animals to be under control of experimental diet for the bulk of the experimental period. Several animal species should be studied. Bioavailability of Cd in a food or a sludge-grown food can only be determined experimentally.

The bioavailability of Cd may also be influenced by the chemical species of Cd present in crops. Several studies have reported the existence of an inducible protein which can bind Cd in plants (Bartolf et al., 1980; Weigel and Jager, 1980; Wagner and Trotter, 1982; Casterline and Barnett, 1982). Unfortunately, these studies used phytotoxic levels of Cd in their research. It is possible that high levels of Zn or other metals might also affect the chemical species of Cd in edible plant portions. Thus, research is needed on potential shifts in chemical species of Cd in sludge fertilized crops and its effect on Cd bioavailability, if any.

Cd in tobacco: Tobacco is an especially high risk crop in terms of potential for Cd effects on humans. Among all crops studied to date, tobacco accumulates more Cd per unit soil Cd than any other (Chaney et al., 1978a; Maclean, 1976; Davis and Carlton-Smith, 1980). In many areas, tobacco is normally grown on strongly acid soils to prevent crop loss from root diseases. This soil pH management leads to maximum Cd uptake under normal crop production conditions. In contrast, most other crops are best grown at pH 6.5 to 7. Tobacco is normally high in Cd compared to leaves of other crop plants, and high leaf Cd levels in some production areas are being studied (Frank et al., 1977; Westcott and Spincer, 1974). When tobacco is grown on acidic sewage sludge-amended soils, crop Cd levels can be increased from 1 to as high as 44 mg Cd/kg in dry leaves (Chaney et al., 1978a) with only 1 mg/kg soil Cd. Gutenmann et al. (1982) applied 19 kg sludge Cd/ha and found tobacco Cd increased from 3.2 to 67.4 mg/kg.

Cadmium in tobacco is an important source of Cd for humans. Individuals who smoke one pack of cigarettes per day have about 50% higher Cd in kidney cortex than non-smokers (Lewis et al., 1972; Elinder et al., 1976). About 15% (5-25%) of cigarette Cd enters the mainstream smoke (Szadkowski et al., 1969; Menden et al., 1972; Westcott and Spincer, 1974). Filters can remove much of this Cd and reduce Cd exposure of smokers (Westcott and Spincer, 1974; Franzke et al., 1977). Based on the potential of sludge-applied Cd to increase risk of chronic kidney disease in smokers if sludge were applied to tobacco cropland, EPA (1979a) regulated and discouraged this practice. Some tobacco growing states have even prohibited sludge applications on land suited for tobacco production.

Models for limits on Cd application: Several food crops are of especial importance to evaluating Cd-risk for humans. While grains supply much Cd to individuals in the general population (Braude et al., 1975; Jelinek and Braude, 1978; Ryan et al., 1982), individuals are extremely unlikely to grow a significant portion of their food grains. Rather, individuals are likely to grow leafy and root vegetables, legume vegetables, garden fruits, and potatoes. If the Cd:Zn ratio of an acidic Cd-enriched garden soil is high, edible crop tissues of leafy, root, and legume vegetables, garden fruits, and potatoes can be greatly increased in Cd concentration with no injury to the crop and provide excessive bioavailable Cd. If the Cd:Zn ratio of an acidic Cd-enriched garden soil is low (< 0.01), these crops are not greatly increased in Cd when Zn phytotoxicity limits crop yield, (see data for lettuce in Table 5), and bioavailable Cd would be only slightly increased. The difference in risk from low Cd:Zn and high Cd:Zn gardens is due to: 1) Zn-phytotoxicity at low pH in the low Cd:Zn garden causing the gardener to add limestone which reduces crop Cd or have little yield (hence, reduced exposure), 2) interactions between Cd and Zn in plant uptake and translocation to edible plant tissues (Chaney et al., 1976; Chaney and Hornick, 1978); and 3) interactions in the diet which influence Cd bioavailability.

It is much more difficult to evaluate Cd bioavailability from foods grown on waste-amended soils than from Cd-amended purified diets. Freeze-dried lettuce and chard grown on acidic soils amended with domestic sludge were fed at a high percentage of diet to mice or guinea pigs (Chaney et al., 1978b, 1978c). Although dietary Cd was increased by up to 5-fold by lettuce or chard grown on acidic, domestic sludge-amended soil, kidney Cd was not increased. Bertrand et al. (1980) found similar results for forage. In other studies with high Cd and/or higher Cd:Zn sludges, feeding sludge-grown crops has caused increased kidney Cd (Chaney et al., 1978b; Miller and Boswell, 1979; Heffron et al., 1980; Lisk et al., 1982; Telford et al., 1982; Williams et al., 1978; Hinesly et al., 1976, 1979).

A recent paper by Boyd et al. (1982) shows another difficulty of assessing risk of sludge-applied Cd. They applied 8.6 kg Cd/ha in one 112 mt/ha application (violated 5 kg Cd/ha and 0.5 kg Cd/ha/yr limits of EPA, 1979a). However, snapbean and squash showed small increases in tissue Cd, and Cd in rat kideys was equal to that in the control, or lower (Table 8). Sludge-fertilized cabbage and redbeets were six times higher in Cd than control crops, yet kidney Cd was increased less than two-fold. Thus, sludge Cd had little effect on kidney Cd when vegetable crops were grown, even though Cd regulations (annual and cumulative Cd for acid soils) and recommendations (sludge Cd $\leq$ 25 mg/g and Cd/Zn $\leq$ 0.015) were exceeded. Two other cabbage feeding studies failed to meet "Environmental Toxicology" standards (cabbage grown on 1.2 m deep pile of high Cd sludge, and fed at 30% of diet to rats or 45% of diets to sheep); reported kidney Cd increases are thus irrelevant to sludge risk analysis (Haschek et al., 1979; Babish et al., 1979). Clearly, more sludge-soil-plant-animal studies are needed to characterize the

Table 8. Cadmium and zinc in edible crop tissue and kidney of rats fed four control and sludge-fertilized crops for 84 days (Boyd et al. 1982).

Crop	Crop Analyses				Kidney Analyses			
	Cd, mg/kg dry		Zn, mg/kg dry		Cd, mg/kg dry		Zn, mg/kg dry	
	C*	S	C	S	C	S	C	S
Beans	0.1	0.1	30	34	3.4	1.1	89	90
Squash	0.2	0.3	15	54	4.9	3.8	100	86
Cabbage	0.2	1.3	17	93	7.1	9.7	80	103
Redbeets	0.5	3.2	27	125	5.4	12.6	83	95

* C and S denote control and sludged, respectively. Applied 112 dry mt/ha of sludge containing 77 mg Cd/kg and 3477 mg Zn/kg, applying 8.6 kg Cd/ha.

bioavailability of Cd in crops grown on waste-amended soils. It seems very likely that factors besides background soil pH, and annual and cumulative Cd application will eventually have to be considered in setting allowed Cd loadings on cropland (EPA, 1979a; Chaney et al., 1980a).

Much of the potential risk from Cd in waste-amended soils has now come under regulation in the U.S., although these regulations do not have to be enforced until September, 1984. The highest risk case, application of sludges to gardens as fertilizers or soil conditioners, has not yet been regulated (Comptroller General, 1978; Chaney et al., 1980b). Further, pretreatment of Cd-bearing industrial wastes, segregation of waste streams, and avoidance of Cd use for non-critical applications offer great opportunity to avoid all Cd health effects (Dage et al., 1979; Gurnham et al., 1979).

In the process of developing Federal regulations for land application of sewage sludge (EPA, 1979a), EPA prepared a "worst practical case" scenario relating sludge-applied soil Cd to potential for kidney dysfunction to protect the higher risk sensitive individuals (EPA, 1979b). The worst case which may occur appears to be the "acid garden" case. Although grain crops can absorb Cd, individuals in the U.S. do not grow their own food grain on acidic, Cd-enriched soils. Similarly, consumption of liver and kidney enriched in Cd from sludge utilization is a minor source of dietary Cd. This approach protects individuals with soils least able to keep Cd from reaching their foods.

Thus, the "acid garden" scenario was used. It presumed that 1) the soil contains the fully allowed Cd application, 5 kg/ha; 2) the soil is continuously acidic, about pH 5.5; 3) the gardener obtains 50% of his annual supply of garden vegetables from the acidic, sludge-amended soil, including potatoes, leafy, root, and legume vegetables, and garden fruits; 4) the individual eats these amounts of garden vegetables for 50 years from the acidic sludge-amended garden; and 5) the individual is part of the sensitive-to-cadmium portion of the population. Further, EPA used a normal Cd ingestion value of 39 µg/day, a value selected from the range of annual estimates (28-51 µg/day during 1970-1979) of daily dietary intake of Cd obtained in FDA's Total Diet Studies. The 39 µg Cd/day for the teenage diet model was subtracted from the 71 µg/day WHO-FAO (1972) provisional daily Cd ingestion to obtain a maximum allowed increase due to sludge use. The WHO-FAO tolerable intake is for population average adult exposure. Fecal analysis indicates that U. S. adults ingest about 20 µg/day (Kowal et al., 1979), and adjustment of the teenage diet to adult diet gives a similar estimate of 23 µg/day. Thus, this part of the EPA (1979a) rule offered another safety factor. Based on new dietary consumption research findings, FDA has recently revised the methods used to estimate consumption of contaminants in US diets (Pennington, 1983). Individual foods are analyzed (to avoid detection limit questions on food composites). Thus, dietary Cd ingestion can be estimated for 8 age-sex groups rather than only teenage males.

It appears now that several assumptions of EPA's "acid garden" scenario may provide excessive protection at least for "Domestic" sludges. First, individuals who grow 50% of their garden vegetables have such a large time and work investment in their gardens that they learn about the effects of acid soils on yield of vegetable crops, and carefully manage soil pH at 6.5 to 7. Second, presuming that a low Cd, low Cd:Zn ratio sludge supplied the soil Cd, and that soil pH declines slowly due to fertilizer use, phytotoxicity in sensitive crops will cause a "50% gardener" to learn about soil pH management and interrupt the necessary 50 year "acid garden" exposure. Third, vegetables supply microelements which counteract Zn, Fe, and Ca deficiencies; these deficiencies are the identified basis for sensitive individuals. Thus, consumption of the vegetables which comprise the minimal Cd risk to sensitive individuals in the EPA scenario (71 µg/day) may push them out of the sensitive population. Recall that increased Cd in "domestic" sludge-grown chard and lettuce did not increase kidney Cd (Chaney et al., 1978b, 1978c). Alternatively, the presence of higher levels of other elements may have simply changed Cd speciation in foods to a form with lower bioavailability. In their discussion of Cd dose-response models, Ryan et al., (1982) concluded that U.S. sensitive individuals are protected at the 150 µg Cd/day level of exposure (150-20 = 130 µg/day vs 71-39 = 32 µg/day). Alternatively, the FAO-WHO 71 µg Cd/day value for adult diets may need to be adjusted from adult diet to teenage diet if the 39 µg/day result is to be used (=119 µg/day, with 119-39 = 80 µg/day increase tolerable in teenage diet model).

Further, the slope of the crop Cd vs soil Cd relationship varies with sludge Cd concentration (Chaney et al., 1982); low Cd sludges give appreciably lower slope (hence lower potential exposure at 5 kg Cd/ha regulated cumulative maximum Cd). The slopes for crop uptake from low Cd sludges are much lower than the slopes used in risk estimates (see EPA, 1979b; Ryan et al., 1982). High rates of low Cd sludges had little or no effect on Cd in major garden vegetable crops (Dowdy and Larson, 1975; Dowdy et al., 1978; Chaney et al., 1982; Harris et al., 1981).

Some scientists have expressed a concern that the safety factor for dietary Cd is small (e.g. Jelinek and Braude, 1978). The review presented here, and Ryan et al. (1982) indicate that the safety factor is larger than previously believed. We believe that this new information should be used to demonstrate the existence of a larger safety factor when low Cd domestic sludges are utilized, rather than to support higher limits for Cd application on cropland. It remains clear that high Cd, high Cd/Zn sludges do not enjoy the improved safety factor shown for "Domestic" sludge. The larger safety factor should reduce public fear of recommended quality sludges, since it provides greater protection of sensitive individuals who select diets with higher than average Cd content. As a result of these newer understandings, and in response to concerns expressed by the food industry, the regulatory and advisory agencies developed a policy statement on utilization of low Cd sewage sludge on cropland for production of fruits and vegetables (EPA-FDA-USDA, 1981).

In summary, the "Soil-Plant Barrier" does not protect the food chain from excessive Cd. Unregulated application of Cd-bearing wastes can cause health effects in humans. Cadmium is not easily kept out of food crops; conversion of treated land to gardens is a worst case scenario upon which regulations to limit Cd applications were based (EPA, 1979a). Recent research on gardens polluted with Cd by mining wastes or smelter emissions support the view that gardens can provide much Cd in locally grown foods to the family maintaining the garden for many years (Davies and Ginnever, 1979; Chaney et al., 1980, unpublished). Some aspects of the Cd-waste-soil-plant-animal food chain are not well established, and research is needed to avoid unnecessarily restrictive limits in the regulations. Recent research indicates a substantial difference in potential risk from Cd in high Cd sludges vs. low Cd, low Cd/Zn sludges, and that for the "Domestic" sludge case, the safety factor for dietary Cd is much larger than previously believed. The larger safety factor conclusion rests on: 1) Sensitive individuals are protected at 150-200 µg Cd/day; 2) adult U.S. diets contain about 20 µg Cd/day; 3) Zn phytotoxicity can sharply limit maximum achievable crop Cd levels; 4) Cd in crops grown on soils amended with low Cd, low Cd/Zn sludges has very low bioavailablity; and 5) crop uptake of Cd per unit soil Cd is much lower for low Cd sludges.

Potential Effects of Other Sludge-borne Elements on the Food-chain

A few elements besides Cd have been of high interest or concern regarding risks from sludge utilization, and are considered briefly here. Other elements are not discussed here because they are low in sludges (As) or high levels are tolerated by animals (Cr^{3+}, Al, Sn). The discussion of "Soil-Plant Barrier" above indicates that only a few elements merit the concerns expressed in 1973. Further, the special low risk case resulting from utilization of recommended quality sludges (Chaney, 1980) substantially alters the potential for risk from many elements in sludge.

Cobalt: Excessive dietary Co can cause toxicity in ruminant animals, while monogastric animals are more tolerant (NRC, 1980b). Diets with over 10 mg Co/kg (as soluble salts) have injured cattle and sheep.

Unfortunately, plants tolerate higher levels of foliar Co than ruminants tolerate in their diets. Most plants show symptoms or substantially reduced yield at 50-100 mg Co/kg in foliage (Chaney, 1983b). However, no feeding studies have been conducted with plant-absorbed Co. Sludges are normally low in Co (Sterrett and Lester, 1981), and there is little reason for concern about sludge Co unless the sludge is high in Co.

Copper: Phytotoxicity due to excessive soil Cu occurs in most plants at 25-40 mg Cu/kg (Walsh et al., 1972). Copper toxicity in sensitive sheep and cattle occurs at 25-100 mg Cu/kg in the diet (as soluble Cu salts). However, increased dietary Zn, Cd, Fe, or Mo would counteract added Cu (Bremner, 1979, 1981).

Crops have to be grown on acidic soils quite high in sludge to contain 25-40 mg Cu/kg. For the "Domestic" sludge case, these plants would have increased Zn which would prevent any toxicity due to the small possible increase in plant Cu. Similarly, ingested sludge and high Cu manures have caused no Cu toxicity problems even though dietary Cu reached 50 mg/kg (Bremner, 1981; Poole, 1981).

Fluorine: Fluorine is normally low in plants because F is precipitated and/or sorbed strongly in soils. Sludges are normally low in F; 80% of sludges surveyed contained lower than 450 mg/kg. However, a few sludges are very high in F, with the highest reported 3.35% F (Chaney, 1983b). Davis (1980) found that this high F sludge could temporarily cause increased F uptake by ryegrass. However, others have not found increased F in sludge fertilized plants.

Direct ingestion of sludge circumvents the "Soil-Plant Barrier". Kienholz et al. (1979) found increased bone F in cattle fed sludge with normal F concentration.

Iron: Iron is normally restricted to 40-300 mg/kg dry foliage by plant and soil properties. Fertilizing with sludges, even sludges very rich in Fe, does not increase Fe appreciably above normal levels. Soil on contaminated forages often provides more total Fe than forage crops.

Although animals tolerate increased ferric iron, 1000 mg/kg ferrous Fe added to forage diets reduced liver Cu in sheep and cattle. On low Cu diets, Fe can induce Cu-deficiency (NRC, 1980b). Fe-toxicity was observed in cattle grazing tall fescue pastures sprayed with a digested sludge containing 11% Fe (Decker et al., 1980). Sludges lower in Fe and sludge composts (ferric Fe) did not cause Fe-toxicity. When digested sludges are surface-applied to pastures, sludge Fe concentration should be analyzed. Sludges over about 4% Fe have caused Fe-induced Cu-deficiency; these observations were probably related to the crop, method of grazing, and low sludge Cu concentration (Chaney, 1983b).

Lead: Sludge applied Pb has been found to <u>not</u> increase plant Pb unless sludge Pb concentration is extremely high (Chaney, 1983a, b). Often, sludge application reduced plant Pb even though soil Pb increased. This has been observed in all soil types, and for all crops studied -- grains, fruits, tuber and edible root, and even leafy vegetables. The reduced uptake may result from sludge-increased soil Pb

sorption capacity, or phosphate which reduced Pb translocation within plants.

Although sludge-applied Pb, mixed in the soil, appears to comprise no potential risk in any scenarios presently used by regulatory agencies, new information is now available on risk to children ingesting moderately high levels of Pb. Classic Pb poisoning of children can cause an encephalitis which often leads to severe retardation. Ingestion of Pb-paint contaminated soils has caused excessive blood Pb in children with pica for soil. Recently, a neurobehavioral impairment (lower IO and school achievement, and problem classroom behavior) was found to increase with Pb concentration in deciduous teeth (Needleman et al., 1979, 1980; Winneke et al., 1981). Thus, higher than normal Pb ingestion may be undesirable in children. Because some sludges may be surface applied as lawn fertilizers or mulches, their Pb concentration and its bioavailability may be important. Sludge Fe, P, Ca, and organic matter would reduce Pb absorption. It appears that "Domestic" sludges comprise no risk to livestock, or to human adults through plant uptake or soil ingestion (Chaney, 1983b). Research is needed to determine whether ingestion of small amounts of sludges low in Pb adds to Pb risk of infants and children.

Mercury: Mercury is noted for the food-chain poisoning resulting from methyl mercury accumulation by aquatic organisms. However, the terrestrial ecosystem is quite different than the aquatic. Mercury is bound so strongly in soil and in plant fibrous roots that uptake by plants is not the limiting factor. Some volatilization of Hg vapor can occur when soils are rich in Hg.

Sludge Hg appears to be important in two sludge utilization cases. First, direct ingestion of sludge rich in Hg will increase liver Hg (Kienholz, 1980). Second, if sludges are used in the composts for mushroom production, mushrooms can exceed 0.5 mg Hg/kg fresh weight, the level regulated in fish. Domsch et al. (1976) found excessive Hg in mushrooms grown on refuse-sludge compost. Frank et al. (1974) found even some horse manure is high enough in Hg to cause excessive mushroom Hg. Mushroom species vary widely in Hg and methyl-Hg accumulation (Stijve and Roschnik, 1974).

In research on production of mushrooms on composts including composted municipal refuse (Schisler and Grable, 1976), the excessive Hg accumulation observed by Domsch et al. (1976) was confirmed. Further, the Hg availability to mushrooms from refuse compost appears to be greater than that in traditional mushroom composts (L. C. Schisler, Pennsylvania State University, personal communication). Thus, sludge compost should not be used in mushroom production until research shows that Hg levels in the mushrooms will not exceed levels regulated by FDA.

Molybdenum: Excessive plant Mo can poison ruminant animals, but not monogastric animals under an enriched garden scenario. The ingested molybdate is converted to tetrathio-molybdate which can form an insoluble and unavailable compound with Cu. This causes a severe Mo-induced Cu-deficiency, a practical problem in sheep and cattle in many nations. Plants tolerate quite high levels of Mo without causing symptoms or substantial yield reduction. If forages contain 5-10 mg Mo/kg, such that the Cu:Mo ratio falls below two, induced Cu deficiency can occur. In contrast to metal cations, anionic molybdate uptake by plants is increased as soil pH is raised. Most excessive plant Mo problems occur on calcareous soils.

Sewage sludges are very seldom high in Mo. Williams and Gogna (1981) reported Mo uptake by white clover and ryegrass grown on alkaline soil amended with sludges low (5 mg/kg) or high (103 mg/kg) in Mo, or with equal amount of molybdate. The high Mo sludge caused high Mo uptake by white clover; sludge Mo availability was approximately equal to that of molybdate. The low Mo sludge caused no appreciable change in plant Mo. Davis (1981b) reported Mo and Cu uptake by forage crops on soil from a sludge farm (515 mg Cu/kg and 6.0 mg Mo/kg). Clovers and brassica forages had unfavorable Cu:Mo ratios. He also reported the expected soil pH effect on Mo uptake from soils amended with a Mo-rich sludge. Based on these findings, sludges to be applied to neutral pH or calcareous soils for forage production should be analyzed for Mo to protect ruminant animals.

Nickel: Nickel represents elements for which phytotoxicity completely protects the food chain. Ruminants tolerate at least 50 mg Ni/kg (NRC, 1980b), while forage crops show visual symptoms of toxicity at 50-100 mg/kg. Cattle suffered no effects from forage diets with 250 mg Ni/kg as $NiCO_3$, while soluble Ni salts caused the animals to reduce food consumption. Alexander et al. (1979) reported no toxicity or Ni bioaccumulation in meadow voles fed soybean grain containing 30 mg Ni/kg, about the highest level possible without severe yield reduction.

Selenium: Although plant absorbed Se has poisoned livestock in several areas of the Western U.S. (Kubota and Allaway, 1972), sludge-borne Se has not been shown to cause excessive plant or animal Se (Kienholz, 1980). Fly ash can cause excessive Se uptake if poorly disposed, or correct Se deficiency in livestock if used wisely (Gutenmann et al., 1979). Annual monitoring of sludge for Se can identify sludges with potentially excessive Se. Uptake of Se is favored on alkaline soils, or soils where Se-accumulator crops leave residues of seleno-amino acids.

Zinc: Excessive dietary Zn (300-1000 mg/kg) can injure ruminant animals by inducing Cu-deficiency (NRC, 1980b). Sheep with low dietary Cu were injured when dietary Zn exceeded 300 mg/g (Campbell and Mills,

1979). Smelter Zn pollution injures horses, which are sensitive to excess. Gunson et al. (1982) reported erosion of cartilage in joints, a manifestation of severe Cu-deficiency (Bremner and Campbell, 1980). However, Zn causes appreciable and/or visible injury to plants when foliar Zn is 500 mg/kg (see plant uptake section). Sludge risk scenarios include strongly acid soils. Although high Zn sludges may be equivalent to Zn salts in risk scenarios, "Domestic" sludges apply Cu and other nutrients. Acid soil-grown forage crops would not be low in Cu or Fe; further, forage crops are not Zn-accumulators. Thus, livestock are protected from excess crop Zn problems when the Zn is supplied by "Domestic" sludges. Sludge ingestion studies have found no Zn toxicity problems, likely due to the rich Cu and Fe supply in sludge. It is conceivable that ingestion of high Zn, low Cu sludges could lead to induced Cu deficiency in ruminants.

The "acid garden" scenario for Cd could be applied to Zn. Because leafy vegetables comprise such a small part of human diets, and because Zn is much lower in seeds, fruits, and edible roots than leaves, gardeners would be protected. Actually, many humans consume low dietary Zn, and increased crop Zn could be beneficial.

REGULATORY STRATEGIES FOR METAL CONTROL

Metals to be Regulated

The USEPA (EPA, 1979a) has only regulated the application of cadmium to land in sewage sludge and other wastes. Other metals which have been identified as potentially phytotoxic or food-chain contaminants (Cd, Pb, Hg, As, Se, Zn, Cu, Ni, Mo) have not been regulated at the Federal level. Many states have considered Pb, Zn, Cu, Ni, Hg, and even Cr. Cadmium has received regulatory attention because of the relative ease with which it is taken up by crops, and because of the relatively small difference between current dietary Cd intake and the presumed safe intake level (Ryan et al., 1982).

Although there appears to be little present justification for general regulation of metals other than Zn, Cu, Ni and Pb, Chaney (1983b) lists several which should require attention in the form of further research or sludge monitoring programs. These are sludge borne metals which can escape the "Soil-Plant Barrier" (Se, Mo) or are added to soils in industrial wastes (Be, Co), or may cause animal health problems when sludge is ingested (above metals plus F and As). Regulations may be required in some instances to prevent direct ingestion of sludge or soil which has received sludge. Many states regulate Zn, Cu, and Ni application to reduce risk of phytotoxicity.

Metal Levels in Sludges

One regulatory approach to the control of potential problems from metals in sludges is to restrict the land application of sludges which have metal concentrations in excess of prescribed levels. This approach is relatively easy to monitor, and has the advantage of encouraging sludge pretreatment by individual treatment plants. The levels of individual metals that constitute a "safe" sludge is, and has been, a topic of considerable debate. Chaney (1973) suggested that a sludge that would be appropriate for land application should not exceed the following metal concentrations:

	Chaney (1973)	Chaney & Giordano (1977)
	------ mg/kg dry weight ------	
Zn	2000	2500
Pb	1000	1000
Cu	800	1000
Ni	100	200
B	100	--
Hg	15	10
Cd	0.5% of Zn	25 (1.5% of Cd)

Chaney felt that these levels were reasonably attainable. Note that, at 0.5% of the 2000 mg/kg maximum Zn concentration, the maximum Cd content would be 10 mg/kg. Chaney and Giordano (1977) subsequently suggested maximum levels of metals in domestic sludge (shown above).

Doty et al. (1978) presented the following recommended maximum concentrations for Pennsylvania sludges which they attribute to Baker

and Chaney (no reference given), and which are designed to allow safe annual sludge applications at a rate of 22.4 mt/ha:

	mg/kg dry weight
Cd	50
Cu	1000
Cr	1000
Pb	1000
Hg	10
Ni	200
Zn	2000

Although these are suggested maximum concentrations, Pennsylvania does not regulate metal levels in sludges. Japan regulates maximum sludge Cd, Hg, and As levels at 5, 2, and 50 mg/kg (Ishikawa and Dodo, 1982). The ranges of maximum allowable concentrations in European and Canadian sludges for As, Cd, Cr, Cu, Hg, Mo, Ni, Pb, Se and Zn are 10-75, 10-40, 200-1,200, 500-3,000, 5-25, 20-40, 100-500, 300-2,000, 14-40 and 1,850-10,000 mg/kg, respectively (M. D. Webber, Dept. of Environment, Burlington, Ontario, personal communication).

At the present time, metal levels in sludges are not regulated by USEPA except that sludges which contain < 2 mg Cd/kg dry weight are exempt from current regulations on pH control. Many states (e.g. PA, MD, VA, NY, CN) regulate maximum sludge metal concentrations.

Early in 1980, USEPA released in pre-proposal draft form rules that would constitute a comprehensive federal sludge management policy (Newkumet, 1980). The approach proposed by USEPA was to establish maximum concentrations of cadmium and lead in sludge for various uses. Sludge fertilizers for unrestricted use would have a fixed nitrogen to cadmium ratio which would limit increases in soil Cd over a 25-year period to 0.5-2.5 mg/kg. A restricted use category for sludge fertilizers (non-food chain uses) would allow a lower N:Cd ratio. Similar ratios of N:Pb would limit Pb accumulation in the top one centimeter of soil over a 25-year period to 250-1000 mg/kg. Sludge products to be used as soil conditioners for unrestricted use could contain 2.5-10 mg Cd/kg and 150-1000 mg Pb/kg if the material also had a Cd:Zn ratio $\leq$ 1.5% and contained at least 10% $CaCO_3$ equivalent by weight. Restricted soil conditioning sludge products could have 10-25 mg Cd/kg and 250-1000 mg Pb/kg.

These proposals have generated considerable discussion since they were first proposed. Although EPA has revised their draft in light of newer information, the agency has not proposed them for public comment. This approach offers significant incentives to treatment plants to improve the quality of their sludges, and opens the door to more widespread sale and distribution of sludge products. On the other hand,

there is some concern about the ability of plants in large cities to reduce metal levels to background levels to the point where they can meet the proposed concentrations. In addition, more recent information on the dietary intake of Cd (Ryan et al., 1982) and the risks associated with other metals should be evaluated before establishing final rules on metal concentrations in sludge products.

Annual and/or Cumulative Metal Loadings

The current regulatory approach in the United States and many other countries is to restrict the cumulative and annual application of metal to soil. In the U.S., Cd is the only metal that is currently regulated, but it is likely that the same approach would be appropriate for other potentially toxic metals which accumulate in soil. The W-124 (formerly NC-118) Regional Research Committee (EPA, 1980) reviewed the development of guidelines for metal loadings as follows:

1. Chumbley (1971) was one of the first to propose limiting metal accumulations in soil. He proposed a maximum accumulation of 560 kg/ha of Zn equivalent, where Zn equivalent = Zn + 2 Cu + 8 Ni. The coefficients of the equation reflect the relative phytotoxicities of the three metals.

2. Leeper (1972) suggested combining the Zn equivalent approach with CEC and proposed a maximum Zn equivalent accumulation of 5% of the CEC.

3. In 1974, USEPA published a draft technical bulletin which proposed to increase the allowable Zn equivalent to 10% of CEC and decrease the relative phytotoxicity of Ni with respect to Zn from 8 to 4. In addition, they proposed that the Cd:Zn ratio not exceed 1:100. The Cd:Zn ratio had been proposed by Chaney (1973) as a means of preventing excessive Cd uptake since Zn would become phytotoxic before Cd had accumulated to excessive levels.

4. In the same year, an NC-118 subcommittee met to review the proposed USEPA guidelines and suggested that: (1) the Zn equivalent approach implied more knowledge about the relative phytotoxicities of Zn, Cu and Ni than actually existed; (2) Cd and Pb additions to soil should also be controlled due to human health concerns; (3) the annual rate of Cd application should not exceed 2.24 kg/ha (2 pounds/acre) (based on field data from Wisconsin which showed slight increases in grain Cd at 4.48 kg/ha annual rate but not at 2.24 kg/ha); (4) the cumulative Cd application should not exceed 11.2 kg/ha. This was based on the notion that, after five years at the maximum annual application rate of 2.24 kg/ha, the accumulation would still be within the range encountered for natural mineral soils (Allaway, 1968). It was felt that, after five years, there would be further research data to give a more accurate value; (5)

allowable accumulations of Pb, Ni, Zn and Cu should reflect current knowledge of their relative phytotoxicities. Based on these criteria, NC-118 suggested the following maximum accumulations for medium textured soils in the North Central region of the U.S., assuming that soil pH would be $\geq$ 6.5:

	lbs/ac	kg/ha
Pb	1000	1120
Zn	500	560
Cu	250	280
Ni	100	112
Cd	10	11.2

These were intended to be lifetime accumulations.

5. In 1976, the NC-118 Committee, at the request of USDA, modified their guidelines to distinguish between fine, medium and coarse textured soils (Table 2). In addition, the allowable Ni addition was increased to the same level as Cu. These guidelines or regulations have been adopted by many state regulatory agencies.

6. In 1978, EPA reprinted the North Central Research Publication No. 235 which included the revised Ni levels (Knezek and Miller, 1978).

7. In 1979 (EPA, 1979a) USEPA published final, interim final and proposed regulations for land application of sewage sludges. Cd was the only metal addressed, and the section of the rules (interim final) that apply to metals can be summarized as follows:

   1) The annual Cd application will not exceed 0.5 kg/ha for tobacco, leafy vegetables or root crops grown for human consumption.

2) For other food-chain crops, the annual Cd application rate will not exceed:

	kg/ha
Present to June 30, 1984	2.0
July 1, 1984 to Dec. 31, 1986	1.25
After Jan. 1, 1987	0.5

3) The cumulative Cd application to soils used for the production of food-chain crops will not exceed those given in Table 9.

4) No maximum annual and cumulative Cd applications are specified, but the only food-chain crop produced is animal feed, the pH of the soil/sludge mixture is 6.5 wherever crops are grown, there is a facility plan which demonstrates how the animal feed will be used to preclude its ingestion by humans, and future property owners are notified that the land has received Cd applications and should not be used for food-chain crops. (This "dedicated site" approach will be discussed later).

The phased reduction of the annual Cd application rate of 2.0 kg/ha originally proposed by NC-118 in 1974 (actually 2.0 pounds/acre) was justified by USEPA as being necessary for the long-term protection of the nation's food-chain, and the phased-in reductions would provide the necessary lead time for treatment plants to develop their pretreatment programs. The CAST Report (1980) concluded that, while there was a clear positive relationship between the total amount of Cd applied and Cd uptake at a given pH, there were no clear-cut differences betwen annual versus cumulative additions. This follows from previous discussion in this paper which indicated that previous Cd additions did not rapidly revert to less available forms. There appears, then, to be little justification for required annual Cd application rates < 2.0 kg/ha as long as the cumulative provisions are followed. However, lower Cd sludges allow much greater sludge benefit to be applied at the desired cumulative Cd limit. British scientists conclude that the original Zn (equivalent) limitations were for extractable metals. Using extractable:total ratios for the EDTA and acetic acid extracts used in the UK, Williams (1982) indicated that British total soil metal restrictions would be similar to the NC-118/USDA values.

Soil pH and CEC

Soil pH: Chumbley (1971) suggested that soil pH should be > 6.5 to reduce phytotoxicity of Zn, Cu and Ni. In 1974, NC-118 (USEPA, 1980)

Table 9. Maximum cumulative Cd applications to soil (EPA, 1979a).

CEC (meq/100 g)	Maximum Cumulative Application (kg/ha)		
	Background ≥pH 6.5	Background* pH <6.5	Background pH <6.5
<5	5	5	5
5-15	10	10	5
>15	20	20	5

*Soil/sludge pH is 6.5 whenever food-chain crops are grown.

proposed that soil pH should be 6.5 at the time of sludge application and maintained at $\geq$ 6.2 after application. The 1976 CAST Report recommended that soil pH be maintained at $\geq$ 6.5.

In the current USEPA regulations (EPA, 1979a), the soil/sludge mixture pH at the time of application must be $\geq$ 6.5 wherever food-chain crops are grown. The regulations provide for lower cumulative Cd limits (5 kg/ha) for soils whose background pH is < 6.5 and the soil/sludge pH will not be $\geq$ 6.5 at some time after sludge application. This provision is particularly important for those soils, primarily in the unglaciated regions of the eastern, southeastern and northwestern U.S., that have high lime requirements to reach pH 6.5.

Soil pH is the factor which has most consistently been shown (CAST, 1980) to affect Cd uptake of crops at a given level of Cd in the soil. Soil acidity is regularly neutralized by liming in normal farming operations, and pH 6.5 is close to the optimum for most crops and for those soil processes which lead to improved crop growth. A few crops are grown in strongly acidic soils (< 5.5) to prevent plant diseases (some potato and tobacco cultivars).

CEC: The current USEPA regulations (EPA, 1979a) on cumulative Cd additions distinguish between these CEC classes: < 5, 5-15 and > 15 meq/100 g soil. The W-124 Regional Research Committee (EPA, 1980) and the 1980 CAST Committee (many of the W-124 members also served on the CAST Committee) reviewed the existing data on the effects of CEC on Cd uptake by plants and concluded that it was difficult to establish a direct link between CEC and metal uptake because most studies were confounded by pH and soil organic matter content. The Committees felt, however, that CEC reflected a number of the soil properties which result in metal retention and decreased plant uptake. In addition, CEC is a routinely measured soil parameter and familiar to agriculturists. Several problems were identified, however, with the use of CEC to regulate Cd accumulation by crops: (1) on calcareous soils, Cd uptake appears to be insensitive to changes in CEC, (2) effects of CEC on uptake of other metals (e.g. Ni, Cu, Zn) may not hold for Cd, and (3) the CEC relationship may be most valid for soils with a mixture of organic matter, aluminosilicate clay minerals and Fe, Al and Mn oxides, but may not hold for soils dominated by any one of these constituents.

Future research should concentrate on the effects of soil minerals, organic matter and pH on metal uptake and regulations should be refined to reflect advances, as they occur, in our knowledge of the soil-metal system.

Sludge Characteristics

The regulatory approach used in the U.S. and many other countries is to restrict metal applications to soil regardless of sludge characteristics and sludge metal bioavailability. But recent work (Corey et al., 1981) would indicate that the bioavailability of a metal can vary depending on the chemistry of the sludge, and that, in some cases, the metal-binding capacity of the sludge may be as, or more, important than soil characteristics in determining metal uptake by crops.

Research should focus on developing extraction procedures, adsorption isotherms or other methods of determining the metal retention capacity of different sludges which might, in the future, provide the means to regulate sludge-metal additions to soil on the basis of sludge as well as soil characteristics.

Background Metal Levels in Soils

Extensive information on background levels of metals in uncontaminated soils has been developed in recent years (Allaway, 1968; Page, 1974; Baker and Chesnin 1976; Pierce et al., 1982; Logan and Miller, 1983). A survey of metal levels of uncontaminated soils in the U.S. is presently being conducted by the Soil Conservation Service (USDA-USEPA-USFDA cooperative study). These data will provide a basis for comparison with soil tests from sites which are known to have received, or are suspected of having received metal additions in the past. As land application of sludges and other wastes which contain metals increases, the need to monitor prior applications also increases, especially if there has been inadequate documentaion of previous metal additions. At the present time, most of the available data is on total metal content of soil, and research indicates (CAST, 1980) that total metal content of soil gives reasonable correlations with metal uptake by crops when other factors, such as pH, are constant. In the future, however, new soil test procedures may be sufficiently accurate to measure available metal levels in soils, and provide a better indication of metal uptake than total metal content of soil.

Food-chain Versus Non Food-chain Plants

The current USEPA regulations are designed to prevent excessive uptake of metals by food-chain crops. These include crops ingested directly by humans such as fruits, vegetables (and includes tobacco) and grains, as well as crops which are primarily fed through livestock. Land used to grow tobacco, leafy vegetables and root crops grown for human consumption (EPA, 1979a) can receive no more than 0.5 kg/ha/yr

compared to the phased reduction of the annual rate for other crops from 2 down to 0.5 kg/ha by 1987. The cumulative metal accumulations are the same for all food-chain crops, however, based on the acidic garden scenario. While there are no restrictions on the accumulation of metals in soil for growth of non food-chain species, there is an implicit understanding in the regulations that prior accumulations of metal while non food-chain species were grown would apply to food-chain crops if there was a planned land use change to crop production.

There does not appear to be any new research since the promulgation of the 1979 regulations that would indicate that they are ineffective in controlling excessive Cd uptake. Of greater concern, is the lack of monitoring of small users of sludge products (the home gardener) who is likely unaware of the metals content of the product he is using and of the health risks associated with excessive metal ingestion. This problem was considered by USEPA (Newkumet, 1980) in their proposed regulations for the sale and distribution of sludge products, and it appears that this is the only manageable approach to controlling the commercial dissemination of sludge-borne metals into the human environment.

FDA "Safe" Limits

An alternative method of controlling metal uptake by the food-chain is to establish maximum concentrations of metals in crops and livestock that would be "safe." Chaney (1973) in his Conference paper raised the possibility of establishing permissible Cd contents of foods in the marketplace. The difficulties of doing this, however, are many. A "safe" limit can only be established for a particular crop or food in terms of its contribution to the total diet, the transmission of the metal from the crop or food into humans, and the existing and maximum allowable metal burden in the diet (based on full toxicological understanding). The Food and Drug Administration (FDA) has not been able to establish "safe" limits in crops and foods even for a single metal, cadmium, which has received enormous attention in recent years (Ryan et al., 1982). FDA presently has limits on Hg in fish and Pb in canned milk and baby formulas.

Even if the dietary scenarios can be established, setting a metal concentration limit in crops is difficult. Crops differ greatly in their uptake of metals, translocation of the metals from roots to shoots, and in relative transport of metals into fruits, tubers, and grains (Chaney and Hornick, 1978; Chaney, 1983b). There are also significant differences among cultivars of the same crop species (CAST, 1980).

Using the "safe" limit approach would also entail extensive crop monitoring, and it is not clear at this time how a nationwide monitoring program would be run or enforced or could be afforded. Interpretation

of a crop monitoring program will require some knowledge of the background levels of metals in major crops. Presently, USDA-USEPA-FDA are conducting a nationwide survey of metals in crops grown on uncontaminated soils. The complete data will not be available for another year or so, but some of the preliminary results have been reported for wheat (Meyer et al., 1982). These results will be invaluable in determining the variability in metal content of crops, and may help to determine if the "safe" level approach is a viable regulatory alternative.

Intensive (dedicated site) Versus Extensive (low application rate) Strategies

The 1979 USEPA regulations provide for two approaches to land application of sewage sludges (EPA, 1979a). The first approach controls the annual and cumulative Cd additions to cropland and is designed to regulate sludge applications at rates which allow for efficient crop utilization of sludge nitrogen. This is usually < 15 dry mt/ha. This extensive approach will result in a slow accumulation of metals on a significant portion of agricultural land in the U.S. (Logan, 1983). As reviewed in this paper, these levels may cause no practical change in Cd in crops, at least for low Cd sludges (Chaney et al. 1983b). At crop utilization rates, 0.5-40% of the available cropland in a state may be required for land disposal of all POTW digested sludges (USDA, 1978). Critics of this approach will argue that it is potentially dangerous to allow even low level contamination of such a large percentage of our food-producing resource. On the other hand, increases in our knowledge of food-chain effects of specific metals will hopefully keep pace with the slow rate of metal addition to land with this approach. This conclusion should be tempered, however, by the realization that chronic levels of toxic metals in the human diet may take decades to exhibit recognizable disease symptoms.

The second approach proposed by USEPA (EPA, 1979a) involved higher applications of sludges to land at "closely controlled facilities." This would permit greater metal additions than in the previous approach, but management would control the disposition of food-chain crops and would monitor surface and groundwater to prevent their contamination. This approach is currently used, to various degrees, by many POTW's (Forster et al., 1981), although, in many cases, there is inadequate monitoring of crops, soil and water. This "dedicated site" approach is favored by those who feel that contamination under controlled conditions of a small part of our food-producing soil resource is less of a long-term risk than a strategy of more extensive landspreading (Logan, 1983).

Regardless of the relative merits of these two approaches, it is difficult in practice to define "dedicated site" management systems. Forster et al. (1981) surveyed Ohio POTW's and found that several had been spreading sludge on the same land for many years. In some cases

this was POTW-owned land, but in other instances neighboring farmland was used. There was little indication from the survey that the 1979 USEPA regulations for "dedicated sites" was being followed.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

George L. Braude, FDA, Washington, DC: Where risks are not amenable to precise definition (such as for cadmium), we should err on the side of caution. EPA just spent 33 million to buy a town and correct a mistake. 20 years from now, can we taxpayers be sure not to have to buy metal contaminated land - as new information is developed about plant uptake, toxicology, etc.?

Terry J. Logan: Firstly, there are significant differences between dioxin contamination, the toxic compound involved in your example, and heavy metals, such as cadmium. We know a lot more about the health effects of cadmium and its transmission through various pathways than we do about the toxic organics. Secondly, we are looking at chronic, and not acute, toxicity with cadmium when we consider health risks. And thirdly, the federal regulations on cadmium accumulation in the food chain are viewed by many as being conservative. I do not foresee any new information which could radically change our understanding of health risks from cadmium.

James O. Evans, USDA Forest Service, Washington, DC: Differences in the fate of metals in sludge when applied to land surface versus being incorporated in the soil itself have been well established by the speaker and others. Perhaps, our chief concern on a short-term basis should be with the metals in sludge on soil, whereas on a long-term basis our chief interest might be with the soil organic fraction and the soil itself, assuming that sludge assimilation eventually occurs. My question is, approximately how long might it be before this assimilation occurs?

Terry J. Logan: We have indicated in our paper that sludge metal-binding capacity may be important in determining metal bioavailability when sludges are land-applied. Although sufficient long-term field studies do not exist to answer your question, the available evidence indicates that there is no drastic change (increase or decrease) in bioavailability of metals applied to soil in sludges. This could mean that the sludge-metal solid phases are sufficiently stable in soil to prevent significant change in bioavailability, or that the bioavailabilities of sludge-metal forms are similar to metal forms in soil.

Frank M. D'Itri, Michigan State University: Little mention has been made with respect to the role of redox potential in determining the speciation of heavy metals in wastewater, digestor sludges and sludges applied to the land. For example, under reducing conditions, methylated tin compounds will undergo transmethylation reaction with inorganic mercury to produce methylmercury.

Terry J. Logan: There is always the possibility of forming various reduced species of metals in wastewater and sludges, particularly anaerobically digested sludges. However, the soil environment is almost always aerobic, especially in the long term, and reduced metal species are quickly oxidized once sludge is applied to soil.

Rufus Chaney: Normal sludges have not caused increased mercury or tin. Volatilization of sludge-mercury has not been observed.

Harvey Luce, University of Connecticut: Recent research indicates that certain soils have the capacity to oxidize Cr^{+3} to Cr^{+6}. Should this knowledge increase concern as to the potential of groundwater contamination by sludges that are high in Cr?

Rufus Chaney: There is a difference between chromium oxidation potential in artificial laboratory test systems, and finding oxidation in the environment. High Cr sludges may be found to eventually allow some Cr oxidation, but sewage sludges with several thousand ppm have not been found to cause increased chromate in groundwater.

Charles F. Jelinek, Food and Drug Administration: Cadmium is certainly the primary metal of concern, but we should not disregard lead. Because of the adverse effects of trace levels of lead on infants and young children, the lead levels in food eaten by infants and pregnant women need to be controlled. Even though lead is not taken up by plants through the root system, it can enter the food and feed chain if sludge is applied to the land by spraying. Because of these factors controls for lead as well as cadmium should be established.

Richard Thomas, National Coordinator, Innovative/Alternative Technology Program, EPA, Construction Grants Programs, Washington, DC: I want to make some observations as an oldtimer with 20 years in research who is now responsible for the alternative technology part of the federal grants program to improve publicly owned treatment works (POTW).

1. *Congress has shown increasing confidence in land treatment with each revision of law since 1972 and shows no inclination to change their thinking.*

2. *Implementing these laws has led to building over 1,000 new land treatment systems since 1973 and I expect to see even more new systems in the next 10 years.*

3. *As one who must make recommendations on guidance I am here to find out if our present guidance is (1) too conservative and hence costly, (2) adequate and needing only fine tuning, or (3) too risky and hence subjecting the public to unacceptable risks.*

4. *This morning I have heard that the major research efforts of the last 10 years have not identified any detectable risk but this is still not a positive indication that present design guidance is conservative, adequate, or risky.*

5. *I hope that the workshop sessions produce something more definitive and useful as the output from 10 years of research.*

AN OVERVIEW OF PUBLIC HEALTH EFFECTS

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INTRODUCTION

The legislative mandate for the greater use of land application of municipal wastewater and sludge is found in the Clean Water Act of 1977 (PL 95-217), Title II (Grants for Construction of Treatment Works), Section 201, which states that the:

> Administrator shall encourage waste treatment management which results in the construction of revenue producing facilities providing for (1) the recycling of potential sewage pollutants through the production of agriculture, silviculture, or aqua-culture products, or any combination thereof...

Moreover, the Act requires that a construction grant not be made unless:

> ...the grant applicant has satisfactorily demonstrated to the Administrator that innovative and alternative wastewater treatment processes and techniques which provide for the reclaiming and reuse of water, otherwise eliminate the discharge of pollutants, and utilize recycling techniques, land treatment, new or improved methods of waste treatment management for municipal and industrial waste (discharged into municipal systems) and the confined disposal of pollutants, so that pollutants will not migrate to cause water or other environmental pollution, have been fully studied and evaluated by the applicant taking into account section 201(d) of this Act and taking into account and allowing to the extent practicable the more efficient use of energy and resources.

The two most important mechanisms in the U.S. for the recycling of municipal sewage are land treatment of wastewater and land application of sludge. There are three types of land treatment systems in general use: slow rate (or "irrigation"), rapid infiltration (or "infiltration-percolation"), and overland flow. Many of the examples of "land treatment" systems in the U.S. utilize wastewater treated by conventional means up to tertiary level (secondary in the case of overland flow). The objectives of these systems are usually to produce clean irrigation water (e.g., for golf course application) or highly treated water for groundwater recharge. From a wastewater treatment point of view, land application in these systems is a form of tertiary treatment or effluent "polishing," rather than true land treatment. In land treatment systems raw wastewater is given a minimum preapplication treatment, or "pretreatment," e.g., by a stabilization pond, before being applied to the land, and the land itself is the site of the major portion of the wastewater treatment.

Land application of sludge consists of the low-rate application (compared with a purely disposal operation) to agricultural, forest, or reclaimed land of municipal wastewater sludge which has been "stabilized" in some way, e.g., anaerobic digestion or composting. That land application of sludge is an important and probably growing practice in the U.S. is indicated by the results of a recent survey of 1008 publicly owned treatment works, account for over 2 million dry metric tons per day of sludge (Peirce and Bailey 1982). The survey found 17% of the total sludge to be utilized in large scale food-chain landspreading, 12% in large scale non-food-chain landspreading, and 21% in distribution and marketing systems (much of which probably ends up in gardens and lawns).

With the application to land of large volumes of wastewater and sludge, it is evident that considerable potential for adverse health effects exists. The major health concerns with land treatment of wastewater and land application of sludge are somewhat different. Thus, the potential exposure of humans through the routes of aerosols and groundwater is frequently emphasized with wastewater, and through the food chain with sludge. Nevertheless, the agents, or pollutants, of concern from a health effects viewpoint are almost the same in waste-water and sludge, so that a unified approach to the public health effects of land application of both is reasonable and convenient.

It is the purpose of this report to briefly examine the potential health effects of land application of wastewater and sludge, and to provide an appraisal of these effects. The agents of concern can be divided into the two broad categories of pathogens and toxic substances. The pathogens include bacteria (e.g., Salmonella and Shigella), viruses (i.e., enteroviruses, hepatitis virus, adenoviruses,

rotoviruses, and Norwalk-like agents), protozoa (e.g., Entamoeba and Giardia), and helminths (or worms, e.g., Ascaris, Trichuris, and Toxocara). The toxic substances include organics, trace elements (or heavy metals, e.g., cadmium and lead), nitrates, and sodium. Nitrates and sodium are usually not viewed as "toxic" substances, but are here so considered because of their potential hematological and long-term cardiovascular effects when present in water supplies at high levels. These agents form the basis of the main sections of this report. The major health effects of these agents are listed in Figure 1.

BACTERIA

The pathogenic bacteria of major concern in wastewater and sludge are listed in Table 1. All have symptomless infections and human carrier states, and many have important nonhuman reservoirs as well. The pathogenic bacteria of minor concern are listed in Table 2; this list is perforce somewhat arbitrary since almost any bacterium can become an opportunistic pathogen under appropriate circumstances, e.g., in the immunologically compromised or in the debilitated. Recent reviews of pathogens in wastewater include those by Benarde (1973), Burge and Marsh (1978), Elliott and Ellis (1977), Kristensen and Bonde (1977), and Menzies (1977).

Table 1. Pathogenic bacteria of major concern.

Name	Nonhuman reservoir
Campylobacter jejuni	Cattle, dogs, cats, poultry
Escherichia coli (pathogenic strains)	--
Leptospira spp.	Domestic and wild mammals, rats
Salmonella paratyphi (A,B,C)*	--
Salmonella typhi	--
Salmonella spp.	Domestic and wild mammals, birds, turtles
Shigella sonnei, S. flexneri, S. boydii, S. dysenteriae	--
Vibrio cholerae	--
Yersinia enterocolitica, Y. pseudotuberculosis	Wild and domestic birds and mammals

*Correct nomenclature: Salmonella paratyphi A, S. schottmuelleri, S. hirschfeldii, respectively.

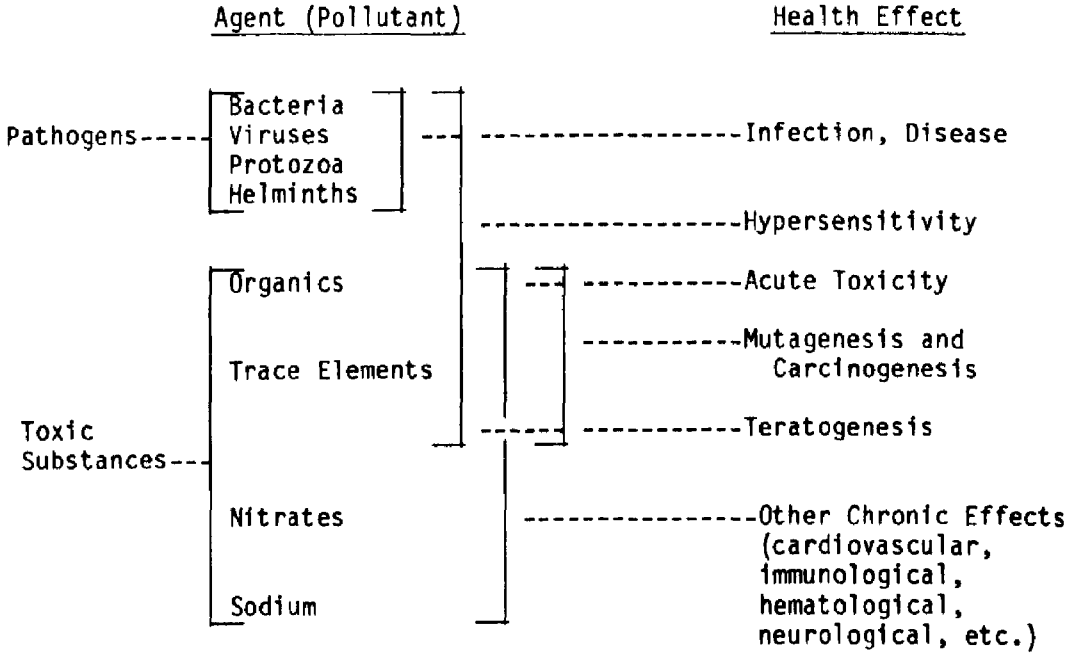

Figure 1. Major health effects of pollutants.

Table 2. Pathogenic bacteria of minor concern.

Aeromonas spp.
Bacillus cereus
Brucella spp.
Citrobacter spp.
Clostridium perfringens
Coxiella burnetii
Enterobacter spp.
Erysipelotrix rhusiopathiae
Francisella tularensis
Klebsiella spp.
Legionella pneumophila
Listeria monocytogenes
Mycobacterium tuberculosis
M. spp.
Proteus spp.
Pseudomonas aeruginosa
Serratia spp.
Staphylococcus aureus
Streptococcus spp.

Campylobacter jejuni (formerly *C. fetus* subsp. *jejuni*) is a recently-recognized cause of acute gastroenteritis with diarrhea. It is now thought to be as prevalent as the commonly recognized enteric bacteria *Salmonella* and *Shigella*, having been isolated from the stools of 4-8% of patients with diarrhea (MMWR 1979).

Pathogenic strains of the common intestinal bacterium *Escherichia coli* are of three types--enterotoxigenic enteropathogenic, and enteroinvasive (WHO Scientific Working Group 1980). All produce acute diarrhea, but by different mechanisms. Fatality rates may range up to 40% in newborns. Outbreaks occasionally occur in nurseries and institutions, and the disease is common among travelers to developing countries.

Leptospira spp. are bacteria excreted in the urine of domestic and wild animals, and enter municipal wastewater primarily from the urine of infected rats inhabiting sewers. Leptospirosis is a group of diseases caused by the bacteria, and may manifest itself through fever, headache, chills, severe malaise, vomiting, muscular aches, and conjunctivitis, and occasionally meningitis, jaundice, renal insufficiency, hemolytic anemia, and skin and mucous membrane hemorrhage. Fatality is low, but increases with age, and may reach 20% or more in patients with jaundice and kidney damage (Benenson 1975). In the

U.S., 498 cases were reported in 1974--78 (Martone and Kaufmann 1980). Direct transmission from humans is rare, with most infection resulting from contact with the urine of infected animals, e.g., by swimmers, outdoor workers, sewer workers, and those in contact with animals.

Salmonella paratyphi (A, B, C) causes paratyphoid fever, a generalized enteric infection, often acute, with fever, spleen enlargement, diarrhea, and lymphoid tissue involvement. Fatality rate is low, and many mild attacks exhibit only fever or transient diarrhea. Para-typhoid fever is infrequent in the U.S. (Benenson 1975).

Salmonella typhi causes typhoid fever, a systemic disease with a fatality rate of 10% untreated or 2-3% treated by antibiotics (Benenson 1975). It occurs sporadically in the U.S., where 647 cases were reported in 1979 (MMWR 1980a), but is more common in the developing countries.

Salmonella spp., including over 1000 serotypes, cause salmonellosis, an acute gastroenteritis characterized by abdominal pain, diarrhea, nausea, vomiting, and fever. Death is uncommon except in the very young, very old, or debilitated (Benenson 1975). In 1979, 30,476 cases were reported to the Center for Disease Control (CDC) (MMWR 1980a).

Shigella sonnei, S. flexneri, S.- boydii, and S. dysenteriae cause shigellosis, or bacillary dysentery, an acute enteritis primarily involving the colon, producing diarrhea, fever, vomiting, cramps, and tenesmus. There is negligible mortality associated with shigellosis (Butler et al., 1977). In 1979, 15,265 cases were reported to CDC (MMWR 1980b).

Vibrio cholerae causes cholera, an acute enteritis characterized by sudden onset, profuse watery stools, vomiting, and rapid dehydration, acidosis, and circulatory collapse. Fatality rates are about 50% untreated, but less than 1% treated (Benenson 1975). Cholera is rare in the U.S., there being no reported cases between 1911 and 1972, although one case occurred in 1973 in Texas and 11 in 1978 in Louisiana (Blake et al. 1980).

Yersinia enterocolitica and Y. pseudotuberculosis cause yersiniosis, an acute gastroenteritis and/or mesenteric lymphadenitis, with diarrhea, abdominal pain, and numerous other symptoms. Death is un-common. Yersiniosis occurs only sporadically in the U.S., and is transmitted from either infected animals or humans.

Most bacteria of concern in wastewater get there from human feces, although a few, such as Leptospira, enter through urine. The contribution from wash water, or "grey water," is probably relatively insignificant, except as it may contain opportunistic pathogens. The presence and levels in wastewater and sludge of any of the pathogens listed in Tables 1 and 2 depend, of course, on the levels of infection in the contributing population.

Bacteria: Potential Exposure

The human exposure to aerosol bacteria at land treatment sites can be roughly estimated from the data at Kibbutz Tzora, Israel, where raw wastewater was sprayed (Katzenelson and Teltch 1976), thus yielding higher bacterial levels than those found at Deer Creek, Ft. Huachuca, or Pleasanton, where treated wastewater was sprayed. Thus, an adult male, engaged in light work, breathing at a rate of $1.2 m^3/hr$, and exposed to 34 coliforms/m^3 (the Kibbutz Tzora average) at 100 m downwind from a sprinkler, would inhale approximately 41 coliforms per hour. Since the ratio of aerosolized Salmonella to coliforms is $1:10^5$ (Grunnet and Tramsen 1974) the rate of inhalation of Salmonella would be about 10^5-fold less, an extremely low rate of bacterial exposure. More recent data from Kibbutz Tzora allows a more accurate estimate of human exposure (Teltsch et al. 1980). During a period of time in 1977-78, when the wastewater total coliforms were 2.4×10^6 to 1.4×10^7/100 ml and Salmonella was 0-60/100 ml, the density of aerosol Salmonella at 40 m, the maximum distance found, was 0-0.054 m^3, with a mean of 0.014/m^3. This would result in an inhalation rate of 0.017/hr at 40 m, higher than the previous estimate, but still an extremely low rate of bacterial exposure (cf. the infective dose discussion below). Harding et al. (1981) have studied the production of microbial aerosols by the land application of liquid municipal sludge at three sites using tank-truck application and three sites using high-volume spray guns. Very low bacterial aerosol levels were found at the tank-truck sites, but elevated levels of fecal coliforms, fecal streptococci, and mycobacteria were found at the spray sites.

Survival times of bacteria on crops have been summarized by Bryan (1977), Sepp (1971), Feachem et al. (1978), Rudolfs et al. (1951a), and Larkin (1978a).

The survival of bacteria on plants, particularly crops, is especially important since these may be eaten raw by animals or humans, may contaminate hands of workers touching them, or may contaminate equipment contacting them. Such ingestion or contact would probably not result in an infective dose of a bacterial pathogen, but if contaminated crops are brought into the kitchen in an unprocessed state they could result in the regrowth of pathogenic bacteria, e.g.,

Salmonella, in a food material affording suitable moisture, nutrients, and temperature (Bryan, 1977). It should be kept in mind that many bacteria on plants, as well as soil, are not contaminants from human beings. For example, Klebsiella spp., Enterobacter spp., Serratia spp., and Pseudomonas aeruginosa are believed to be part of the natural flora of vegetables (Remington and Schimpff 1981).

Thus, the consumption of subsurface and low-growing food crops, e.g., leafy vegetables and strawberries, harvested from an application site within about six months of last application, is likely to increase the risk of disease transmission, because of contamination with soil and bacterial survival in cracks, leaf folds, leaf axils, etc. Possible approaches to avoid this problem are (1) use of the subsurface or covered drip irrigation method for aerial crops (Sadovski et al., 1978a, 1978b), (2) growth of crops the harvested portion of which does not contact the soil, e.g., grains and orchard crops, or (3) growth of crops used for animal feed only, e.g., corn (maize), soybeans, or alfalfa. The last alternative is probably the most common and most economic. In the situation where the harvested portion does not contact the soil nor is within splash distance, stopping application a month prior to harvest would be prudent.

Although properly designed slow-rate land treatment systems probably pose little bacterial threat to groundwater, considerable threat exists at rapid-infiltration sites where the water table is shallow, particularly if the soil is porous. Over 60 million people in the United States are served by public water supplies using groundwater, and about 54 percent of the rural population and 2 percent of the urban population obtain their water from indivdual wells (Duboise et al., 1979). Thus, it is imperative that land treatment systems do not result in the transmission of disease through groundwater. This is not to imply that groundwater in the U.S. is now pristine. Almost half of the waterborne disease outbreaks in the U.S. between 1971 and 1977 were caused by contaminated groundwater (Craun, 1979), and a recent examination of individual groundwater supplies in a rural neighborhood of Oregon (Lamka et al., 1980) showed more than one-third to be fecally contaminated. But, thus far, no disease outbreaks have been attributed to wastewater land treatment systems (Gerba and Lance, 1980).

Land application of sludge probably poses less of a bacterial threat to groundwater. Thus, Liu (1982) in Canada has found that after 4 years of heavy sludge application sewage bacteria were incapable of moving through the soil columns tested, and over 90% of the surviving sludge bacteria were still detained in the top 20 cm layer of soil. He concluded that there was little possibility of bacterial contamination of groundwater by the practice of sludge farmland application, provided that the water table was not too high and the soil

was well-drained. Similar results have been found in leachate experiments in South Africa (Nell et al., 1981).

Bacteria: Effects on Animals

The disease hazards to farm animals from land application of sludge have been reviewed by Argent et al. (1977) and Carrington (1978). The major bacterial concerns with respect to animals grazing at land application sites are Salmonella infections and bovine tuberculosis (Mycobacterium bovis and M. tuberculosis); both can be passed on to man.

That the transmission of salmonellosis to cattle grazing at land application sites is at least possible was demonstrated by Taylor and Burrows (1971), who showed that calves grazing pastures, to which 10^6 Salmonella dublin organisms/ml of slurry had been applied, became infected. No infection occurred when the rate was decreased to 10^3/ml, suggesting that Salmonella may only be of concern when high concentrations are present. At the San Angelo, Texas, slow-rate land treatment site, although Salmonella was isolated from the soil and the seepage creeks, the proportion of cattle grazing the pastures that were shedding Salmonella in their manure was not unusually high (Weaver et al., 1978). Feacham et al. (1978) concluded that there is no clear evidence that cattle grazed at land treatment sites are more at risk from salmonellosis than other cattle, probably because the required infectious doses are high and Salmonella infections are transmitted among cattle in many other ways. On the basis of Salmonella measurements in wastewater and sludge in England, Jones et al., (1980) concluded that a four-week waiting period would prevent salmonellosis in grazing animals.

Argent et al. (1981) have examined the Salmonella content of sludges treated in several ways, applied raw sludge (11 Salmonella/100 ml) to a field at the rate of 44.8 m^3/ha, and confined 10 lambs to the field for 2 months. None of the lambs became infected, as measured by feces, rumen, and tissue samples, and clinical symptoms. Ayanwale et al. (1980) raised goats on corn silage grown on sludge-amended land, and found no Salmonella infections in spite of the presence of Salmonella in the sludge, supporting the position that the potential public health hazard resulting from the use of sludge as fertilizer when properly treated has so far proven not to be a threat. Nevertheless the significance of Salmonella in land applied sludge is an issue yet to be settled. Evidence in Switzerland from studies of carrier rates and serotypes in cattle grazed on sludge-treated pastures has indicated a positive association and a cycle of infection from man to sludge to animals to man. Experience in the Netherlands is similar, but there is no evidence of such a link in the United Kingdom, despite the compulsory reporting of incidents (WHO 1981).

Several investigations on tuberculosis infection of cattle grazing on wastewater-irrigated land have been performed in Germany, with the conclusion that if application is stopped 14 days before pasturing, there is no danger that grazing cattle will contract bovine tuberculosis (Sepp, 1971).

Other possible bacterial concerns with respect to animals grazing at land treatment sites are Leptospira (causing leptosoirosis), Brucella (causing brucellosis), and Bacillus anthracis (causing anthrax. Wastewater, however, probably contains insignificant numbers of these pathogens, and plays a negligible role in the transmission of these diseases (Feachem et al., 1978).

Bacteria: Infective Dose and Risk of Infection

Upon being deposited on or in a human body a pathogen may be destroyed by purely physical factors, e.g., desiccation or decomposition. Before it can cause an infection, and eventually disease, it must then overcome the body's natural defenses. In the first interaction with the host, whether in the lungs, in the gastrointestinal tract, or other site, the pathogen encounters nonspecific immunologic responses, i.e., inflammation and phagocytosis. Phagocytosis is carried out primarily by neutrophils or polymorphonuclear leukocytes in the blood, and by mononuclear phagocytes, i.e., the monocytes in the blood and macrophages in the tissues (e.g., alveolar macrophages in the lungs). Later interactions with the host result in specific immunologic responses, i.e., humoral immunity via the B-lymphocytes, and cell-mediated immunity via the T-lymphocytes (Bellanti, 1976).

With these barriers to overcome it is understandable that an infection resulting from inoculation by a few bacterial cells is a most unlikely occurrence; usually large numbers are necessary. Some representative oral infection dose data for enteric bacteria, based upon numerous studies using non-uniform techniques, are presented in Table 3 (adapted from Bryan, 1977).

Although the terms, "infective dose," "minimal infectious dose," etc., are used in the literature, it is obvious from Table 3 that these are misnomers, and that we are really dealing with dose-response relationships, where the dose is the number of cells to which the human is exposed, and the response is lack of infection, infection without illness, and infection with illness (in an increasing proportion of the test subjects). The response is affected by many factors, making it highly variable. Some of the most important factors are briefly discussed below.

Table 3. Infective dose to man of enteric bacteria.

Bacterium	No Infection or no Illness	Infections without illness	Percent of volunteers developing illness			
			1-25	26-50	51-75	76-100
Clostridium perfringens				10^8	10^9	10^9
Escherichia coli						
0124:K72:H-		10^{10}	10^8			
0148:H28			10^8			10^{10}
0111:B4					10^6-10^9	
Several strains	10^4	10^4-10^6	10^6	10^8	10^8-10^{10}	10^{10}
Salmonella typhi						
Ty2W			10^8			
Zermat vi					10^4	
Most strains	10^3		10^5	10^5-10^8		10^8-10^9
S. newport			10^5	10^6		
S. bareilly			10^5	10^6		
S. anatum	10^4-10^6		10^5-10^8	10^6		
S. meleagridis	10^4-10^6		10^6	10^7	10^7-10^8	
S. derby	10^5-10^6			10^7		
S. pullorum	10^4-10^9			10^9		10^9-10^{10}
Shigella dysenteriae			10-10^2	10^2-10^4	10^3	10^4
S. flexneri			10^2-10^4		10^3-10^9	10^6-10^8
Streptococcus faecalis var. liquefaciens	10^8		10^9	10^{10}		
Vibrio cholerae						
$NaHCO_3$-buffered	10	10^3		10^3-10^8	10^4-10^6	
Unbuffered	10^4-10^{10}			10^8-10^{11}		

1. The site of exposure determines what types of defense mechanisms are available, e.g., alveolar macrophages and leukocytes in the lungs, and acidity and digestive enzymes in the stomach. The effect of acidity is clearly shown by the cholera (Vibrio cholerae) data in Table 3, where buffering reduced the infective dose by about a thousandfold. Direct inoculation into the bloodstream results in the fewest barriers being presented to the pathogen; Hellman et al. (1976) found 10 tuleremia organisms injected to be comparable to 10^8 by mouth.

2. Previous exposure to a given pathogen often produces varying degrees of immunity to that pathogen, through the induction of specific immune responses. A study in Bangladesh showed that repeated ingestion of small inocula (10^3-10^4 organisms) of Vibrio cholerae produced subclinical or mild diarrhea infection followed by specific antibody production. For this reason the peak incidence of endemic cholera occurs in the one to four-year old age group, and decreases with age thereafter as immunity develops (Levine, 1980).

3. Other host factors, such as age and general health, also affect the disease response. Infants, elderly persons (Gardner, 1980), malnourished people, those with concomitant illness, and people taking antiinflammatory, cytotoxic, and immunosuppressant drugs would be more susceptible to pathogens. An example of human variability (possibly genetic) is the following response of men orally challenged with several different doses of Salmonella typhi (Hornick et al. 1970):

Number of S. typhi	Percent developing typhoid fever
10^3	0
10^5	28
10^7	50
10^9	95

Twenty-eight percent of the men came down with typhoid fever after 10^5 organisms, while five percent were still resistant to 10^9 organisms, four orders of magnitude as many.

4. The number of organisms that must be swallowed for intestinal colonization (subclinical infection), and consequent risk of clinical disease, to be established is affected by treatment with antibiotics (Remington and Schimpff 1981). Due to its normal content of anaerobic bacteria and their products, the gut can resist colonization when an oral dose of about 10^6 organisms is given. Once resistance is reduced by systemic or oral antibiotics, the dose required to induce colonization is only ten to 100 organisms.

5. The timing of the exposure to pathogens, e.g., as a single exposure or an exposure over a long period of time, would be expected to affect the response.

6. Finally, as illustrated by Escherichia coli and Salmonella typhi in Table 3, the virulence, or pathogenicity, of bacteria varies among strains. Thus, three different strains of Shigella flexneri have been found to have infective doses of 10^{10} or higher, 10^5-10^8, and 180 organisms (NRC, 1977).

The risk of infection is probably greatest for Salmonella spp. and Shigella spp., because they are the most common bacterial pathogens in municipal wastewater. The infective dose for Salmonella is high (10^5-10^8 organisms) but this dose might be reached on a contaminated foodstuff under conditions that allow multiplication. A recent review of experimentally-induced salmonellosis and salmonellosis outbreaks, however, has resulted in the conclusion that the infective dose for Salmonella may well be below 10^3 organisms (Blaser and Newman, 1982). On the other hand the infective dose for Shigella is low--as few as 10 to 100 organisms. "Because of this miniscule inoculum it is rather simple for shigellae to spread by contact without interposition of a vehicle such as food, water or milk to amplify the infectious dose" (Keusch, 1979).

Consequently, it would be prudent for humans to maintain a minimum amount of contact with an active land application site, and to rely on "time" to reduce the bacterial survival, as discussed earlier, when growing crops for human consumption.

Bacteria: Epidemiology

A number of epidemiological reports have attested to the fact that transmission of enteric disease can occur when untreated wastewater is used in the cultivation of crops to be eaten raw (Geldreich and Bordner, 1971; Hoadley and Goyal, 1976; and Sepp, 1971). Salmonellosis has been traced to the consumption of wastewater-irrigated celery, watercress, watermelon, lettuce, cabbage, endive, salad vegetables, and fruits; shigellosis to wastewater-irrigated pastureland; and cholera to wastewater-irrigated vegetables in Israel.

Perhaps the largest epidemiological study of the health effects of land treatment was a retrospective study of 77 kibbutzim (agricultural cooperative settlements) in Israel practicing slow-rate land treatment with nondisinfected oxidation pond effluent, and 130 control kibbutzim (Katzenelson et al., 1976). The incidence of

typhoid fever, salmonellosis, shigellosis, and infectious hepatitis was 2-4 times higher in the land-treatment kibbutzim than the controls. The study, however, did not rule out a number of pathways of infection other than aerosols, e.g., direct contact via clothing or bodies of sewage irrigation workers, and there were problems with the data reporting methods. Consequently, it is generally felt that no conclusive findings may be based on the report, and the study has been repeated, correcting for the deficits of the original study (Shuval and Fattal, 1980). The final report of the revised study (Shuval et al, 1983) did not confirm the extreme differences found in the 1976 report, although a small significant excess risk of total enteric disease was found during effluent irrigation periods. None of the findings provided direct support for the hypothesis that the main pathway of pathogens transmissions was aerosolized effluent resulting from sprinkler irrigation.

Other epidemiological reports on the health effects of land application have been more superficial. Examination of the workers on sewer farms in Berlin and Memmingen in Germany has not shown them to have a higher rate of infectious diseases or worm infestation than the rest of the population (Sepp, 1971). At land treatment sites near Paris, grain for cattle, beef cattle, and vegetables (e.g., beans, onions, and celeriac) are raised (Dean, 1978). The vegetables are checked for Salmonella, with none having been found, and no disease has been traced to the farms. During a cholera outbreak, no cholera bacteria were found on the vegetables. At Werribee Farm in Melbourne, Australia, there has never been a reported epidemic or outbreak of disease among employees or residents, although no precautions other than normal hygiene practices have been taken, and the general health of employees and residents is no different from that of the community in general (Croxford, 1978).

Although these retrospective studies are reassuring, a better measure of the health effects of land application will come from well-planned prospective epidemiological studies. Two such studies, on land treatment, are currently underway--at Lubbock, Texas, and in Israel. A third study, on land application of sludge, is currently underway in Ohio. The results of these three projects may well modify the conclusions and recommendations of this report in the future.

VIRUSES

Transmission of viruses by feces is the second most frequent means of spread of common viral infections, the first being the respiratory route. Transmission by urine has not been established as being of epidemiological or clinical importance, although some viruses, e.g., cytomegalovirus and measles, are excreted through this

route. The gastrointestinal tract is an important portal of entry of viruses into the body, again second to the respiratory tract (Evans, 1976).

The human enteric viruses that may be present in wastewater and sludge are listed in Table 4 (Melnick et al., 1978; Holmes, 1979). These are referred to as the enteric viruses and new members are constantly being identified. Since no viruses are normal inhabitants of the gastrointestinal tract and none of these have a major reservoir other than man (with the likely exception of rotaviruses), all may be regarded as pathogens, although most can produce asymptomatic infections.

Table 4. Human wastewater viruses.

Enteroviruses
Poliovirus
Coxsackievirus A
Coxsackievirus B
Echovirus
New Enterovirus
Hepatitus A Virus
Rotavirus ("Duovirus", "Reovirus-like Agent")
Norwalk-Like Agents (Norwalk, Hawaii, Montgomery County, etc.)
Adenovirus
Reovirus
Papovavirus
Astrovirus
Calicivirus
Coronavirus-Like Particles

Upon entry into the alimentary tract, if not inactivated by the hydrochloric acid, bile acids, salts, and enzymes, enteroviruses, hepatitis A virus, rotavirus, adenovirus, and reovirus may multiply within the gut. The multiplication and shedding of adenovirus and reovirus here has not been shown to be of major epidemiological importance in their transmission (Evans, 1976). The rotavirus often produces diarrhea in children, but the local multiplication of enteroviruses and (possibly) hepatitis A virus in cells lining the area rarely produces local symptoms, i.e., diarrhea, vomiting, and abdominal pain. Most enterovirus infections, even with the more virulent types, cause few or no clinical symptoms. Occasionally, after continued multiplication in the lymphoid tissue of the pharynx and gut, viremia may occur, i.e., virus enters the blood stream, leading to further virus proliferation in the cells of the reticuloendothelial system, and finally to involvement of the major target organs--the central nervous system, myocardium, and skin for the enteroviruses, and the liver for hepatitis A virus (Melnick et al., 1979; Evans, 1976).

Polioviruses cause poliomyelitis, an acute disease which may consist simply of fever, or progress to aseptic meningitis or flaccid paralysis (slight muscle weakness to complete paralysis caused by destruction of motor neurons in the spinal cord). Polio is rare in the United States, but may be fairly common in unimmunized populations in the rest of the world. No reliable evidence of spread by wastewater exists (Benenson, 1975).

Coxsackieviruses may cause aseptic meningitis, herpangina, epidemic myalgia, myocarditis, pericarditis, pneumonia, rashes, common colds, congenital heart anomalies, fever, hepatitis, and infantile diarrhea.

Echoviruses may cause aseptic meningitis, paralysis, encephalitis, fever, rashes, common colds, epidemic myalgia, pericarditis, myocarditis, and diarrhea.

The new enteroviruses may cause pneumonia, bronchiolitis, acute hemorrhagic conjunctivitis, aseptic meningitis, encephalitis, and hand-foot-and-mouth disease. The prevalence of the diseases caused by the coxsackieviruses, echoviruses, and new enteroviruses is poorly known, but 7075 cases were reported to the Center for Disease Control (CDC) in the years 1971-75 (Morens et al., 1979). These enteroviruses are practically ubiquitous in the world, and may spread rapidly in silent (asymptomatic) or overt epidemics, especially in late summer and early fall in temperate regions. Because of their antigenic inexperience, children are the major target of enterovirus infections, and serve as the main vehicle for their spread. Most of these infections are asymptomatic, and natural immunity is acquired with increasing age. The poorer the sanitary conditions, the more rapidly immunity develops, so that 90% of the children living under poor hygienic circumstances may be immune to the prevailing enteroviruses (of the approximately 70 types known) by the age of 5. As sanitary conditions improve, the proportion of nonimmunized in the population increases, and infection becomes more common in older age groups, where symptomatic disease is more likely and is more serious (Melnick et al., 1979; Benenson, 1975). Thus, decreasing the human exposure to the common enteric viruses through the water and food route has its disadvantages, as well as advantages.

Hepatitis A virus causes infectious hepatitis, which may range from an inapparent infection (especially in children) to fulminating hepatitis with jaundice. Recovery with no sequelae is normal. Approximately 40,000-50,000 cases are reported annually in the U.S. About half the U.S. population has antibodies to hepatitis A virus, and the epidemiological pattern is similar to that of enteroviruses, with childhood infection common and asymptomatic (Duboise et al., 1979).

Rotavirus causes acute gastroenteritis with severe diarrhea, sometimes resulting in dehydration and death in infants. It may be the most important cause of acute gastroenteritis in infants and young children, especially during winter (Konno et al., 1978), but also may strike older children and adults (Holmes, 1979).

Norwalk-like agents include the Norwalk, Hawaii, Montgomery County, Ditchling, W, and cockle viruses, and cause epidemic gastroenteritis with diarrhea, vomiting, abdominal pain, headache, and myalgia or malaise. The illness is generally mild and self-limited (Kapikian et al., 1979). These agents have been associated with sporadic outbreaks in school children and adults (Holmes, 1979).

Adenoviruses are primarily causes of respiratory and eye infection, transmitted by the respiratory route, but several strains are now believed to be important causes of sporadic gastroenteritis in young children (Richmond et al., 1979; Kapikian et al., 1979).

Reoviruses have been isolated from the feces of patients with numerous diseases, but no clear etiological relationship has yet been established. It may be that reovirus infection in humans is common, but associated with either mild or no clinical manifestations (Rosen 1979).

Papovaviruses have been found in urine, and may be associated with progressive multifocal leukoencephalopathy (PML), but are poorly understood (Warren, 1979).

Astroviruses, caliciviruses, and coronavirus-like particles may be associated with human gastroenteritis, producing diarrhea, but are also poorly understood (Holmes, 1979; Kapikian et al., 1979).

Viruses: Potential Exposure

Aerosols have been of concern as a potential route of transmission of disease caused by enteric viruses because, as with bacteria, once they are inhaled they may be carried from the respiratory tract by cilia into the oropharynx, and then swallowed into the gastrointestinal tract. Some enteroviruses may also multiply in the respiratory tract itself (Evans, 1976). Another reason for concern is the theoretically possible transmission of respiratory viruses through wastewater aerosols. On the basis of actual viral sampling of wastewater, however, Johnson et al. (1980) concluded that the likelihood of finding respiratory viruses in treated wastewater is very small.

The level of enteric viruses in aerosols at land treatment sites is strongly dependent on the degree of pretreatment of the wastewater. Thus, at the Muskegon County (Michigan) Wastewater Management System, where large lagoons are used, no animal viruses were detected in air samples collected at the spray irrigation area (Clark et al., 1981). On the basis of data obtained at the Pleasanton, California, land treatment site (Johnson et al., 1980), it can be calculated that an adult male, engaged in light work, breathing at a rate of 1.2 m^3/hour and exposed to 0.014 PFU/m^3 at 50 m downwind from a sprayer, would inhale approximately 0.13 PFU of enterovirus during and 8-hour work day. This is probably an insignificant level of exposure. However, since the recovery of enteric viruses from environmental samples is not perfectly efficient, isolation of viruses increases as more cell culture types are used, and some enteric viruses cannot yet be isolated on cell cultures, the actual exposure to enteric viruses may be as much as ten to a hundred times the reported level (Teltsch et al., 1980). Thus, it might be prudent to recommend a 100 m or 200 m minimum exposure distance of the general public to a land treatment spray source.

The concentration of viruses in aerosols at liquid sludge spray-application sites has been examined by Harding et al. (1981). On a special virus run, 1470 m^3 of air was sampled and no human enteric viruses were detected from the pooled sample. This converts to a concentration of less than 0.0016 PFU/m^3 of air, and the implication is that aerosolization of viruses in liquid sludge does not present a significant health risk.

Regrowth of viruses on crops and food is not a problem, the way it is with bacteria, but the crop route of exposure is still of concern. Much of the literature on survival time of viruses on crops has been summarized by Feachem et al. (1978). The data are similar to those for bacteria, and likewise appear to support a one-month waiting period after last application before harvest.

Because of the possible contamination of subsurface and low-growing crops with soil, in which viruses have a longer survival time, about one hundred days would probably be a safe waiting period. As with bacteria, this period could be shortened by (1) the use of subsurface or covered drip irrigation (Sadovski et al., 1978a; 1978b), (2) the growth of crops the harvested portion of which does not contact the soil, or (3) the growth of crops used for animal feed only. At a site where (2) and (3) were practiced, the Roswell, New Mexico, slow-rate land treatment site where secondary effluent has been applied by ridge-and-furrow irrigation for 33 years, no enteroviruses were found on or in the leaf and grain portions of corn (Koerner and Haws, 1979).

Once enteric viruses get into groundwater, they can survive for long periods of time, 2 to 188 days having been reported in the litera-ture (Akin et al., 1971), and probably migrate for long distances (Keswick and Gerba, 1980). Low temperatures prolong survival, but the factors affecting survival in groundwater are poorly understood. It might be possible, for example, that entry of viruses into the groundwater would be tolerable if sufficient underground detention time could be provided before movement of the groundwater to wells or streams (Lance and Gerba, 1978). Until these factors are well-understood, it would be prudent to assume that groundwater underlying coarse sandy or gravelly soils, and in the vicinity of rapid-infiltration land treatment sites or septic tank-leaching field systems, is contaminated with viruses. Groundwater drawn from such sources for use as potable water supplies should be disinfected; this advice is consistent with that of the World Health Organization Scientific Group on Human Viruses in Water, Wastewater and Soil (WHO, 1979).

With reference to viruses in soil and groundwater at land treatment sites Gerba and Lance (1980) have concluded:

> Although the presence of viruses in groundwater has been demonstrated, it would appear that with proper site selection and management the presence of viruses could be minimized or eliminated. The key is to define the processes involved in the survival and transport of pathogens in groundwater. With proper design, land treatment could be used as an effective method for reducing the number of pathogens in wastewater. With the proper soil type, viruses and bacteria can be reduced to levels as effective as chlorination as currently practiced, after the travel of wastewater through only a few centimeters of soil.

The problem of viruses in groundwater is still not settled, however, as evidenced by the far more cautious view taken by Wellings (1982). There appears to be little reliable information on viruses getting into groundwater beneath sludge application sites, although one would expect the threat to be low.

An unusual route of exposure to viruses as a result of land application is through mosquitoes. Zaim and Newson (1980) studied the effects of wastewater spray irrigation on indigenous mosquito populations in Michigan agricultural and forested land. Mosquito populations remained negligible in the farmed portion of the site, but a dramatic increase occurred in the species diversity in the wooded portions. Most of the species found bred outside the sprayed area, but three Culex species, including potential transmitters of St. Louis encephalitis, bred in pools and ponds in the most heavily sprayed area. Other species,

from outside the sprayed area, are vectors of dog heartworm and California encephalitis. The authors concluded that spray irrigation of wastewater enhances the potential for pest and disease transmission in the vicinity of such sites.

Viruses: Effects on Animals

Human polioviruses, coxsackieviruses, echoviruses, and reoviruses have been recovered from, or found to produce infection in, at least six species of animals--dogs, cats, swine, cattle, horses, and goats (Metcalf, 1976). Dogs and cats were found to be involved in a majority of instances, probably because of their intimate association with man in the household. The present state of information on virus transmission in animals and man does not appear to allow an evaluation of the effect of land application on animal infections or the role of animals as reservoirs of human disease (Metcalf, 1976).

Polley (1979) noted that, under experimental conditions, rotaviruses of human origin have infected pigs, calves, and lambs, but concluded that in Canada their transmission to livestock via effluent irrigation was a slight and unproven risk.

Viruses: Infective Dose, Risk of Infection, and Epidemiology

In contrast with bacteria, where large numbers of cells are usually necessary to produce an infection, a few virus particles are currently thought to be able to produce an infection under favorable conditions. The most important studies on the oral infective dose of enteric viruses in humans are summarized in Table 5 (modified from National Research Council, 1977). The results are highly variable, and may reflect differences in experimental conditions as well as states of the hosts. The recent data do suggest, however, that the infective dose of enteroviruses to man is low, possibly of the order of 10 virus particles or less. The same factors discussed earlier, that affect bacteria also affect the virus dose-response relationship.

Since a potential route of exposure to viruses at land application sites is aerosols, it is of great importance to compare the infective dose through the respiratory route with that through the ingestion route. Couch et al. (1965) and Gerone et al. (1966) reported the human inhalation infective dose of coxsackievirus A21 to be <18 TCD_{50} which is comparable with the oral infective dose of the enteroviruses.

Table 5. Oral infective dose to man of enteric viruses.

Virus	Subjects	Dose*	Percent Infected	Reference
Vaccine poliovirus	Infants	0.2 PFU**	0	Koprowski 1956
		2 PFU	67	
		20 PFU	100	
		$10^{5.5}$	50	Gelfand *et al.* 1960
		$10^{7.5}$	100	
		$10^{6.5}$	60	Krugman *et al.* 1961
		$10^{7.5}$	75	
		5.5×10^{6} PFU	89	Holgiun *et al.* 1962
		$10^{3.5}$	29	Lepow *et al.* 1962
		$10^{4.5}$	46	
		$10^{5.5}$	57	
		$10^{3.5}$	68	Warren *et al.* 1964
		$10^{5.5}$	79	
	Premature Infants	1	30	Katz and Plotkin 1967
		2.5	33	
		10	67	
	Infants	7-52†	1	Miner *et al.* 1981
		24-63	10	
		55-93	50	
Echovirus 12	Young Adults	10 PFU	18	Schiff *et al.* (personal communication) 1980
		100 PFU	67	

* Tissue culture dose 50% (TCD_{50}) unless indicated
**Plaque-Forming Unit
†95% Confidence Units

Theoretically, a single virus particle is capable of establishing infection both in a cell in culture and in a mammalian host (Westwood and Sattar, 1976). If this were to be the case, extreme care should be taken to avoid human exposure to enteric viruses through aerosols or crops grown on land application sites. On the other hand, the concept that a single virus particle often constitutes an infective dose in the real world has been argued against (Lennette, 1976) on the basis of the oral poliovaccine studies, nonimmunologic barriers, human immunologic responses, and probabilistic factors.

Viruses do not regrow on foods or other environmental media, as bacteria sometimes do. Therefore, the risk of infection is completely dependent upon being exposed to an infective dose (which may be very low) in the material applied. In any event, as is the case with bacteria, it would seem prudent for humans to maintain a minimum amount of contact with an active land application site, and to rely on the viral survival data discussed earlier for limiting the hazard from crops grown for human consumption on wastewater- or sludge-amended soils.

Fecally-polluted vegetable-garden irrigation water in Brazil has been found to contain polioviruses and coxsackieviruses, and has been associated with earlier epidemics (Christovao et al., 1967a; 1967b). At the Muskegon land treatment site spray irrigation workers did not have increased illness or virus isolation rates (Clark et al., 1981). Nor did they, with a minor exception, have increased prevalence of infection by hepatitis A, poliovirus (1, 2, 3), coxsackievirus (B2, B5), or echovirus (7, 11), as measured by serology. The exception was a high antibody titer to coxsackievirus B5 in the spray nozzle cleaners, a group with presumably high exposure to wastewater. Some virologists, however, feel that current epidemiological techniques are probably not sufficiently sensitive to detect the low levels of viral disease transmission that might occur from a modern land application site (Melnick, 1978; WHO, 1979).

PROTOZOA

The protozoa and helminths (or worms) are often grouped together under the term, "parasites," although in reality all the pathogens are biologically parasites. Little attention has been given to the presence of parasites in wastewater, and their potential for contaminating food crops in the United States, probably because of the popular impression that the prevalence of parasite infection in the U.S. is minimal (Larkin et al., 1978b). However, because of the increasing recognition of parasite infections in the U.S., the return of military personnel and travelers from abroad, the level of recent immigration and food imports from countries with a high parasitic disease prevalence, and the existence of resistant stages of the organisms, a consideration of parasites is warranted.

The most common protozoa which may be found in wastewater are listed in Table 6. Of these, only three species are of major significance for transmission of disease to humans through wastewater: *Entamoeba histolytica*, *Giardia lamblia*, and *Balantidium coli*. *Toxoplasma gondii* also causes significant human disease, but the wastewater route is probably not of importance. *Eimeria* spp. are often identified in human fecal samples, but are considered to be spurious parasites, entering the gastrointestinal tract from ingested fish.

Table 6. Types of protozoa in wastewater.

Name	Protozoan Class	Nonhuman Reservoir
Human Pathogens		
Entamoeba histolytica	Ameba	Domestic and wild mammals
Giardia lamblia	Flagellate	Beavers, dogs, sheep
Balantidium coli	Ciliate	Pigs, other mammals
Toxoplasma gondii	Sporozoan (Coccidia)	Cats
Dientamoeba fragilis	Ameba	
Isospora belli	Sporozoan (Coccidia)	
I. hominis	Sporozoan (Coccidia)	
Human Commensals		
Endolimax nana	Ameba	
Entamoeba coli	"	
Iodamoeba butschlii	"	
Animal Pathogens		
Eimeria spp.	Sporozoan (Coccidia)	Fish, birds, mammals
Entamoeba spp.	Ameba	Rodents, etc.
Giardia spp.	Flagellate	Dogs, cats, wild mammals
Isospora spp.	Sporozoan (Coccidia)	Dogs, cats

Entamoeba histolytica causes amebiasis, or amebic dysentery, an acute enteritis, whose symptoms may range from mild abdominal discomfort with diarrhea to fulminating dysentery with fever, chills, and bloody and mucoid diarrhea. Most infections are asymptomatic, but in severe cases dissemination may occur, producing liver, lung, or brain abscesses, and death may result. Amebiasis is rare in the U.S. (Krogstad *et al.*, 1978), and is transmitted by cysts contaminating water or food.

Giardia lamblia causes giardiasis, an often asymptomatic infection of the small intestine, which may be associated with chronic diarrhea, malabsorption of fats, steatorrhea, abdominal cramps, bloating, fatigue, and weight loss. The carrier rate in different areas of the U.S. may range between 1.5 and 20% (Benenson, 1975), and it is transmitted by cysts contaminating water or food, and by person-to-person contact (Osterholm *et al.*, 1981).

Balantidium coli causes balantidiasis, a disease of the colon, characterized by diarrhea or dysentery. Infections are often asymptomatic, and the incidence of disease in man is very low (Benenson, 1975). Balantidiasis is s transmitted by cysts contaminating water, particularly from swine.

Toxoplasma gondii causes toxoplasmosis, a systemic disease which rarely gives rise to clinical illness, but which can damage the fetus if infection, and subsequent congenital transmission, occurs during pregnancy. Approximately 50% of the population of the U.S. is thought to be infected (Krick and Remington, 1978), but the infection is probably transmitted by oocysts in cat feces or the consumption of cyst-contaminated, inadequately-cooked meat of infected animals (Teutsch et al., 1979), rather than through wastewater.

The active stage of protozoans in the intestinal tract of infected individuals is the trophozoite. The trophozoites, after a period of reproduction, may round up to form precysts, which secrete tough membranes to become environmentally-resistant cysts, in which form they are excreted in the feces (Brown, 1969).

The types and levels of protozoan cysts actually present in wastewater depend on the levels of disease in the contributing human population, and the degree of animal contribution to the system.

Protozoa: Potential Exposure

Because of the large size of protozoan cysts and helminth eggs, compared with bacteria and viruses, it is unlikely that they will find their way into either aerosols or groundwater at land application sites.

Because of their exposure to the air, protozoan cysts deposited on plant surfaces would be expected to die off rapidly. The fact that cysts can survive long enough to get into the human food supply under poor management conditions is confirmed by the recent isolation of high levels of Entamoeba histolytica, E. coli, Endolimax nana, and Giardia lamblia on the wastewater-irrigated fruits and vegetables in Mexico City's marketplaces (Tay et al., 1980). Rudolfs et al. (1951b) found contaminated tomatoes and lettuce to be free from viable Entamoeba presence of organic matter in the form of fecal suspensions. They concluded that field-grown crops ". . . consumed raw and subject to contamination with cysts of E. histolytica are considered safe in the temperate zone one week after contamination has stopped and after two weeks in wetter tropical regions."

Therefore, if the recommendations, based on bacteria, for harvesting human food crops are followed, it is unlikely that any public health risk will ensue.

Protozoa: Health Effects

Although it would be theoretically possible for protozoan diseases to be transmitted through animals at a land application site, little relevent information on the subject appears to exist. However, in view of the survival times, the four week waiting period before the resumption of grazing, recommended on the basis of bacteria, should prevent any problem from developing.

Human infections with Giardia lamblia and the nonpathogenic Entamoeba coli have been produced with ten cysts administered in a gelatin capsule (Rendtorff 1954a; 1954b). Infections have been produced with single cysts of Entamoeba coli, and there is no biological reason why single cysts of Giardia would not also be infectious (Rendtorff, 1979). This is probably true for E. histolytica as well (Beaver et al., 1956). The pathogenicity of protozoa is highly variable among strains, and human responses likewise are variable. Thus, many infections are asymptomatic.

Because of the low infective doses of protozoan cysts, it would be prudent for humans to maintain a minimum amount of contact with an active land application site. However, if the recommended waiting periods for crop harvest are followed, the risk of infection should be minimal, because of the cysts' sensitivity to drying.

A few epidemiological reports have linked the transmission of amebiasis to vegetables irrigated with raw wastewater or fertilized with night soil (Bryan, 1977; Gelreich and Bordner, 1971).

HELMINTHS

The pathogenic helminths whose eggs are of major concern in wastewater and sludge are listed in Table 7. They are taxonomically divided into the nematodes, or roundworms, and cestodes, or tapeworms. The trematodes, or flukes, are not included since they require aquatic conditions and intermediate hosts, usually snails, to complete their life cycles, and thus are unlikely to be of concern at land treatment sites. Some common helminths, pathogenic to domestic or wild animals, but not to humans, are listed in Table 8 (after Reimers et al., 1980), since their eggs are likely to be identified in wastewater. Several of the human pathogens listed in Table 7, e.g., Toxocara spp., are actually

animal parasites, rather than human parasites, infesting man only incidentally, and not completing their life cycle in man.

Table 7. Pathogenic helminths of major concern in wastewater.

Pathogen	Common Name	Disease	Nonhuman Reservoir
NEMATODES (Roundworms)			
Enterobius vermicularis	Pinworm	Enterobiasis	
Ascaris lumbricoides	Roundworm	Ascariasis	
A. suum	Swine roundworm	Ascariasis	Pig*
Arichuris trichiura	Whipworm	Trichuriasis	
Necator americanus	Hookworm	Necatoriasis	
Ancylostoma duodenale	Hookworm	Ancylostomiasis	
A. braziliense	Cat hookworm	Cutaneous larva migrans	Cat, dog*
A. caninum	Dog hookworm	Cutaneous larva migrans	Dog*
Strongyloides stercoralis	Threadworm	Strongyloidiasis	Dog
Toxocara canis	Dog roundworm	Visceral larva migrans	Dog*
T. cati	Cat roundworm	Visceral larva migrans	Cat*
CESTODES (Tapeworms)			
*Taenia saginata***	Beef tapeworm	Taeniasis	
T. solium	Pork tapeworm	Taeniasis, Cysticercosis	
Hymenolepis nana	Dwarf tapeworm	Taeniasis	Rat, mouse
Echinococcus granulosus	Dog tapeworm	Unilocular hydatid disease	Dog*
E. multilocularis		Alveolar hydatid disease	Dog, fox, cat*

* Definitive host; man only incidentally infested
**Eggs not infective for man

Enterobius vermicularis, the pinworm, causes itching and discomfort in the perianal area, particularly at night when the female lays her eggs on the skin. A 1972 estimate of the prevalence of pinworm infections in the U.S. was 42 million (Warren, 1974). Although it is by far the most common helminth infection, the eggs are not usually found in feces, are spread by direct transfer, and live only for a few days.

Table 8. Animal-pathogenic helminths in wastewater.

Pathogen	Definitive Host
Trichuris suis	Pig
T. vulpis	Dog
*Toxascaris leonina**	Dog, cat
Ascaridia galli	Poultry
Heterakis gallinae	Poultry
Trichosomoides crassicauda	Rat
Anatrichosoma buccalis	Opossum
Cruzia americana	Opossum
Capillaria hepatica	Rat
C. gastrica	Rat
C. spp.	Poultry, wild birds, wild mammals
Hymenolepis diminuta	Rat
H. spp.	Birds
Taenia pisiformis	Cat
Hydatigera taeniaeformis	Dog
Macracanthorhynchus hirudinaceous	Pig

*Toxascaris leonina may produce visceral larva migrans in experimental animals, but its role in human disease is undefined (Quinn *et al.* 1980).

Ascaris lumbricoides, the large roundworm, produces numerous eggs, which require 1-3 weeks for embryonation. After the embryonated eggs are ingested, they hatch in the intestine, enter the intestinal wall, migrate through the circulatory system to the lungs, enter the alveoli, and migrate up to the pharynx. During their passage through the lungs they may produce ascaris pneumonitis, or Loeffler's syndrome, consisting of coughing, chest pain, shortness of breath, fever, and eosinophilia, which can be especially severe in children. The larval worms are then swallowed, to complete their maturation in the small intestine, where small numbers of worms usually produce no symptoms. Large numbers of worms may cause digestive and nutritional disturbances, abdominal pain, vomiting, restlessness, and disturbed sleep, or, occasionally, intestinal obstruction. Death due to migration of adult worms into the liver, gall bladder, peritoneal cavity, or appendix occurs infrequently. The prevalence of ascariasis in the U.S. was estimated to be about 4 million in 1972 (Warren, 1974).

Ascaris suum, the swine roundworm, may produce Loeffler's syndrome, but probably does not complete its life cycle in man (Phills *et al.*, 1972).

Trichuris trichiura, the human whipworm, lives in the large intestine with the anterior portion of its body threaded superficially through the mucosa. Eggs are passed in the feces, and develop to the infective stage after about four weeks in the soil (Reimers et al., 1980), and direct infections of the cecum and proximal colon result from the ingestion of infective eggs. Light infections are often asymptomatic, but heavy infections may cause intermittent abdominal pain, bloody stools, diarrhea, anemia, loss of weight, or rectal prolapse in very heavy infections. Human infections with T. suis, the swine whipworm, and T. vulpis, the dog whipworm, have been reported, but are uncommon (Reimers et al., 1980). The prevalence of trichuriasis in the U.S. was estimated to be about 2.2 million in 1972 (Warren, 1974). Reimers et al. (1980) have found Ascaris, Trichuris, and Toxocara to be the most frequently recovered helminth eggs in municipal wastewater sludge in southeastern United States.

Necator americanus and Ancylostoma duodenale, the human hookworms, live in the small intestine attached to the intestinal wall. Eggs are passed in the feces, and develop to the infective stage in 7-10 days in warm, moist soil. Larvae penetrate bare skin, usually of the foot (although Ancylostoma may also be acquired by the oral route), pass through the lymphatics and blood stream to the lungs, enter the alveoli, migrate up the pharynx, are swallowed, and reach the small intestine. During lung migration, a pneumonitis, similar to that produced by Ascaris, may occur (Benenson, 1975). Light infections usually result in few clinical effects, but heavy infections may result in iron-deficiency anemia (because of the secreted anticoagulant causing bleeding at the site of attachment) and debility, especially in children and pregnant women. The prevalence of hookworm in the U.S. (usually due to Necator) was estimated to be about 700,000 in 1972 (Warren, 1974).

Ancylostoma braziliense and A. caninum, the cat and dog hookworms, do not live in the human intestinal tract. Larvae from eggs in cat and dog feces penetrate bare skin, particularly feet and legs on beaches, and burrow aimlessly intracutaneously, producing "cutaneous larva migrans" or "creeping eruption." After several weeks or months the larva dies without completing its life cycle.

Strongyloides stercoralis, the threadworm, lives in the mucosa of the upper small intestine. Eggs hatch within the intestine, and reinfection may occur, but usually noninfective larvae pass out in the feces. The larva in the soil may develop into an infective stage or a free-living adult, which can produce infective larvae. The infective larvae penetrate the skin, usually of the foot, and complete their life cycle similarly to hookworms. Intestinal symptoms include abdominal pain, nausea, weight loss, vomiting, diarrhea, weakness, and constipation. Massive infection and autoinfection may lead to wasting and death in patients receiving immunosuppressive medication (Benenson, 1975). The prevalence of strongyloidiasis in the U.S. was estimated to be about

400,000 in 1972 (Warren, 1974). Dog feces is another source of threadworm larvae.

Toxocara canis and T. cati, the dog and cat roundworms, do not live in the human intestinal tract. When eggs from animal feces are ingested by man, particularly children, the larvae hatch in the intestine and enter the intestinal wall, similarly to Ascaris. However, since Toxocara cannot complete its life cycle, the larvae do not migrate to the pharynx, but instead wander aimlessly through the tissues, producing "visceral larva migrans," until they die in several months to a year. The disease may cause fever, appetite loss, cough, asthmatic episodes, abdominal discomfort, muscle aches, or neurological symptoms, and may be particularly serious if the liver, lungs, eyes (often resulting in blindness), brain, heart, or kidneys become involved (Fiennes, 1978). The infection rate of T. canis is more than 50% in puppies and about 20% in older dogs in the U.S. (Gunby, 1979), and Toxocara is one of the most common helminth eggs in wastewater sludge (Reimers et al., 1980).

Taenia saginata and T. solium, the beef and pork tapeworms, live in the intestinal tract, where they may cause nervousness, insomnia, anorexia, loss of weight, abdominal pain, and digestive disturbances, or be asymptomatic. The infection arises from eating incompletely cooked meat (of the intermediate host) containing the larval stage of the tapeworm, the cysticercus, however, rather than from a wastewater-contaminated material. Man serves as the definitive host, harboring the self-fertile adult. The eggs (contained in proglottids) are passed in the feces, ingested by cattle and pigs (the intermediate hosts), hatch, and the larvae migrate into tissues, where they develop to the cysticercus stage. The hazard, then, is principally to livestock grazing on land-treatment sites. The major direct hazard to man is the possibility of his acting as the intermediate host. While Taenia saginata eggs are not infective for man, those of T. solium are infective for man, in which they can produce cysticerci. Cysticercosis can present serious symptoms when the larvae localize in the ear, eye, central nervous system, or heart. Taeniasis with Taenia solium is rare in the U.S. and with T. saginata is only occasionally found. However, human infections with these tapeworms are fairly common in some other areas of the world.

Hymenolepis nana, the dwarf tapeworm, lives in the human intestinal tract, where it may be asymptomatic or produce the same symptoms as Taenia. Infective eggs are released, and internal autoinfection may occur, or more usually, eggs may be passed in the feces. No intermediate host is required, and upon ingestion, eggs develop into adults in the intestinal tract. The prevalence of infection in southern U.S. is 0.3 to 2.9 percent, mostly among children under 15.

Echinococcus granulosus and E. multilocularis, two dog tapeworms, do not live in the human intestinal tract. Dogs and other carnivores are their definitive hosts. Eggs in animal feces are usually ingested by an herbivore, in which they hatch into larval forms, which migrate into tissues, where they develop into hydatid cysts. When the herbivore is eaten by a carnivore the cysts develop into adult tapeworms in the carnivore's intestinal tract. If man ingests an egg, he can play the role of the herbivore, just as in cysticercosis. A hydatid cyst can develop in the liver, lungs, or other organs, where serious symptoms can be produced as the cyst grows in size or ruptures. The disease is rare in the U.S., but has been reported from the western states, Alaska, and Canada, particularly where dogs are used to herd grazing animals, and where dogs are fed animal offal.

The presence and levels in wastewater of any of these helminth eggs or of those from animal feces (Ancylostoma, Toxocara, and Echinococcus), depend on the levels of disease in the contributing population, and the degree of animal contribution to the system. Since helminth eggs are denser than water, most will settle to the bottom during a sedimentation unit process, and primary effluent should have fairly low densities of eggs. As a consequence, sludge will have high densities of viable helminth eggs, and will require either proper treatment before land application, or use under conditions that minimize exposure.

Helminths: Potential Exposure

As with protozoa, the large size of helminth eggs makes it unlikely that they will find their way into either aerosols or groundwater at land application sites.

Helminth eggs and larvae, in contrast to protozoan cysts, live for long periods of time when applied to the land, probably because soil is the transmission medium for which they have evolved, while protozoa have evolved toward water transmission. Thus, under favorable conditions of moisture, temperature, and sunlight, Ascaris, Trichuris, and Toxocara can remain viable and infective for several years (Little, 1980). Hookworms can survive up to six months (Feachem et al., 1978), and Taenia a few days to seven months (Babayeva, 1966); other helminths survive for shorter periods.

Because of desiccation and exposure to sunlight, helminth eggs deposited on plant surfaces die off more rapidly. Thus, Rudolfs et al. (1951c) found Ascaris eggs, the longest-lived helminth egg, sprayed on tomatoes and lettuce, to be completely degenerated after 27-35 days.

At rapid-infiltration land treatment sites there should be little risk to public health from helminths, as long as the site is dedicated to rapid infiltration. However, because of the growth of crops and presence of people at slow-rate, overland flow, and sludge application sites, and the longevity of helminth eggs, it might be advisable to select a preapplication treatment method which will completely remove or inactivate helminth eggs. From a less conservative point of view, Fitzgerald (1979) reviewed the potential impact on public health of parasites in soil/sludge systems, concluding that the proper utilization of wastewater sludge did not pose any great threat to the health of society through actual transmission of pathogens that might be present in sludge.

Helminths: Effects on Animals

The most serious threat to cattle at land application sites is the beef tapeworm, *Taenia saginata* (Feachem *et al.*, 1978; WHO, 1981). The increased incidence of cysticercosis in cattle results in economic losses (because of condemnation of carcasses), as well as increased incidence of disease in man. The application of wastewater sludge to pastures has resulted in outbreaks of cysticercosis in grazing cattle in England (Macpherson *et al.*, 1978, 1979), but wastewater land treatment sites at San Angelo, Texas (Weaver *et al.*, 1978), and Melbourne, Australia (Croxford, 1978; McPherson, 1978), have resulted in no increase of cysticercosis in grazing cattle.

Nevertheless, because of the longevity of helminth eggs in the soil, and the fact that cattle consume considerable quantities of soil as they graze it, it would be prudent to select a pretreatment method which will completely remove helminth eggs at land application sites where cattle are allowed to graze. Arundel and Adolph (1980) have suggested that stabilization ponds remove *Taenia saginata* quite efficiently. They found no cysticercosis in cattle grazed on pasture irrigated with effluent from lagooning, compared with a 3.3% infection rate from trickling filter effluent, 9.0-12.5% from activated sludge effluent, and 30.0% from raw sewage.

Helminths: Infective Dose, Risk of Infection, and Epidemiology

Single eggs of helminths are infectious to man, although since the symptoms of helminth infections are dose-related, many light infections are asymptomatic. However, *Ascaris* infection may sensitize individuals so that the passage of a single larval stage through the lungs may result in allergic symptoms, i.e., asthma and urticaria (Müller, 1953).

Because of the low infective doses of helminth eggs, and their longevity, it would be prudent for humans to maintain a minimum amount of contact with an active or inactive land application site, unless the wastewater or sludge has been pretreated to remove or inactivate helminths.

A few epidemiological reports have linked the transmission of Ascaris and hookworm to the use of night soil on gardens and small farms in Europe and the Orient (Geldreich and Bordner, 1971).

PATHOGENS: CONCLUSIONS AND RESEARCH NEEDS

The level of preapplication treatment required for the protection of public health may be as little as properly-designed sedimentation at land treatment sites with limited public access, where crops are protected by appropriate crop choice and waiting periods, and groundwater is protected by appropriate hydrological studies and selection of application rate. Where protection of groundwater cannot be assured, wastewater stabilization ponds should be considered for virus removal. Because of potential contamination of crops and infection of animals, complete removal of helminth eggs should be considered for slow-rate irrigation and sludge application sites. These relatively simple pretreatment requirements would be appropriate for many land treatment systems in the United States, e.g., for many slow-rate sites where crops for animal feed are grown. Pretreatment elimination of helminths from sludge is less simple, however.

This appraisal assumes only a minimum level of preapplication treatment for wastewater, i.e., properly-designed sedimentation. In situations with greater public access (e.g., water disposal on golf courses), shorter waiting periods before grazing or harvest of crops (e.g., agriculture in arid areas), or threat of groundwater contamination (e.g., shallow water table), more extensive preapplication treatment may be required. This treatment may consist of wastewater stabilization ponds, conventional treatment unit processes, or even disinfection. The exact degree of pretreatment required for these situations is site-specific, and recommendations should be determined separately for each system.

Because of the potential exposure to aerosolized viruses at land treatment sites, it would be prudent to limit public access to 100-200 m from a spray source. At this distance bacteria are also unlikely to pose a significant risk. Human exposure to pathogenic protozoa or helminth eggs through aerosols is unlikely.

Suppression of aerosol formation by the use of downward-directed, low-pressure nozzles, ridge-and-furrow irrigation, or drip irrigation is recommended where these application techniques are feasible. An important research need in this area is the comparison of the respiratory infective dose of enteric viruses with the oral infective dose.

The survival times of pathogens on soil and plants are summarized as follows. Since pathogens survive for a much longer time on soil than plants, the tentatively recommended waiting periods before harvest are based upon probable contamination with soil. What is a safe waiting period before crop harvest for human consumption is really an unsettled issue, and represents an important research need.

	Soil		Plants	
Pathogens	Absolute Maximum	Common Maximum	Absolute Maximum	Common Maximum
Bacteria	1 year	2 months	6 months	1 month
Viruses	6 months	3 months	2 months	1 month
Protozoa	10 days	2 days	5 days	2 days
Helminths	7 years	2 years	5 months	1 month

Aerial crops with little chance for contact with soil should not be harvested for human consumption for at least one month after the last wastewater or sludge application; subsurface and low-growing crops for human consumption should not be grown at a land application site for at least six months after last application. These waiting periods need not apply to the growth of crops for animal feed, however.

Properly designed slow-rate land treatment and sludge application sites probably pose little threat of bacterial or viral contamination of groundwater.

Considerable threat of bacterial contamination exists, however, at rapid-infiltration sites where the water table is shallow, particularly if the soil is porous. Likewise, considerable potential for viral contamination of groundwater exists at rapid-infiltration sites, and appropriate preapplication treatment or management techniques should be instituted, e.g., intermittent application of wastewater. Until then, groundwater drawn for use as potable water supplies should be disinfected. The factors controlling the passage of viruses through the soil into the groundwater, especially at rapid infiltration sites, is an important research area.

Human exposure to pathogenic protozoa or helminths through groundwater is unlikely.

There appears to be little danger of bacterial, viral, or protozoal disease to animals grazing at land treatment sites if grazing does not resume until four weeks after application. The risk to animal health from pathogens in sludge also is probably small (Yeager, 1980), but the issue is not settled. Removal of helminth eggs during preapplication treatment of wastewater should eliminate the potential of disease from those long-lived parasites, but the need for complete inactivation of helminths in sludge before land application is a research issue.

Because of the possibility of picking up an infection, it would be wise for humans to maintain a minimum amount of contact with an active land application site.

Epidemiological studies to date suggest little effect of land treatment on disease incidence. However, well planned and implemented prospective studies have not been completed, and many questions on the public health consequences of land application of wastewater and sludge remain (Larkin, 1982).

Research needs include the role of animals at land application sites in transmitting human diseases to animals or man off-site. Moreover, well-planned and funded prospective acute disease epidemiological studies at land treatment and sludge application sites involving a large number of exposed people should be completed.

ORGANICS

The potential health effects of toxic organic compounds are myriad. Systems affected range from the dermatological to the nervous to the subcellular, and effects produced range from rash to motor dysfunction to cancer. The degree of toxicity of organic compounds varies widely, from essentially harmless (e.g., most carbohydrates) to moderately toxic (e.g., most alcohols) to extremely toxic (e.g., dioxins).

A glance at the current edition of the Merck Index will reveal that the number of organic compounds described thus far is almost unfathomable. Almost any of these may appear in wastewater, depending upon its sources. Thus the discussion below must perforce be rather general, and the presence of any particular toxic organic in high concentration in wastewater or sludge may require a site-specific evaluation of potential health effects.

Most common organics in domestic wastewater derive from feces, urine, paper products, food wastes, detergents, and skin excretions and contaminants (from bathing). In medium-strength sewage (700 ppm total

solids content) organics make up about 75% of the suspended solids and about 40% of the filterable solids (colloidal and dissolved), consisting primarily of proteins (40-60%), carbohydrates (25-50%), and fats and oils (10%) (Metcalf and Eddy, 1972). After secondary treatment the more refractory and high-molecular weight organics predominate, e.g., fulvic acid, humic acid, and hymathomelanic acid (Chang and Page, 1978). In general, however, the chemical nature of domestic wastewater remains poorly characterized.

Although most of the organics found in domestic wastewater are probably harmless in a land application context, it has recently been found that fecal material itself commonly contains mutagens. Thus there is evidence that one of the causes of colorectal cancer is the presence of carcinogens or co-carcinogens produced by the bacterial degradation in the gut of bile acids or cholesterol (Thornton, 1981). The mutagenicity of feces can be increased by anaerobic incubation and by the presence of bile and bile acids (Van Tassell et al., 1982), and is lower in vegetarians than non-vegetarians (Kuhnlein et al., 1981). High levels of chromosome-breaking mutagenic activity have also been found in the feces of animals--dog, otter, gull, cow, horse, sheep, chicken, and goose (Stich et al., 1980). The chemical nature of the fecal mutagens is unknown. In the case of the latter animal mutagens, evidence suggests that at least part of the mutagenic action is due to hydrogen peroxide and the ensuing radicals which can be formed during oxidation of many organic compounds.

Ten domestic and industrial secondary effluents in Illinois were recently examined for mutagenicity by Johnston et al. (1982) with the results that all 10 effluents assayed showed significant mutagenicity. Mutagenic activity per unit volume of effluent varied over a 4,500-fold range, and toxicity varied over a 120-fold range. Selective extraction of whole effluents appeared to unmask mutagenic activity, probably by separating mutagens and substances that interfere with the mutagen assay. In several effluents there was evidence of several mutagenic compounds present, and it appeared that the mutagens were predominantly nonpolar, neutral compounds. There was no obvious influence of disinfection by chlorination on the effluent mutagenicity, in spite of the fact that one would expect many mutagens to be formed by the action of chlorine on humic substances and other organics found in wastewater.

Mutagenicity likewise has been observed in urban sludge (Hopke et al., 1982).

The major contributors of toxic organics to municipal wastewaters are usually assumed to be industrial discharges. However, household wastewater discharge may represent an important contributor since many consumer products in daily use contain toxic substances. A recent study (Hathaway, 1980) identified consumer products containing toxic compounds

on EPA's list of 129 priority pollutants, which may eventually end up in wastewaters. The most frequently used products are cleaning agents and cosmetics, containing solvents and heavy metals as main ingredients. Next are deoderizers and disinfectants, containing naphthalene, phenol, and chlorophenols. Discarded into wastewater infrequently, but in large volumes, are pesticides, laundry products, paint products, polishes, and preservatives. The organic priority pollutants most frequently used and discharged into domestic wastewater were predicted to be the following:

benzene	naphthalene
phenol	toluene
2,4,6-trichlorophenol	diethylphthalate
2-chlorophenol	dimethylphthalate
1,2-dichlorobenzene	trichloroethylene
1,4-dichlorobenzene	aldrin
1,1,1-trichloroethane	dieldrin

Thus many of the toxic organics, including mutagens (or potential carcinogens), to be found in wastewater and sludge come from within our own homes or even within our own bodies. This should be kept in mind when evaluating the public health effects of land application. The broad range of concentrations of toxic organics detected in municipal wastewaters and sludges (e.g., in the recent surveys of DeWalle et al. (1981) and Feiler (1980), however, suggests that wastewater or sludge applied to land should be regularly monitored for toxic organics. This measure is emphasized by the occasional discharge of toxic substances into municipal wastewater systems with resulting medical effects in treatment plant workers, such as the recent hexachlorocyclopentadiene episode in Louisville, Kentucky (Kominsky et al., 1980).

ORGANICS: Potential Exposure and Health Effects

While most of the nonvolatile organics in wastewater at a land treatment site would be expected to enter the soil, much of the volatile organics would probably be released to the air. The fraction entering the air would depend, of course, upon the processes, detention time, and application rates used at a particular facility. Although some of the volatile organics (and nonvolatile organics) may be released in the form of true aerosols, most would be expected to be released as gases by evaporation (volatilization).

At the Muskegon slow-rate land treatment system, the behavior of four of the most common toxic organics has been studied (Clark et al., 1981): trichloroethane, trichloroethylene, tetrachloroethylene, and chloroform (trichloromethane). Air stripping in the aerated lagoons was shown to be significant for these compounds. The maximum concentrations in air, together with the associated wastewater concentrations, of these

compounds immediately downwind from the aerated lagoons and spray irrigation rigs are shown in Table 9. All of these air concentrations are well below the 8-hour occupational standards of 45,000 $\mu g/m^3$, 535,000 $\mu g/m^3$, 670,000 $\mu g/m^3$, and 50,000 $\mu g/m^3$, respectively (ACHH, 1979).

Table 9. Air and wastewater concentrations at Muskegon Land Treatment System

	Aerated Lagoons		Spray Irrigation Rigs	
	Wastewater	Air	Wastewater	Air
	($\mu g/l$)	($\mu g/m^3$)	($\mu g/l$)	($\mu g/m^3$)
Trichloroethane	ND*	90	3	2.7
Trichloroethylene	118	73	68	9.3
Tetrachloroethylene	8.9	46	40	8.6
Chloroform	480	202	--	ND

*ND = not detected (<0.1 $\mu g/l$)

Comparison of the wastewater concentrations in Table 9 with the maximum values found in recent surveys of municipal wastewaters (DeWalle et al., 1981; Feiler, 1980) suggests that an increase of three orders of magnitude over the Muskegon values is the maximum that would ever be likely to occur. Even such an increase would probably still result in acceptable air concentrations.

Organic compounds in wastewater may be volatilized, immobilized by adsorption, or transported through the soil column, possibly to reach the groundwater. Adsorbed organics may be subsequently chemically or photochemically degraded, microbially decomposed, or desorbed. In view of the multitudinous variety of organic compounds existing, it is difficult to generalize about their biodegradation in soil. It appears, however, that most organics do become microbially decomposed in the soil, at least to some extent. This is especially true of naturally-occurring compounds, or those resembling them, because of the eons of evolution that have developed microbial enzyme systems to do the job. The more structurally complex the molecule is, e.g., condensed rings or dense branching, and more halogenated it is, the more difficult is biodegradation.

It has recently been found that it might be possible for carcinogenic and teratogenic nitrosamines to be formed from secondary and tertiary amines at land treatment sites. Thus, Thomas and Alexander (1981) have shown that dimethylamine and trimethylamine can be formed in municipal wastewater from naturally-occurring precursors. Dimethylamine may then go on to be microbially nitrosated, forming N-nitrosodimethylamine, a process which can occur under conditions resembling land

treatment of wastewater (Greene et al., 1981). Whether this actually occurs under field conditions, resulting in a threat to groundwater, is unknown.

Data on the transport of organics to groundwater have been presented by Bouwer and Rice (1978) and Tomson et al. (1981) for rapid infiltration sites, and by Jenkins and Palazzo (1981) and Demirjian et al. (1981) for slow rate sites. From these data it is evident that toxic organics can be transported to groundwater below land treatment sites, although the degree can be controlled by the level of preapplication treatment and application rate as well as choice of effluent and site characteristics. This certainly is cause for concern, but it should be kept in mind that groundwater is not the pristine substance it was once thought to be (Burmaster, 1982). The synthetic organic compounds most commonly found in groundwater in the U.S., deriving primarily from industrial wastes, are (Environmental Health Letter 21(6):7, 1982):

trichloroethylene	benzene
tetrachloroethylene	chlorobenzene
carbon tetrachloride	dichlorobenzene
1,1,1-trichloroethane	trichlorobenzene
1,2-dichloroethane	1,1-dichloroethylene
vinyl chloride	cis-1,2-dichloroethylene
methylene chloride	trans-1,2-dichloroethylene

Some biodegradation of organics occurs in groundwater (Gerba and McNabb, 1981), but it is difficult to avoid the conclusion that once toxic organics get into the groundwater they will remain there for a long time.

At the low concentrations found in the soil at land application sites, very few organic compounds are likely to be toxic to plants. These plants, although not injured themselves, may accumulate organics that may be toxic to the animals to which they are fed or to humans who use them as food, either directly or through animal products. The sparse data that exist thus far, e.g., that at Muskegon (Demirjian et al., 1981), however, suggest that plant uptake may be of little significance. Under certain site-specific conditions, however, high concentrations of particular organics in the wastewater or sludge may cause problems. For example, PCBs in cabbage grown on sludge-amended soil have probably caused degenerative changes in liver and thyroid of sheep (Kienholz, 1980), and one can extrapolate a similar phenomenon to a land treatment site.

Hansen et al. (1976) studied young swine fed for 56 days on corn grown on sludge-fertilized land. It was essentially a negative study: electroencephalograms, electrocardiograms, clinical chemistry, and

histopathology were all normal. However, they observed elevated levels of hepatic microsomal mixed function oxidase (MFO) activity in the swine fed sludge-fertilized corn. Associated with this were non-statistically significant increased liver weights. Other liver enzymes (alkaline phosphatase and lactate dehydrogenase) were normal. This increased MFO activity may be caused by toxic organics, metals, or be of no significance, but the authors concluded that further study should be performed before such grain can be recommended as the major dietary component for animals over long periods.

Similar results were found by Telford et al. (1982), who examined sheep fed silage corn grown on soil amended with municipal sludge at a high rate (224 t/ha). The sheep had significantly higher hepatic microsomal p-nitroanisole O-demethylase activity than controls, but no mutagenic responses for animal feed or feces, and no histopathological effects. In contrast, the same research group (Lisk et al., 1982) found no hepatic microsomal MFO response in swine fed corn grown on high-rate sludge-amended soil. Liver:body weight ratios, corn, feces, and urine mutagenicity, and histopathology were also unremarkable, suggesting absence or low levels of organic toxicants in the corn.

Studies at New Mexico State University (Smith, 1982), involving feeding of sludges to rats, sheep, and cattle, indicate no hazard from toxicants, based on uptake, MFO activity, and histopathology.

A more serious route of exposure by animals to toxic organics is the soil itself. Most grazing animals ingest a certain amount of soil together with their food plants. Thus dairy cows may ingest 100-500 kg of soil per year, with an average of about 200-300 kg/hr; expressed in other terms, dairy cows may consume soil up to 14% of dry matter intake when available forage is low and no supplemental feed is used (Kienholz, 1980; Fries, 1980). Lipophilic organics present in the soil may concentrate in animal fat. For example, feeding experiments with PCBs indicate that the steady-state milk fat concentrations are about five times the diet concentrations, which could result in milk fat levels of 0.7 ppm for each 1 ppm of PCBs in surface soil (Fries, 1980). Body fat levels would be expected to be similar. In a study of the pasture application of wastewater sludge with a high textile industry component, deHaan (1977) found 1.2 ppm of dieldrin (almost 19 times the acceptable level in The Netherlands) in the milk of grazing cows.

Turning to humans, Baker et al., (1980) described the metabolic consequences of exposure to high levels of PCBs from contaminated wastewater sludge used as a soil amendment in Bloomington, Indiana. No skin or systemic symptoms were noted, and of the hematologic, hepatic, and renal functions measured, only serum triglyceride levels increased, suggesting altered lipid metabolism. Serum PCB levels were normal. Naylor and Loehr (1982) have recently performed a detailed toxicological

analysis of the potential human health risks of the consumption of sludge-contaminated soils and crops associated with organic priority pollutants. They concluded that land application of sludge is not likely to result in the ingestion of amounts of organic priority pollutants exceeding the acceptable daily dose.

At the Whittier Narrows groundwater recharge project, secondary and tertiary effluent has been infiltrated into the Montebello Forebay area of Los Angeles County since 1962 (Nellor et al., 1982), resulting in long-term average exposures to humans of up to 11% reclaimed water in groundwater. Monitoring and epidemiological studies demonstrated that the recharge operation has not resulted in any adverse impact on the area's groundwater or the health of the population drinking the water.

Organics: Conclusions and Research Needs

The tremendous number of organic compounds possibly present in wastewater and sludge, together with their myriad health effects and poorly understood behavior in the environment, represent a considerable potential for adverse health effects. Most of these can probably be prevented by simple design and monitoring measures; this, of course, would not be true in the case of high discharges of particular chemicals.

Although removal rates of organics from wastewater by aerosolization and volatilization are high, exposure through this route is unlikely to present any significant health effect.

Toxic organics can enter the groundwater, particularly at rapid infiltration sites, and the application and soil factors controlling this transport, together with the factors governing their movement and decomposition within groundwater, are significant research needs.

The levels of toxic organics likely to be present in soils at land application sites will probably result in extremely low levels in aboveground portions of plants, but levels in roots, tubers, and bulbs may present a health hazard. The feeding of land application site-grown plants to animals is unlikely to pose a health problem, but grazing animals may accumulate significant levels of toxic organics. The issue of accumulation of organics from the soil by plants and animals (particularly into milk), and into the human food supply, is poorly understood, and more research is required.

TRACE ELEMENTS

The trace elements (including the "heavy metals") of public health concern in wastewater and sludge, i.e., those for which primary drinking water standards (U.S. EPA, 1977) exist (but excluding silver since its effect is largely cosmetic), are:

	Primary Drinking Water Standards (mg/l)
Arsenic (As)	0.05
Barium (Ba)	1.0
Cadmium (Cd)	0.010
Chromium (Cr)	0.05
Lead (Pb)	0.05
Mercury (Hg)	0.002
Selenium (Se)	0.01

Of these, cadmium, lead, and mercury are usually regarded as of most concern, and barium of minor concern. Chromium and selenium are essential elements in man; arsenic and cadmium have been tentatively shown to be essential to experimental animals, and thus may be essential to man as well (National Research Council, 1980). Secondary drinking water standards (U.S. EPA, 1979), i.e., those related to aesthetic quality, also exist for copper, iron, manganese, and zinc. These latter elements, as well as all other trace elements, are toxic if ingested or inhaled at high levels for long periods (Underwood, 1977), but this fact does not warrant considering them in the land application context, where low levels are expected.

Arsenic is popularly known as an acute poison, but chronic human exposure to low doses, as might be expected for all trace elements as a result of land application, may cause weakness, prostration, muscular aching, skin and mucosal changes, peripheral neuropathy, and linear pigmentation in the fingernails. Chronic arsenic intoxication may result in headache, drowsiness, confusion, and convulsions (Underwood, 1977). Epidemiological evidence has implicated arsenic as a carcinogen, but there is little evidence that arsenic compounds are carcinogenic in experimental animals (Sunderman, 1977). Even with high concentrations in soil, however, plants rarely take up enough of the element to constitute a risk to human health (Underwood, 1977; Council for Agricultural Science and Technology, 1976).

Barium has a low degree of toxicity by the oral route. Because of its effect of intensely stimulating smooth, striated, and cardiac muscle in acute exposure, however, it may have cardiovascular effects in low doses, but this has not thus far been demonstrated (Brenniman _et al._, 1979).

Cadmium is widely regarded as the trace element of most concern from a human health effect viewpoint in the land application of sludge, and this status probably carries over into the land treatment of wastewater as well. The critical health effect of chronic environmental exposure via ingestion is renal tubular damage due to accumulation of cadmium in the kidney. The initial consequence of this damage is the loss of low molecular weight serum proteins in the urine, followed by loss of other proteins, glucose, amino acids, calcium, and phosphate. This kidney damage is often irreversible and constitutes a significant adverse health effect (Ryan et al., 1982). There is evidence that the absorption and/or toxicity of cadmium are antagonized by zinc, selenium, iron, and calcium (Sandstead, 1977). The carcinogenicity of cadmium is controversial; the epidemiological evidence is tenuous, and the experimental evidence is conflicting (Ryan et al., 1982).

Chromium is much more toxic in its hexavalent form than its trivalent form, its predominant state in wastewater and soil. Chronic and exposure in experimental animals has been associated with growth depression, and liver and kidney damage (Underwood, 1977). Hexavalent chromium causes respiratory cancer upon chronic exposure to chromate dust (Sundermann, 1977). Most crops absorb relatively little chromium from the soil (Council for Agricultural Science and Technology, 1976).

Lead chronic toxicity is characterized by neurological defects, renal tubular dysfunction, and anemia. Damage to the central nervous system is common, especially in children, who have low lead tolerance, resulting in physical brain damage, behavioral problems, intellectual impairment, and hyperactivity. At soil pH above 5.5 and high labile phosphorus content, common conditions at a land application site, little movement of lead from the soil into plant tops and seed would be expected (Council for Agricultural Science and Technology, 1976; Stewart, 1979).

Mercury in low levels can result in neurological symptoms such as tremors, vertigo, irritability, and depression, as well as salivation, stomatitis, and diarrhea. Mercury can enter plants through the roots and appears to be readily translocated throughout the plant (Council for Agricultural Science and Technology, 1976), although there is some contrary evidence (Stewart, 1979).

Selenium exposure in its chronic form is associated with dental caries, jaundice, skin eruptions, chronic arthritis, abnormal finger- and toenails, and subcutaneous edema. It has also been found to have an inhibitory effect against several types of cancer (Fishbein, 1977). Selenium is readily taken up by plants and passed onto animals, and has caused toxicity in livestock in high-selenium soils (Council for Agricultural Science and Technology, 1976; Underwood, 1977).

Comparison of the ranges of levels of trace elements in untreated municipal wastewater in the U.S. (U.S. EPA, 1981; Feiler, 1980) with recommended irrigation water quality criteria (NAS-NAE, 1972) suggests that the trace elements in wastewater most likely to violate agricultural irrigation criteria are cadmium and chromium. The high levels of these two elements found in certain municipal wastewaters are doubtlessly due to industrial sources, for example, electroplating operations. Since chromium is poorly absorbed from the soil by crops, cadmium is probably the element of most public health concern.

Trace Elements: Potential Exposure

Cadmium is probably the element most likely to restrict the use of wastewater or sludge in land application where crops are grown for human consumption. In the case of crops not for human consumption, other elements may be limiting, for example, molybdenum because of its toxicity to livestock, and nickel because of its phytotoxicity (Page and Chang, 1981).

Trace elements, of course, are conservative materials, in contrast to pathogens and organics, which may become inactivated and decomposed. Thus, one would expect cadmium to build up in the soil at a land application site. Our major hope, from a health effects point of view, is that it will become immobilized in the soil, and be less available for plant uptake as time passes. There is some suggestion of this in Hinesly's data (Hinesly et al., 1979) on cadmium content of corn grain and leaves, grown during and after the termination of wastewater sludge application. The issues of cadmium uptake by plants often annual vs. cumulative application limits, and the effect of discontinuing application remain unsettled, however (Ryan et al., 1982).

Exposure of humans to trace elements through groundwater appears unlikely at slow-rate land treatment sites. At slow-rate sites both in New Hampshire (Jenkins and Palazzo, 1981) and Texas (Hossner et al., 1978) trace elements in the wastewater became stabilized near the surface.

Rapid-infiltration sites, in contrast, appear to present some threat to groundwater. After 30 years of primary wastewater infiltration at the Hollister, California, site, concentrations in shallow groundwater of cadmium (0.028 mg/l) and lead (0.09 mg/l) were above drinking water standards, but arsenic, barium, chromium, mercury, and selenium were below (Pound et al., 1978).

Sludge application sites may tie up trace elements similarly to slow rate treatment sites. Leachate from sludge-applied land in South Africa regularly had cadmium concentrations below the drinking water standard of 10 μg/l (Nell et al., 1981).

Trace Elements: Accumulation in Animals

The examination of trace elements in cattle grazed on sludge-amended pastures has revealed raised levels in liver and kidney, but not in muscle tissue (Bertrand et al., 1981a). No increases were seen when cattle were fed sludge-amended soil-grown forage sorghum (Bertrand et al., 1981b). Trace element levels and disease conditions of cattle grazing on land reclaimed by Chicago sludge have been observed by Fitzgerald (1978) for four years; it was concluded that little risk to man or animals is associated with land application of anaerobically digested wastewater sludge. Experience at Weribee Farm in Melbourne, Australia, where cattle are grazed on wastewater-irrigated pastures, has shown higher organ levels of cadmium and chromium than in farm cattle grazed on non-irrigated pastures, but comparable to non-farm cattle (Table 10). Organ levels of lead, however, did not increase, in spite of increases in both soil and pasture plants.

Table 10. Toxic trace element concentrations in cattle liver and kidney at Werribee Farm (μg/g dried tissue) (after Croxford, 1978)

	Cadmium	Chromium	Lead
Cattle Liver			
Non-Farm	0.76	0.05	0.5
Farm: Non-Irrigated	0.17	0.05	0.93
Farm: Irrigated	0.38	0.07	1.12
Cattle Kidney			
Non-Farm	3.32	0.06	0.32
Farm: Non-Irrigated	1.24	0.05	2.24
Farm: Irrigated	2.07	0.07	1.41

Since trace elements accumulate in very small quantities in animal muscle tissue, there is probably little concern about non-visceral meats in the marketplace. Liver and kidneys of animals do, however, accumulate high levels of cadmium, just as they do in man, so that these meats may be of concern to those people consuming large quantities of them.

Trace Elements: Human Accumulation of Cadmium and Health Effects

The significance of this concern with cadmium getting into the human food chain depends upon the cadmium levels presently existing in human food, the total dietary intake of cadmium, and the potential increase in cadmium levels in human food due to land application. (Drinking water and ambient air contribute relatively little to total daily cadmium intake (Pahren et al., 1979).)

The cadmium levels presently existing in human food can be estimated, at least for the U.S., by data from the U.S. Food and Drug Administration's Compliance Program ("market-basket survey"). These levels, together with the calculated normal dietary intake and vegetarian dietary intake of cadmium, are summarized in Table 11. It should be noted that root and leafy vegetables have the highest concentrations of cadmium. More accurate estimates of cadmium (and other trace element) concentrations in crops grown in the U.S., together with concentrations in the soils in which they are growing, will be available from a survey jointly supported by the U.S. EPA, U.S. FDA, and U.S. Department of Agriculture. In this survey 6,000 crop samples and 18,000 soil samples are being analyzed over a 4-year period, and the results should be available in 1983.

The present total dietary intake of cadmium was estimated above to be about 28 μg/day. Other estimates based on the market basket method have resulted in higher values: 26-61 μg/day in 15-20-year-old U.S. males, by the U.S. FDA, and 52 μg/day in Canadians (Kirkpatrick and Coffin, 1977).

A more direct, and potentially more accurate, method of estimating dietary intake of cadmium is by measuring the cadmium content of human feces. This method is feasible because the absorption of cadmium from the gut is low--rarely more than 10%, and usually 4-6%--and the excretion of cadmium into the gut is also very low. A recent study, using existing fecal cadmium data in Chicago and Dallas, and estimating daily feces production, resulted in a final estimate of the average daily intake of cadmium in food for U.S. inhabitants of 13-16 μg/day (Kowal et al., 1979). (Since the ingestion rate of the teenage male is often used in discussion of cadmium intake, values of 24 μg/day, 19 μg/day, and 18 μg/day for 10-19-year-old males from Chicago 1974, Chicago 1976, and Dallas, respectively, were estimated.)

This estimate of the average daily intake of cadmium in food can be compared with other estimates by the fecal analysis method, where the daily feces production of each individual was measured rather than estimated. In Sweden rates of 16 μg/day in nonsmokers and 19 μg/day in

Table 11. Cadmium concentration in foods and calculated dietary intake (from Ryan *et al.*, 1982)

Food Classes	ppb Cd[a]	Normal Diet[b] g/day	Normal Diet[b] μg Cd/day	Vegetarian Diet[c] g/day	Vegetarian Diet[c] μg Cd/day
Dairy products	5.7	549	3.1	584	3.3
Meat, fish, poultry	15.3	204	3.1	-	-
Grain and cereal products	23.2	331	7.7	203	4.7
Potatoes	48.0	138	6.6	43	2.1
Leafy vegetables	40.5	42	1.7	252	10.2
Legume vegetables	6.2	51	0.3	166	1.0
Root vegetables	32.3	25	0.8		
Garden fruits	14.7	69	1.0		
Fruits	3.0	173	0.5	284	0.8
Oils, fats, shortenings	15.3	56	0.9	107	1.6
Sugars and adjuncts	10.0	65	0.7	110	1.1
Beverages	3.0	534	1.6	600	1.8
		2,237	28.0	2,349	26.6

[a]From FDA Compliance Program Evaluation 1974 Total Diet Studies

[b]Adjusted on a caloric basis from FDA 1974 Total Diet Studies to represent the normal diet which compares with the adult lacto-ovo-vegetarian diet.

[c]Loma Linda lacto-ovo-vegetarian diet. Based on response of 183 southern Californians in a food frequency questionnaire by the Department of Biostatistics and Epidemiology, Loma Linda University School of Health, 1978. Leafy vegetables class includes root vegetable and garden fruit classes from normal diet.

smokers (former and present) have been reported (see Kowal et al., 1979, for references). The increased rate for smokers was partly attributed to their increased food intake. Nine µg/day fecal cadmium has been measured in Sweden, compared with a value of 10 µg/day measured by the total diet collection method. In Germany 31 µg/day has been measured, compared with 48 µg/day measured by the market basket method. In Japan, where cadmium levels in food are higher, the fecal analysis method has resulted in several estimates ranging from 24 µg/day to 84 µg/day.

> It has generally been concluded that ingestion of 200-350 µg Cd/day over a 50-year exposure period is a reasonable estimate for individuals (excluding smokers and occupationally exposed) within the population to reach the critical renal concentration (200 µg Cd/g wet weight in the renal cortex) associated with the initiation of proteinuria. This ingestion limit assumes background exposure levels of air and no exposure from smoking. If these exposures are increased, then the suggested ingestion limit must be correspondingly reduced. Smoking one pack of cigarettes/day will reduce the limit by about 25 µg/day. Again, these exposures are assumed to occur over a 50-year exposure period and, in the case of cigarettes, since many smokers start as teenagers, this addition would be relevant for much (30 to 35 years) of the 50-year exposure period. Therefore, smokers must be considered as being at increased risk. (Ryan et al., 1982)

The issue of cadmium in tobacco is particularly significant since it has been recently found that growing tobacco on soils, amended with municipal sludges at very high rates (224 t/ha) can result in a 30-fold increase in the cadmium content of cigarette smoke (Gutenmann, et al., 1982).

Thus present levels of total dietary intake of cadmium for most people appear to be fairly safe. However, in view of human variability in sensitivity and the variability in food supply, these levels probably should not be allowed to rise greatly.

It is of interest to note that increased consumption by individuals of those leafy and root vegetable crops highest in cadmium, and of organ meats as well, would increase the dietary iron intake. Since iron-sufficient humans absorb only about 2.3% of dietary cadmium compared to an average absorption of 4.6% (in the generally iron-deficient American population) (Flanagan et al., 1978), the increased iron intake would tend to correct for the increased cadmium intake.

Trace Elements: Conclusions and Research Needs

It seems reasonable to conclude that cadmium is the only trace element likely to be of health concern to humans as a result of land application of wastewater or sludge, with the exposure being through food plants or organ meats. Groundwater is unlikely to represent a threat except at rapid infiltration sites.

The potential increase in cadmium levels in human food due to land application of wastewater or sludge is still an unsettled question. Almost all the relevant research on the subject has been done with the land application of wastewater sludge, but, in spite of the high cadmium-application rates associated with this practice, insufficient time has elapsed to allow many firm conclusions to be drawn. It is clear, however, that increased cadmium in the soil results in increased cadmium in the plants grown in that soil, the degree of increase being a function of cadmium amendment, plant species and cultivar, soil pH, organic matter, and time since application (Ryan et al., 1982). Some are optimistic. Thus, Davis and Coker (1980) made an extensive review of cadmium in agriculture, particularly the potential transfer of cadmium from wastewater sludge into the human food chain. It was concluded that when sludge is applied to farmland in accordance with current practice, the hazard attributable to possible effects of the cadmium in the sludge on crops, animals, or the human food chain, is negligible. The degree of risk to man, of course, is dependent upon the amount of the food supply affected and the diet selection of the individual.

The most significant research needs in the area of trace elements continues to be the development of an understanding of the factors controlling the uptake of trace elements by plant crops at land application sites, their entry into the human food supply, and their effect on a variable human population. Assessments of the risk of exposure to lead and mercury should be performed, similar to that recently done for cadmium (Ryan et al., 1982).

NITRATES

Nitrogenous wastes are important constituents of municipal wastewaters, consisting of (1) proteins and other nitrogenous organics from feces, food wastes, etc., (2) urea from urine, and (3) their breakdown products. Bacteria rapidly decompose most forms of organic nitrogen to ammonium (or ammonia) in wastewater or soil. Under aerobic conditions ammonium is oxidized by bacteria (Nitrosomonas) to nitrite, and the nitrite rapidly oxidized by bacteria (Nitrobacter) to nitrate; the two-step process is called "nitrification." Under anaerobic conditions, and in the presence of organic matter, bacteria can use nitrate as a source of oxygen, and convert nitrate to molecular nitrogen, which escapes to

the atmosphere; this is called "denitrification." Both aquatic and terrestrial plants can use ammonium and nitrate as a nitrogen source.

Inorganic nitrogen is normally quite innocuous from a human health point of view, although high ammonia levels can present an aesthetic problem. The major health concern is that infants, less than about three months of age and consuming large quantities of high-nitrate drinking water through prepared formula, have a high risk of developing methemoglobinemia. The incompletely developed capacity to secrete gastric acid in the infant allows the gastric pH to rise sufficiently to encourage the growth of bacteria which reduce nitrate to nitrite in the upper gastrointestinal tract. The nitrite is absorbed into the bloodstream, and oxidizes the ferrous iron in hemoglobin to the ferric state, in which form it is incapable of carrying oxygen. Fetal hemoglobin (HbF), 50-85% of total hemoglobin at birth, is particularly susceptible to this transformation to methemoglobin. Methemoglobin is normally present in the erythrocytes of adults at a concentration of about 1% of total hemoglobin, being formed by numerous agents, but kept to a low level by the methemoglobin reductase enzyme system. This enzyme system is normally not completely developed in young infants. At a methemoglobin concentration of about 5-10% of total hemoglobin the body's oxygen deficit results in clinically-detectable cyanosis. As a result of epidemiological and clinical studies (Shuval and Gruener, 1977; Craun et al., 1981; Fraser and Chilvers, 1981) a primary drinking water standard of 10 mg/l of nitrate-nitrogen (i.e., nitrate expressed as N) has been established (U.S. EPA, 1977) to prevent this condition from developing.

Besides methemoglobinemia, there is also some concern about nitrates resulting in the formation of carcinogenic N-nitroso compounds in the gut, but this phenomenon probably involves higher concentrations than the 10 mg/l water standard (Fraser et al., 1980; Fraser and Chilvers, 1981).

The relevance of land application, of course, centers on the possibility of highly soluble nitrates reaching groundwater which may be used as a potable water supply. In land treatment, the threat again is most likely at rapid-infiltration sites. In the case of land application of sludge, there would probably be minimal threat if the sludge were applied at nitrogen rates, but higher rates or application outside of seasons of nitrogen uptake might be a hazard.

It should be kept in mind that land application sites are not the only source of nitrate in groundwater. Many groundwaters are naturally high in nitrates and in urban areas on-site absorption fields and lawn fertilizers have been shown to be sources of nitrates in groundwaters (Porter, 1980).

In conclusion, land application systems, particularly rapid-infiltration land treatment, threaten to raise the nitrate concentration in their underlying groundwater above the drinking water standard of 10 mg/l as N. This can be prevented, however, by proper siting and management practices, e.g., using high C:N ratio wastewater, matching loading rate to crop uptake, and optimizing the flooding-drying regime. These management practices and the agronomic factors controlling the entrance of nitrates into groundwater are important research needs.

SODIUM

Sodium may enter the groundwater beneath land application sites, just as nitrogen does. For example, at the Wrocław, Poland, sewage farm, an untreated wastewater sodium content of 83-115 mg/l was reflected in a groundwater content of 90-144 mg/l (Cebula and Kutera, 1978). This level may be compared with the sodium content of U.S. drinking waters, 58% of which have 0-20 mg/l, and only 14% of which have over 100 mg/l (White et al., 1967). A primary drinking water standard for sodium has not been established (U.S. EPA, 1977).

The health significance of these raised levels in groundwater, and thus potentially in drinking water, is unclear, particularly since drinking water sodium is a small portion of total dietary intake (2-8 g/day). That a decreased intake of sodium can reduce blood pressure in subjects with mild hypertension has been shown by several clinical studies, e.g., Beard et al. (1982), but the overall significance of restricted sodium consumption to preventing hypertension in the general public has been recently challenged (Kolata, 1982; Boffey, 1982; Puska et al., 1983). Similarly, the evidence that high sodium consumption causes hypertension is tenuous. However, increased groundwater sodium concentrations beneath land treatment sites should be kept in mind as a possible future health concern.

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QUESTIONS AND COMMENTS FROM THE FLOOR:

William Davis, City of Long Beach: 1. Has any of the research and study over the past 10 years identified a specific instance of a public health problem caused by land application of sludge in any form? 2. Has any of the research and study over the past 10 years identified a specific instance of a public health problem caused by land application of wastewater in any form?

Norman Kowal: To our knowledge, no.

C. Richard Dorn, Ohio State University: I can add some information based upon preliminary analysis of data collected in the Ohio Farm Bureau-Ohio State University project studying health effects of municipal sewage sludge on farm land. We have not observed any specific episode of human illness that appeared to be associated with sludge application on 47 sludge-receiving farms in the study. Furthermore, the illness rates on these farms were no higher than the illness rates on 46 control farms. In the laboratory of my project colleague, Dr. Abramo Ottolenghi, and in my laboratory, we have examined the sludge applied and human and animal feces samples from sludge and control farms for Salmonella. We isolated Salmonella from the sludge, people, and animals (both from sludge and control farms). However, there was no difference in the frequency of isolation from sludge and control farms. In addition, these types of the Salmonella isolates did not match nor was there any apparent temporal association with sludge applications and isolates of Salmonella on the sludge receiving-farms. It must be emphasized that we were using low application rates of 2-6 dry tons per acre. Dr. Kowal, in regard to the conflicting negative and positive observations of association between Salmonella in sludge applied to farm land in various European countriesand infections by the same Salmonella serotype in animals, do you have any information about the sludge application rates used in the various countries and could this explain the conflicting observations?

Norman Kowal: I am familiar with the conflict, but have seen no data on application rates.

WORKSHOP FINDINGS

WORKSHOP FINDINGS

Political and Institutional Constraints

William Davis, Moderator
Richard White, Co-Moderator
Lee Christensen, Secretary
Hunter Follett, Co-Secretary

Participants: Henry H. Benjes, Jr., James Butler, Charles M. Cameron, Jr., Leon Chesnisn, Michael Conti, David Cooper, Jack L. Cooper, Roger Dean, Lynn Forster, Henry Hyde, Cecil Lue-Hing, Robert Manson, Robert H. Miller, Merry L. Morris, Lois New, Robert C. Polta, William F. Pounds, Merilyn Reeves, B. R. Sabey, Charles M. Smith, Douglas Southgate, Charles Spooner

"Unless political and institutional constraints on the land application of effluents and sludges are recognized, identified, and resolved, these projects will likely fail, regardless of their technical, scientific and economic feasibility."

This conclusion of the 1973 Joint Conference on Recycling Municipal Sludges and Effluents on Land provided the starting point for the workshop discussion of political and institutional constraints. Goals of the workshop were to review progress to date and identify need for future action. Three fundamental questions were posed as the departure point for discussion: (a) How has research helped in reducing constraints? (b) Is information getting to those who must resolve constraints? (c) Can unqualified assurance be provided that a specific project will not cause harm to human health or environment?

Discussion was spirited and yielded the following generalizations:

1. Research conducted over the last 10 years has created a reservoir of professional expertise and information upon which to base state and local guidelines and for identification and the resolution concerns on the basis of fact. Research has also provided general guidance for land application programs by identification of design and operational constraints.

2. Information is generally flowing to members of the professional community. However, there is need to provide information to the public in an understandable form so the public may have a basis for considering the technical and scientific information that is presented.

3. Good management practices are a prerequisite in order to give assurances that, for a given situation, landspreading of effluents and sludges can be accomplished safely with minimal risk and attendant benefits.

4. Unqualified assurances of no risk cannot be provided. Such assurances won't be believed and represent an expectation that is not matched in other facets of society. Care should be taken to avoid being placed in the position of being asked to provide such assurances. Well designed programs to develop acceptance and avoid situations which generate antagonism among participants are extremely important in this regard.

FEDERAL REGULATIONS

The general consensus of the workshop participants was that federal regulations should "force" the evaluation of all reasonable alternatives for management of effluents and sludges. Such regulations should also provide for public notification of noncompliance with regulatory requirements and good management practices.

Federal regulations should provide guidance and be flexible but should not specify the design of a landspreading system. Federal regulations must not preclude the ability of states and local agencies to combine their unique social, economic, political, and physical factors in a way that is environmentally and technically sound and socially acceptable.

Minimum Standards

It was the consensus of the workshop that minimum standards are needed for the key health parameters. Minimum standards provide an independent benchmark for incorporation of necessary safeguards in projects and assurance that projects are safe for human health. Federal standards should, however, not preclude state initiatives towards more stringent standards.

The question of who should set minimal health standards was not resolved. Considerable discussion of this issue centered around two positions:

1. Federal regulations should contain minimal standards since some states are unwilling or incapable of setting such standards.

2. Standard setting should be delegated to state or local authorities within a broadly stated federal policy.

KEYS TO ACCEPTANCE

Acceptance of landspreading of effluents can best be achieved through well designed programs based upon education, communication, and planning. Elements of such a program are:

1. **Independent Resource Personnel**: Involvement of resource personnel that are not directly responsible for the project development process. For example, this could include university personnel from the subject matter areas of agriculture, engineering, forestry, economics, etc. Involvement of farm organizations, farm advisors, and local health officials should also be considered.

2. **Early and Continuing Public Involvement in Decision Making**: Public access to the decision making process is critical to development of public acceptance of a specific landspreading program. Such access must occur before decisions are made rather than after. In this regard, alternatives should be solicited and evaluated with interested and targeted audiences such as municipal officials, farmers receiving sludge, application site neighbors and consumers. Provisions must be made for feedback of the decision making so that the public may know how their comments have been considered.

3. **Education**: Efforts should be made to inform and educate all affected groups--for example, neighbors to the project site, the intended receiver or user of the effluent or sludge, consumers such as food processors, and municipalities as the generators of the material. Regulatory agencies need to improve understanding of the limits and freedoms of action. Public agencies need to improve their ability to market their services and products by better understanding the needs and desires of the participants in the project.

4. **Good Media Program:** Print, radio, and television media are available for use in communication and education programs. However, without proper care, adverse media coverage may result. Early involvement and education of the media regarding the issues involved and specifics of a project are recommended. Understanding and accommodating the scheduling and time information requirements of the media should also help to avoid adverse press. Also, as in the case of any communication, honesty of information and full disclosure are essential.

5. **Involve Local Elected Officials and Opinion Leaders:** Early involvement of local elected officials and key community leaders is vital to acceptance. Such persons should not learn of a landspreading proposal from the press or third parties but from those with the best information -- generally the project proponents and their resource personnel. It is important to understand these constituencies which local elected officials represent so that their concerns may be addressed in the development of a project. Also, timing of proposals during election campaigns are likely to generate modest levels of support or strong opposition from local elected officials and community leaders.

6. **Communication:** Careful attention to communication of information in a forthright and honest manner is critical. Efforts at technology transfer need to be improved so that information is provided in a manner which is usable by the public and farm community.

7. **Financing:** Federal financial support of research should be continued. Adequate funding of demonstration programs in conjunction with specific land application programs are an important aid in securing acceptance at the local level. Federal support of the USDA Cooperative Extension Service (CES) should be continued at levels which enable the CES to assist municipalities and the agricultural communities in developing and implementing land application programs. In view of the intense competition for the federal dollar, local and state means for financing research and implementation need to be developed. User fees collected by municipalities and revenue from sale of effluent or sludge products are possible revenue sources which should be explored.

INFORMATION AND RESEARCH NEEDS

Land application of wastewater and sludge has expanded over the past ten years due to regulatory, technological, and economic forces. Research has helped in the development of guidelines and provided a basis

for direction of specific land application programs. There are, however, several areas requiring additional information and research. Following is a summary of such needs identified through workshop discussion. No priority rating is assigned to the following items:

1. **Many municipalities are operating successful land application programs.** Summaries of these "case studies" should be written and widely distributed. Design criteria, management criteria, problems and their solutions as well as an evaluation of the public relations and educational program(s) used should be included.

2. **Lines of communication between university personnel, e.g., agricultural research and extension, and municipalities should be opened or expanded to enable university personnel to act as a resource for educational and public relations programs.**

3. **There is a need for a federal interagency task force to continue to address the question of the land application of sewage sludge.** The interagency task force should include at least the following: EPA, USDA, USA Corps of Engineers.

4. **Develop risk assessment methodology for comparison of alternatives.** Factors to be considered should include size of city, treatment processes, regional uniquenesses such as soils and climate, predominant agricultural land uses, etc.

5. **Continue to research the fate of pathogens and organics associated with land application of wastewater and sludge.** This information is needed to confirm past decisions and to reduce conservatism in design which should lower costs.

6. **Public acceptance of land application programs depends in part on the public's perception of the impacts of such programs on land values.** There is an involved need to evaluate the "conventional concepts" in public acceptance programs to see if in fact they contribute to or hinder acceptance.

QUESTIONS AND COMMENTS FROM THE FLOOR:

Curt Schmidt, SCS Engineers: I'd like to compliment Bill on a very fine presentation and ask him why in the world his group specifically said that no design manual was required. I certainly agree that it is presumptious to put out the design manual for large communities such as Los Angeles, New York, and so forth but those of us who are involved in putting a design manual together now, feel that one will be helpful to meet the needs of the smaller size community. I would like you to comment on that, please.

William Davis: The context of that statement was in the area of regulations, mandatory requirements, things which will ultimately get into the Federal Register. Whatever is published as regulation in the Register, no matter whether it is called guidelines, tend to become regulations. There is need for guidance. The Federal Register, however, is not the place to put a design manual. There is need for guidance and design manuals apart from the regulatory process.

Dr. Richard White, Ohio State University: Let me add a few comments to Mr. Davis's. My understanding of the reason for not requiring an extensive design manual is that it is a backlash to the design manual that is in the process or has come out on the subject of hazardous waste. In that context, it should not be a "federal design manual" which just sits on the shelf. This isn't saying that there should not be design criteria. The need for a design manual is probably more on the state level which would be more realistic for a region as opposed to a national design manual that would try to be all encompassing and then really not be functional.

William Davis: It comes out of the concern for flexibility and a desire to have flexibility to accommodate the local and specific circumstances. Are there any other questions?

Steve Campbell, BIOGRO Systems of Annapolis, Maryland: It appears that success of land application programs is directly proportional to involvement of universities - especially extension branches. How does one generate interest and support of the University?

William Davis: One of the findings of the workshop was that the involvement of universities, extension services, etc., is important. We did note that in some cases, because of budgetary constraints and lack of interest, personnel, some universities aren't responsive.

Dr. Richard White, Ohio State University: In part, I think one of the keys to getting university personnel involved is to identify an individual who has an interest with regard to the land application of sludge--whether it be an engineer or a soil scientist or agronomist. One would need to cultivate a relationship with that individual because there are so many pressures and priorities. Rarely will you get something coming down from the top, that is to say from a Dean or a Director. I think the key would be to begin more with a "grassroots" individual who recognizes the importance and is willing to put some time on the problem. I think one key would be--and we discussed it last night--that whether from a municipal source or from a private contractor if some dollars are put up to support a graduate student for a year or two it will develop interest on the part of faculty members quicker than anything else. And it could be a small amount--to pay for the support of one graduate student.

William Davis: Hunter Follett would you care to comment on your efforts to obtain support along the lines suggested?

Hunter Follett, Colorado State University: I think the example that Bill wants me to comment about is the cooperative research project between Colorado State University and the Littleton/Englewood Wastewater Treatment Plant. The treatment plant is supporting a demonstration-research project relative to the application of sludge to dryland wheat. They are providing financial support for a graduate student and to carry on a research program. This support will be continued for four years and is to be renewed on an annual basis. We have started at two different locations, and are experimenting with the land application of sludge to dryland wheat plots using the same techniques, the same equipment, and the same truck that they use to deliver the sludge to the fields. We have put some experimental plots out in that area where they are actually applying the sludge to land. We plan to collect information to develop a research base and to develop an educational program to assist the municipality. We can then use the information to gain support from the farmers in that area. I think this has been a successful approach to getting financial support from a municipality for this type of work.

William Davis: This is probably something I went over very lightly. It is important to have a piece of ground somewhere nearby where the activity is actually being done on a demonstration basis and where data and information are being gathered that are applicable to the project area.

Hunter Follett: I should mention that we have cooperation from the company that is applying the sludge, the county agent, the farmer and the wastewater treatment plant. Therefore, this has been a joint effort from several different organizations.

A. L. Page, University of California: If I may, it appears that there may be a deficiency in the avenues of communication between the various muncipalities and various colleges and universities in the local area. Possibly this is something that could be greatly improved through some mechanism associated with institutional constraints and we may want to include that as an item which would deserve some very serious consideration.

Kenneth Dotson, EPA, Cincinnati: Involvement of farm organizations can encourage state university, College of Agriculture staff to be involved in educational as well as research programs concerning sludge use on land. Please clarify what is meant by conservatism of regulations pertaining to organics and pathogens associated with land treatment of wastewater and/or sludge. Is the conservatism alluded to based on an estimation of the ability of a chemical to produce an adverse health effect or does it arise from the models used to estimate the exposure of man to these agents?

William Davis: Let me give the context as we now see it. In the programs that are going on now, decisions are being made as to what to do relative to concerns involving pathogens and trace organics. There was a feeling that the data base could be improved. Not in modeling efforts but in the information going to the field. When the decision is made to do a project you can only work with whatever data that is there. There is a tendency to build in safety factors on top of whatever the research indicates and designers tend to back farther away if the data base is weak. What is being addressed here is not a particular model but the situation where people that are dealing with the problem in the field in terms of putting projects in and building conservatism into their design.

William Pounds, Pennsylvania Department of Environmental Resources: Not necessarily would I respond totally to that question but I think what we tried to do in Pennsylvania is to use management technology for farming sites and include certain requirements that limit the amount of sludge you are allowed to apply to land, limit sludges to those that are digested or lime stabilized, require incorporation of sludge within certain time periods, etc. In addition to this, we also require that no cattle be grazed for a period of two months on pasture where sludge has been applied. In establishing this program we are not just "shooting from the hip", but were using the best available technology at the time. What I would like to see is support for the statements I have heard from Lee Sommers, and yesterday from Michael Overcash, which indicated that apparently we made the right decision. On occasion, we have made judgments sometimes "by the seat of our pants", but always based on the best available research, and from what I have heard the last two days, apparently we made the correct decisions. I think the statement can be

made that if a site is properly selected and managed with limited applications of sludge, there is a minimum impact from organics. That statement was made, but not strongly reinforced. I think from the quorum such as we have at this workshop, we can make such a statement in the summary. It would go a long way in helping us in terms of institutional and attitudinal constraints.

Jack Cooper, Director, Environmental Affairs, National Food Processors Association: On the subject of political and institutional constraints, our association has a policy on the use of municipal sewage sludge in the production of foods for human consumption. The association represents about 600 companies that pack processed-prepared fruits, vegetables, meats, soups, baby foods, fish and specialty products, including canned, frozen, dehydrated, pickled and other preserved food items. The Association provides extensive technical support to its members through its three laboratories located in Washington, D.C, Berkeley, California, and Seattle, Washington.

In 1981, the NFPA Agricultural and Environmental Committees reviewed the fact that increasing volumes of sewage sludge were being applied to croplands and that crops for human consumption were being grown in sewage sludge-amended soils. These committees also reviewed available federal regulations and guidance. In November 1981, the NFPA Board of Directors adopted the following policy statement:

> *The National Food Processors Association (NFPA) recognizes that the application of sewage sludges generated by publicly owned treatment works (POTWs) to croplands can be a beneficial use when the proper conditions exist and the sludges have been determined to be safe for the use intended. The NFPA, however, has serious concern that sludges produced by POTWs may contain heavy metals, human pathogens, and toxic compounds which could have potential health effects on the consumers of foods produced on lands to which such sludges have been applied. This potential risk to public health has not been adequately evaluated. Therefore, NFPA cannot endorse the application of municipal sludge to land that may or will be used in the production of foods for human consumption until the questions regarding its safety for such use are resolved. Therefore, NFPA urges the EPA, USDA, and FDA to establish an interagency task force to develop definitive policies and regulations which identify the specific conditions under which municipal sludges can be safely applied to land on which foods for human consumption will be produced. Furthermore, NFPA recommends that the interagency task force seek the advice and counsel of NFPA and other groups representing agriculture, industrial, and public interests in the development of appropriate policies and regulations.*

> *In late 1982, NFPA sent to each of its members a letter providing a copy of the EPA 1979 regulations, the EPA/FDA/USDA policy statement, and background information. The letter advised each company to*

review the subject with their own legal counsel. While the Association's position doesn't endorse sludge use in crop production, it certainly doesn't prohibit it. It is up to each company to establish whatever restrictions, if any, that it feels are in its best interest.

Unfortunately there are, and probably always will be, unresolved public health concerns about sewage sludge use in crop production that will prevent NFPA from changing its basic policy. However, on a site-by-site, case-by-case basis, if our individual members are brought into the decision-making process sufficiently early so that they can draw their own conclusions about the safety of an individual operation, and make recommendations for change, were necessary, local resistance from this source may be overcome.

I am encouraged today, where I have heard over and over again the statment that "well managed" projects will not harm public health or the environment.

Dale Baker, Pennsylvania State University and also Chairman of the Northeast Regional Task Force: It is extremely important that we have concerted efforts among experiment station personnel, regulatory people and operations people in the management of sewage sludge and effluent on land. The Directors of the Northeast experiment stations recognized that when they authorized our Task Force to develop a document on Criteria and Recommendations for Land Application of Sludges in the Northeast. *The Northeast Regional document in itself is part of an experiment. It is one way of getting the scientists to make a concerted effort to reach agreement on what is possible in the land application of sludges under different situations. Maybe, at least it's leaked back to me, some people consider the Northeast document to be extremely conservative. It is, however, somewhat liberal with respect to sludge use for land reclamation and land disposal of sludges at dedicated sites. This experiment, in the end, may point to the direction that all other regions will want to take. I am very proud of the fact that we have had people that are very conservative and who have very real concerns about the use of sludge on land working with others who have had experience with sludge on land. It has been a rewarding experience to get those people together educators and researchers) to develop a document that has about 30 authors from 14 states involved. The experiment station record will in the end have to take credit for allowing the group to come up with the document. The main objective is the development of an operational program for every state in the Northeast. If we can encourage uniformity among states, I believe it will be beneficial. It will maximize the utilization of sludges on land because uniform recommendations let everyone know where they stand. No project appears to be an exception, and that is not true if you don't have a concerted effort. I can't speak for the Experiment Station Directors, but the document is near completion; the experiment is in place and I am happy with the results.*

Engineering Systems

Raymond Loehr, Moderator
Sherwood Reed, Co-Moderator
Takashi Asano, Secretary
James Parr, Co-Secretary

Participants: J. W. Biggar, Patrick L. Burke, Stephen Campbell, C. E. Clapp, Donald Deemer, H. A. Elliott, Eliot Epstein, Joseph Farrell, Ronald F. Follett, Craig G. Hebberd, Frank Humenik, Richard Johnson, C. James Martel, James Patterson, Charles Pound, William Rosenkranz, Curtis Schmidt, E. D. Smith, J. L. Smith, Robert G. Smith, Stephen B. Smith, George Tchobanoglous, Dale H. Vanderholm, Jack L. Witherow

INTRODUCTION

The 1972 Amendments to the Federal Water Pollution Conrol Act (PL 92-500) provided the requirement and the incentives to utilize land application of municipal wastewater and sludges. The 1973 workshop defined the research needs for implementation of these concepts.

In the decade since 1973 several thousand systems in the United States are now used to apply wastewater to land in a controlled and managed manner; in the next decade at least another 1000 systems are expected to be designed and installed. Also, there are many systems applying sludge to the land. In 1978, it was estimated that at least 1.0 million dry tons of sludge were being applied to U.S. land or about 25% of the total sludge requiring disposal. The use of land treatment systems is expected to increase as a result of ever-improving research and from the experience gained in operating existing systems.

The research and development efforts of the past decade have clearly established land treatment as a viable cost effective technology using both wastewater and sludge. Now that adequate information is available, the engineering profession routinely consider the concepts of land treatment during design and planning stages with a high level of confidence that the required performance goals can be achieved.

Similarly, sound management and operational procedures for land treatment systems have grown steadily, and is regularly translated into improved environmentally sound and cost-effective land treatment systems. In use in all climates and geographical regions land treatment offers a level of dependibility not found in alternative wastewater treatment and sludge management systems. The safety factor inherent in land treatment system design provide a reserve capacity to accommodate minor design problems, as well as changes in sludge or wastewater characteristics.

Waste treatment, rather than optimum crop production, is the primary goal of land treatment systems, i.e., to treat wastewaters and manage sludges in a manner that insures compliance with all environmental requirements and regulations.

The participants in this workshop came to a consensus about needs presented in the overview paper, *Engineering and Economics* (Pound, Crites and Griffes). This summary focuses on the major needs and recommendations that relate to improved and cost-effective design of land treatment systems. The overview paper was consulted to identify other more detailed needs and recommendations pertinent to specific interests and situations. Two important areas covered by the overview were:

(a) the application of sludge to disturbed lands and the application of wastewater and sludge to forest land were promising alternatives that deserved further attention and evaluation.

(b) the design of land treatment systems should include adequate sediment and erosion control measures.

Although significant advances in design knowledge have been made since the 1973 workshop, progress is needed to make land treatment systems more effective and less costly.

ASSESSMENT OF ACCOMPLISHMENTS SINCE 1973

There has been considerable progress with regard to the engineering design of land treatment and sludge application systems in the past decade. Based upon the experience gained in this period, the following conclusions can be made:

1. **Land treatment is a component in a total treatment system and is not just a disposal process.** Recognition must be made of the pollutant removals that occur in the soil-plant ecosystem since it is an integral part of the total treatment system. Such recognition encompasses: (a) rational engineering design that includes appropriate credit for pollutant removal in the soil-plant ecosystem, and (b) the development of a least cost waste management system.

2. **Experience since 1973 has shown that when land treatment systems are properly designed and operated, the reliability and treatment performance of these systems is equal to or better than that of conventional wastewater treatment systems.**

3. **Experience has shown that when sites for the land application of sludge are properly designed and managed, reliability and performance is equal to or better than that of other sludge management systems.**

4. **The state-of-the-art engineering design of land treatment systems developed since 1973, is adequate to meet the designated water quality criteria.**

NEEDS AND RECOMMENDATIONS

Much useful information has been obtained during the past decade to resolve problems faced by the engineering community in the planning and design of wastewater treatment and sludge management systems. There continues to be a need for additional information to improve and optimize land based wastewater treatment and sludge management systems to meet specific performance goals.

High Priority

1. **A major emphasis should be placed on post-construction performance evaluations and cost documentation of existing land treatment systems.** The original design criteria should be evaluated in terms of actual performance of the existing systems. Modification of the design criteria should be made where appropriate.

A common protocol should be developed for post-construction performance evaluations. The evaluation should assess: (a) the adequacy of the hydraulic application rates, (b) the transformations and fate of nitrogen, phosphorus, organics, and metals, (c) the adequacy of existing

monitoring requirements, (d) the actual cost of land treatment system components, and (e) changes in the land value of adjacent property.

These evaluations will identify: (a) an improved cost data base, (b) more appropriate monitoring requirements, and (c) improved design and operational criteria.

2. **Design criteria for application of wastewater and sludges to forest ecosystems are needed.** Forest lands have great potential to receive and accommodate wastewaters and sludges for soil improvement as well as for nutrient and water utilization by forest species. However, a limiting factor appears to be the lack of reliable and economical application methods. Information also should be obtained on appropriate organic and metal loading rates and their impact on the chemical and physical properties of forest soils.

3. **(a) Management techniques for optimum total nitrogen removal in the field for wastewater and sludge applications should be developed.**

(b) The understanding and modeling of nitrogen and phosphorus transformations and removals, including nitrogen mineralization rates, especially in field situations, should be improved. The development of improved and simple modeling techniques for estimating the rate and extent of nitrogen and phosphorus transformations and removal is desirable. Better management and modeling techniques would improve the design of less costly land treatment systems.

4. **The heavy metal loading criteria should be reassessed.** This reassessment is especially needed for western soils and for long-term loadings. Present criteria are considered by many to be conservative. Less conservative criteria could greatly improve the cost relationships of land treatment systems.

Land application of wastewater effluents and sludges can be limited by overly restrictive and unrealistic criteria which may effectively exclude land application from consideration as a viable option and may lead to the adoption of less environmentally acceptable alternatives.

5. **Specific research needs include: (a) determining the transformation and fate of trace organics in land application and treatment systems, (b) evaluation of overland flow systems in cold climates.**

QUESTIONS AND COMMMENTS FROM THE FLOOR:

Y. A. Demirijian, Muskegon, Michigan: Have you given consideration to the energy aspects of the design since land treatment systems can have a high energy requirement? Have you given consideration for energy conservation?

Raymond Loehr: The energy aspects were discussed and included in the needs and recommendations in an implicit rather than explicit manner. The energy concerns were included as part of the post-construction evaluation. It was expected that such evaluations would identify the energy demands at the land treatment sites and to try to quantify them.

Raymond F. Shipp, Pennsylvania State University: I am curious to know what the discussion was relative to the amount of pretreatment considered desirable for the different systems. Should disinfection be considered a requirement or not?

Raymond Loehr: This point was addressed by pointing out what the group felt about pretreatment requirements. These requirements are site specific and are related to the wastewater characteristics and objective of the system. Let's assume that the objective is to meet drinking water quality criteria. The approach would be to look at the site specific soil situations and see what removal takes place in the soil. With sandy soil you might have completely different pre-application needs than with clayey soils. The question does not revolve about the need for disinfection but about how to meet the objectives and avoid adverse impacts to the system components and equipment. The group discussion focused on whether there was adequate technical knowledge of how to develop a design given whatever the standards that were to be met.

To specifically answer your question, the discussion indicated that there is adequate disinfection technology that could be used where it is needed. The group did not get into the discussion of whether disinfection was needed prior land application but whether the adequate disinfection technology was available. If adequate bacterial removals occurred in the soil, disinfection should not be needed prior to land treatment.

Michael Connor, Harvard School of Public Health: I have two questions. 1. In the statement of goals to achieve groundwater quality standards, is the distinction between meeting drinking water standards and not affecting ambient water quality? For instance, many agricultural areas have ambient nitrate levels which do not meet drinking water standards 2. The public health workshop expressed concern over the potential

of rapid infiltration systems to leach organics and metals into groundwater. Did the workshop address the levels of pre-treatment necessary for rapid infiltration systems?

Raymond Loehr: There was some discussion related to the first question. The ambient water quality discussion related more to drier, arid climates where there might be groundwater which had high dissolved solids. There would be little logic in achieving drinking water quality from a land treatment system if it is likely that the water will not to be used for that purpose, or if it had to be treated before it could be used for drinking water.

The discussion also related to the technology to minimize excess nitrogen in the water that reached the groundwater or the system boundary. Identification of the system boundary may require further definition. However, the discussion indicated that in general there was probably adequate information on how to design land treatment systems so that the water that passed the system boundaries did not have excess nitrogen.

Michael Connor: The second question related to pretreatment requirements. There was concern expressed in the health group on rapid infiltration systems and the potential for organics in such systems to leach to groundwater. Did your group discuss this point?

Raymond Loehr: We discussed that point but again it was in the context mentioned earlier, i.e., whether or not there was adequate technology to protect the quality of the ultimate water use. If with rapid infiltration, there was significant movement of organics, there are technologies that could be used to minimize such problems. Preapplication technologies would be one approach.

A. L. Page, University of California: Ray, could you elaborate briefly on the discussion associated with the research recommendation relative to heavy metal loading and the comment you made that such loadings may be conservative with respect to the current guidelines and regulations.

Raymond Loehr: Our group continues to be guided by those who have spent considerable time evaluating the whole question of the heavy metals in the land, their reversibility, their uptake, and their fate. The item the group discussed was as stated earlier, based on the existing systems, concern that the heavy metal loading criteria might be too conservative, especially for some of the metals. If the metal criteria currently used

were conservative and were to be changed, land treatment systems could be more cost effective. We did not get into any detailed discussion as to whether they were or were not conservative, feeling that this topic would be discussed by other groups. Our discussion was a recognition of the fact there was a constraint on the system which did affect the cost effectiveness of the system and that if there were changes to relieve that constraint, such changes would have an impact on the cost effectiveness of the system. Hence, a reassessment might be in order.

William Davis, the City of Long Beach: That raises a question: In the last 10 years, we have conducted a lot of research and lot of data has indicated that adjustment of the regulations might improve the cost effectiveness of land application programs. Does the research conducted provide the basis to adjust the regulation?

Raymond Loehr: I'm the wrong one to answer that question. I assume that this could be addressed by the group will be discussing the heavy metal question this morning? Who is that?

A. L. Page: That would be Lee Sommers.

William Davis: The thought that occurred to me as a result of listening to some of the comments was that perhaps the cost effectiveness of these land application programs could be improved with some adjustment in the regulations. This leads me to the question--Has the research of the last 10 years provided the information that can be used to adjust or improve the regulations so we can improve the cost effectiveness and expand the capabilities of land application programs?

Lee Sommers, Purdue University: From the standpoint of trace metals?

William Davis: Yes, as well as chlorinated hydrocarbons, and nitrates.

Lee Sommers: In our group, we were totally concerned with sludge management practices in relation to crop production. We basically felt that there is sufficient information available to manage inputs of trace elements from sludge and effluents such that crop productivity would be maintained or improved and surface and groundwater would be protected from contamination. Although the current guidelines for trace element inputs that exist in different regions of the country do protect plants as far as yield are concerned, there was a general feeling that metal limits may be overly conservative. However, we don't really have the

information available right now to say how high limits could be without sacrificing crop production. This we identified as our overall research needs. We did not address the human health issue which is really the direct reason for the regulations on cadmium.

William Davis: You did not address the human health question?

Lee Sommers: We did not, because other workshop groups were concerned with the health effects.

William Davis: Could the moderator of the session which addressed health effects of Cd comment in terms of the adequacy of existing regulations, i.e., too conservative, too liberal?

A. L. Page: We have a workshop on organics and inorganics in relation to public health. Would Herbert Pahren, the moderator of this group, care to comment?

Herbert Pahren, EPA, Cincinnati: This will be covered in our report. I'd prefer to address it at that time.

Merilyn B. Reeves, League of Women Voters: In the recommendation that land treatment be considered a component of total effluent treatment was there discussion of how such a recommendation affects the cost effectiveness and competitiveness of land treatment over conventional treatment?

Raymond Loehr: The discussion did revolve around that particular point. But not as if land treatment was an add-on process. The key to decisions about preapplication treatment goes back to the question of the goal that is to be achieved. If a new system is being developed, you have the opportunity to start, with the raw wastewater and find a system to accomplish that goal and include land treatment as a major component of that system. If the system is in place and it is to be upgraded, land treatment might be an add-on. However, the basic question is how does land treatment help meet the pollution control or water quality objectives. Land treatment should not be considered only as an add-on component to do some polishing. Land treatment can be a major component of the entire treatment system and can accomplish considerable pollutant removal. The consensus of the group was that there is a lot of the technology which could be integrated with the land treatment component to achieve the goals identified earlier.

Donald Deemer, ERM, Inc., West Chester, PA: Let me also try to address that question. Emphasis was placed on the fact that land treatment is an important part of the total treatment system. If it is not so considered, i.e., if it is considered strictly as a disposal method, then the economics became prohibitive. The emphasis should be on the fact that the soil is a key component of the treatment system. The soil-plant system can treat the wastewater and the sludge. The system design and the review by the regulatory agency should reflect this fact. Under such conditions, land treatment can be shown to be cost effective when compared to conventional treatment systems. If land treatment is considered only as a disposal method that has to be preceeded by a conventional secondary treatment system, then land treatment rarely is competitive.

Management Considerations in Effluent Use

P. F. Pratt, Moderator
W. E. Sopper, Co-Moderator
Richard Thomas, Secretary
Lowell Leach, Co-Secretary

Participants: Donald D. Adrian, David Ariail, Charles R. Berry, Francis Broadbent, Ronald Crites, Y. A. Demirijian, Frank M. D'Itri, Harvey Doner, A. Erickson, Norman Evans, Herbert D. Grover, I. K. Iskandar, B. D. Knezek, C. R. Lee, Harvey Luce, L. J. Lund, Dale Nichols, Wade Nutter, Betty H. Olson, Antonio Palazzo, Albert T. Wallace, Joel Webster,Flora Mae Wellings

INTRODUCTION

This 1983 workshop was not restricted to management; design and application both crucial to successful land treatment systems, were also examined and discussed.

Besides assessing progress in wastewater and sludge processing on land since the 1973 workshop, the 1983 workshop identified those areas needing continued research. In many areas of concern in 1973, the 1983 conference deemed that sufficient information and data had been generated and made widely available in the last decade. Further work in these areas were assigned a low priority as a research need; although it is recognized that continuing experience in the field will add to the available information and data. Other areas of the three systems considered by this workshop were assigned "medium" or "high" priorities according to the workshop consensus.

For slow rate systems, the workshop concluded that adequate information was available for (1) crop selection, crop response, and crop management, (2) erosion control, (3) effluent loading rates, (4) retention of heavy metals, suspended solids and phosphorus, (5) decomposition of organic materials in effluents, and (6) quality criteria for inorganic materials in effluents used for irrigating crops.

For rapid infiltration systems, the workshop group agreed that there was adequate information on (1) characterization of soils, (2) retention

of suspended solids, (3) reduction in BOD, and (4) heavy metals. Although site selection procedures were not considered to be completely adequate, many mistakes with site selection were not considered to be due to lack of information or methodology, but rather improper application of methods and/or inadequate sampling in three dimensions.

For overland flow systems the workshop agreed that there was sufficient information on (1) site selection, (2) soils and landscapes, (3) fate of nitrogen, (4) loading rates, (5) erosion control, (6) adapted vegetative cover, and (7) heavy metals.

A general research recommendation, equally applicable to both slow rate and rapid infiltration systems, was that systems models be verified by field data to help maximize the efficiency of system management and operation. Also, the need for increased transfer of technology and for training of managers/operators of land treatment systems still exists. Specific research needs in all three systems are characterized below.

SLOW RATE SYSTEMS

High Priority

1. **Risk assessment of potential pathogenic organism problems related to land treatment systems as opposed to other methods of treating and disposing municipal wastewater.** If land treatment systems are used, a knowledge of how pathogens, particularly viruses, are transported in the soil and groundwater and what factors affect survival time is essential. A second need is determining the relative risk of transmission of respiratory viruses through wastewater aerosols compared to the relative risks associated with other alternatives of wastewater disposal such as direct discharge into lakes and streams.

2. **Comprehensive studies of ecosystem dynamics: a. Site aging and wastewater renovation efficacy.** Most land treatment systems are designed for a life of 20 or 30 years. As a cropland or forest land site ages, information is needed on how it changes in renovation efficiency. Most land treatment systems in the U.S. are new and little long-term data are available. The occurrence of significant changes in renovation efficiency, over time may necessitate changes in the way that a system is utilized.

b. **Ecosystem collapse and recovery:** In improperly operated land

treatment systems, the potential exists for ecosystem collapse as a result of overloading the biological system with excessive amounts of wastewater. Ecosystem collapse is defined as the state in which the ecosystem can no longer satisfactorily remove nitrogen and other pollutants in the wastewater. Few land treatment systems have supplemental areas large enough to be used should ecosystem collapse occur. Information, therefore, is needed concerning ways to accelerate ecosystem recovery and recovery time. In forest ecosystems, because of their continued recycling of nutrients and the less frequent removal of applied nutrients are compared to annual crop harvests in agronomic systems, such information is critical.

c. **Study of on-site ecosystem effects:** Comprehensive studies of the effects of wastewater irrigation on the entire ecosystem should include effects on primary and secondary productivity and on movement and distribution of elements throughout the entire soil-plant-animal complex.

Information on forest ecosystem stability is also needed.

d. **Study of off-site effects:** Identifying the effects of land treatment systems on neighboring ecosystems must include a determination of which elements are being transported and their subsequent effects on neighboring human and natural ecosystems.

3. **Study of nitrogen management in forest systems:** Because there are only a few forest slow rate systems in the U.S., there is a paucity of information available on nitrogen management. Unlike cropland systems which have an annual removal of nutrients in crop harvest, forest systems continuously recycle nutrients, often over the system's entire life. There is a need to develop successful management techniques to control nitrogen transformations (mineralization and immobilization, nitrification, and denitrification) and cycling.

4. **Prediction of nitrogen concentrations in groundwater leaving the land treatment system:** To refine and improve current methods being used for predicting average concentrations of nitrogen in the groundwater leaving a wastewater treatment site, there is a need for both additional data and the development of either better models or field verification and refinement of existing models.

Medium Priority

1. **Study of forest ecosystem responses to wastewater irrigation:** More information is needed for the effects of wastewater irrigation on the

various major U.S. forest types as a function of the type soil, geology, and climatic conditions. Such investigations might include tree growth, biomass production, physical and anatomical properties, forest floor accumulation and decomposition, physical and chemical changes in soil properties, seedling reproduction, mortality, soil invertebrates, and microbial communities.

Low Priority

1. **Study of trace organics:** There is a need for more information on trace organics in municipal wastewater and their ultimate fate on land treatment systems. Generally, the concentrations of organic compounds in municipal effluents will be low and have little effect on the design and management of slow-rate systems.

2. **Study of water infiltration and transmissivity:** There is a need for better field techniques to determine the infiltration and percolation of wastewater at potential land treatment sites. Such information is critical to the design of the system in terms of hydraulic loading and water budgets.

RAPID INFILTRATION SYSTEMS

High Priority

1. **Risk assessment for land treatment systems compared to that of alternative systems for the treatment and disposal of municipal wastewater.** For the successful application of land treatment systems, a knowledge of the flux of pathogens, particularly viruses in the soil and groundwater, must be developed for a comparison by risk assessment methodology with alternate disposal and/or treatment systems.

2. **Determination of the fate of trace organics:** Because of the speed of water flow through the soil in rapid infiltration systems, trace organics having relatively slow decomposition rates are likely to move with the wastewater streams into aquifers or into surface waters. The establishment of decomposition rates for various organics and the relative flux of organics versus that of water would help explain potential trace element problems and help evaluate the suitability of effluents for rapid infiltration systems dependent upon the specific trace organics present.

Medium Priority

1. **Management of land surfaces devoted to rapid infiltration:** Applied research dealing with management of vegetative cover, wetting and drying cycles, disposal of surface accumulations of organic materials and cultivation operations can develop techniques for improving the efficiency of these systems.

2. **Prediction and verification of hydraulics and loading rates:** There is a need to improve the procedures and techniques for site selection including such factors as an evaluation of the water flux properties of the soils, the underlying geological strata, and the path that reclaimed water must follow.

3. **Management for nutrient removal:** Field research is needed to learn the fate of nitrogen as it is affected by management factors such as the wetting and drying cycles of various soils. Retention of phosphorus depends on soil sorption capacity and properties of the effluents that are not completely understood and integrated into a model of phosphorus retention.

OVERLAND FLOW SYSTEMS

High Priority

1. **Efficiency to remove algal suspended solids:** More information is needed on the efficiency of overland flow systems to remove suspended solids after pond pretreatment and under various flow rates. Such studies should also include investigation of the accumulation and decomposition of these solids on the soil surface.

2. **Pre-treatment requirements:** Investigation is needed to clarify the degree of pretreatment necessary for overland flow systems; to determine the minimum treatment needed for efficient operation of such systems; and to learn what additional treatment is needed to control odors.

3. **Effects of site aging and restoration of sites after cessation of operations:** Most of the U.S. overland flow systems are experimental. Information is needed on the length of time such systems will satisfactorily operate, and what action must be taken when the operation ceases and the site is converted to other uses.

Medium Priority

1. **Efficiency of systems to remove phosphorus:** Most overland flow systems inefficiently remove phosphorus and most of it is removed through surface soil contact followed by translocation. Methods must be developed to increase phosphorus removal, e.g., by the addition of alum to the raw wastewater stream.

2. **Study of the fate of trace organics:** More information is needed on trace organics and their ultimate fate in overland flow systems. Concentrations of trace organics can be expected to be higher in these systems than in slow rate systems, and, they may require special management practices.

3. **Study of pathogen retention and movement:** There is a need to study the pathogen retention and fate of pathogenic organisms in the overland flow system, particularly in regard to harvesting the cover vegetation and its disposal or use. Information is also needed relative to pathogen retention at these sites and their post-operation land use.

COMMENTS AND QUESTIONS FROM THE FLOOR:

Robert Smith, University of California, Davis: You mentioned removal of suspended solids as the high priority item for overland flow systems. Is this specific to algal solids? Can you elaborate on that?

Parker Pratt: One of the general papers indicated that removal of algae was not very efficient and the workshop felt that some research should be done relative to the removal of algae via overland flow systems.

Robert Smith: So your specific comment concerned algae--not suspended solids in general.

Parker Pratt: Yes, specifically the removal of algae.

A. L. Page: Parker, for the benefit of the group that may not be familiar with how slow rate systems, rapid infiltration systems, and overland flow systems operate relative to the intent of developing these systems, could you elaborate on these systems in terms of the operation and how they are used.

Parker Pratt: I'll give my concepts and then Richard Thomas can correct me, if necessary. In the slow rate system, the idea is to grow crops where the water application is near what we would use in irrigated agriculture in the west. The input is slow enough so that the soil is sufficiently aerated for crops to grow, yet fast enough to dispose of/or treat the amount of wastewater required. A few meters of water application are involved in the operation. In the west, sometimes as high as 4 to 5 meters of water are used, but the norm is only one or two meters.

In rapid infiltrations systems the volume is 30-40 meters of water per year or more. In overland flow systems water is introduced at the top of a run. The water flow is through a grass, plant canopy or slope and transmission time is half an hour; during this time phosphrous is removed, there is a modification, denitrification and oxidation of suspended solids. At the end of the flow water is channelled into a pond or to an additional treatment system. So, overland flow is part of a system, not the end of the system.

Richard Thomas, EPA, Washington: I want to add a clarifying addition on overland flow. Basically, it can be a complete treatment system. It is a designated use of a specific land area for wastewater treatment. It is a treat and discharge system so it is comparable to any treatment and discharge system.

Management Considerations in Sludge Use

Lee E. Sommers, Moderator
V. V. Volk, Secretary
Lee W. Jacobs, Co-Secretary

Participants: Dwight C. Baier, Robert Bastian, Thomas E. Bates, D. R. Bouldin, A. Lloyd Brown, B. L. Carlile, G. K. Dotson, R. H. Dowdy, Dennis Draman, Grace A. Draman, James O. Evans, Anne E. Giblin, Paul Giordano, Donna J. Griffin, Phil Hegeman, P. A. Helmke, Charles Henry, Thomas D. Hinesly, Wesley Jarrell, Lyle Jarrett, Y. Kanehiro, Robert F. Keefer, L. D. King, Paul Koenig, Shiou Kuo, D. R. Linden, Peter S. Machno, Milton W. Meyer, Raymond J. Miller, George A. O'Connor, Daniel O'Neill, A. L. Page, James Perry, Warren Sahs, Eileen M. Seaker, Robert C. Smith, Stanley M. Smith, J. J. Street, Dean H. Urie, Gary G. Van Riper, J. Vlamis, D. E. Williams, Ralph Young, Susan Young

INTRODUCTION

Primarily, sludge management was discussed with respect to plant nutrients, trace metals, organics, and systems management. Emphasis was on cropland, but because of slightly different research needs, subsections on forest lands and disturbed soils were developed. Human health considerations associated with the use of sewage sludge were not discussed.

In the decade since 1973, significant advances have been made in defining the benefits and the problems associated with use of sewage sludge on cropland. Excellent yields of high quality crops can be obtained when sewage sludge is substituted for conventional fertilizer materials. Guidelines are now available to permit the environmentally safe use of most sewage sludges on agricultural land. Improved management practices have enhanced public acceptance and increased percentage of sludge utilized on cropland.

Although utilization of sludge in forest lands is a relatively new alternative, existing research and development projects indicate sludge treatment of forests improves forest production while maintaining environmental quality. Because forest lands produce non-food chain crops, sewage sludge management is relatively straightforward. On the

other hand, long-term crop cycles, slope, soil depth, and application techniques may require development of special guidelines.

Sewage sludge is also successfully being used to reclaim disturbed land. Less information is available on the effects of sewage sludge on disturbed lands than its effects on cropland. Most studies show that the proper utilization of sludge in land reclamation at application rates higher than those used for agricultural land, will not cause degradation of soil, groundwater, or vegetation quality. In fact, sludge application improves the environmental quality.

GENERAL DISCUSSION

Nitrogen

In the past 10 years, significant advances have been made in utilization of sewage sludge as a source of nitrogen in agricultural cropping systems. Application of sewage sludge to cropland does not pose an increased threat to nitrate contamination of groundwater, provided that the amount of plant-available nitrogen (N) added to soils in sewage sludge does not exceed the N fertilizer recommendation for the crop grown. Existing research data can be used to determine the plant-available N content of sewage sludge, i.e., a portion of the inorganic and organic N. Current estimates can be used for ammonia volatilization following surface applications of sludge and for organic N mineralization rates. Methodology is available to advance our knowledge of N transformations in soil-sludge-plant systems.

Phosphorus

The application of sewage sludges to cropland at rates consistent with nitrogen fertilizer recommendations will probably result in phosphorus (P) additions in excess of those needed for crop growth. Such excessive additions in sewage sludges are of concern due to the potential for P entry into natural waters resulting in eutrophication. Phosphorus leaching is not a problem in the majority of mineral soils with possible exceptions being organic or selected sandy soils. Reduced plant growth due to excessive soil P is not a serious problem. Established soil testing procedures can be used to monitor available P accumulation in sewage sludge-treated soils. To minimize P problems, sewage sludge applications can be based on the P rather than N fertilizer recommendation for the crop grown.

Trace Metals

Trace elements, such as Cu, Zn, Mo, and Fe, present in sewage sludge that is added to agricultural soils at agronomic rates, can alleviate micronutrient deficiencies in plants. An increase in plant trace metal content following sewage sludge application can also reduce the need for supplements of the above trace elements, plus Se, in animal rations. In general, current guidelines for the cumulative application of Cu, Zn, and Ni are safe but should be refined to reflect regional soil differences.

Many current guidelines use soil cation exchange capacity (CEC) to limit metal (Zn, Cu, and Ni) additions to soils. However, the CEC may not be the most appropriate soil parameter to use in all regions of the U.S. Specific soil properties (i.e., texture, organic matter, iron oxides) are important and should be considered when limits for cumulative metal additions to soils are established. Total cumulative application of trace elements should be based on both the amount added in sewage sludge and the available metal content in the soil prior to any sludge application.

The majority of sludges do not contain excessive levels of Se or Mo. Some trace elements (Se and Mo), however, may be assimilated by plants grown on sewage sludge-treated soils at levels sufficiently high to cause animal health problems. Sufficient quantities of elements, such as Cu, Se, and Pb, may be ingested by livestock while grazing plants grown on sewage sludge-treated soil and thereby cause animal health problems. Livestock consume fair quantities of soil while grazing.

Organics

Concentrations of the majority of synthetic organic chemicals in sewage sludges are generally low. Most organics added to soils in sewage sludges will be decomposed within one growing season, with the exception of recalcitrant compounds (e.g., PCB's). Current federal regulations for polychlorinated biphenyl compounds in sewage sludges are adequate to protect animal health.

Management

Recommendations for sludge application rates are based on periodic analysis of plant nutrients and trace metals in the sewage sludge. Currently, insufficient information exists to integrate organics into the design of systems for sludge utilization on agricultural croplands.

No groundwater monitoring for nitrate is needed where the plant-available N in sewage sludge does not exceed the fertilizer N recommendations for the crop grown.

Good soil and water conservation practices should be utilized at all sludge application sites to minimize erosion losses and surface runoff, and thereby reduce the potential non-point source water pollution.

RESEARCH NEEDS FOR SEWAGE SLUDGE USE IN AGRICULTURAL CROPLAND

High Priority

1. **Establish phytotoxic limits of Cu, Zn, and Ni applied in sewage sludge and evaluate residual plant availability of metals following termination of sludge applications.** Emphasis should be placed on regional soil differences (clay type, iron and aluminum oxides), and management practices, (crop grown, pH).

2. **Develop models to predict and further refine estimates of N behavior in sewage sludge-treated soils, with emphasis on ammonia volatilization, organic N mineralization, and denitrification.**

Intermediate Priority

1. **Investigate the fate of organic chemicals in sewage sludge applied to soils with respect to (a) decomposition and mobility in soils, and (b) potential accumulation in plants and animals.** Methods to study the fate of organics need to be developed.

2. **Identify metal reactions between the soil-plant interface to determine metal availablity to plants and animals. Establish thermodynamic properties of the system in addition to empirical observations.**

3. **Evaluate soil tests, perhaps on a regional basis or as a function of specific soil properties, to relate soil metal levels to plant uptake or phytotoxicity.** Existing soils tests are applicable for detection of trace metal deficiencies rather than phytotoxicities.

Low Priority

1. Determine effects of sewage sludge on specific microbiological processes in soils and plants such as legume inoculation, plant disease, and nutrient transformation.

RESEARCH NEEDS FOR SEWAGE SLUDGE USE ON FOREST LANDS

High Priority

1. Quantitatively determine N cycling in forest lands so valid criteria can be developed for sludge utilization.

2. Develop sewage sludge application and management techniques including metal limits, surface runoff leachate quality, and other site selection criteria.

Low Priority

1. Evaluate metal effects on the forest ecosystem, including the flora and fauna.

2. Study the long-term response of trees to sewage sludge.

3. Determine the effects of sludge use on physical properties of wood.

RESEARCH NEEDS FOR SEWAGE SLUDGE USE ON DISTURBED LANDS

High Priority

1. Determine types of sewage sludges and optimum rates of application for establishment of permanent and self-maintaining vegetative cover on drastically disturbed land.

2. Determine the responsiveness of plant species to relatively high loading rates of sewage sludge as well as their tolerance to the adverse conditions of disturbed land. In selected regions, the response of native plant species to sewage sludge is needed.

Low Priority

1. Develop better management techniques for land reclamation with sewage sludge, e.g., depth and method of incorporation into soils, methods of establishing woody species in conjunction with grass-legume cover and monitoring approaches.

2. Determine the effects of sludge incorporation on soil formation processes in barren soil materials, the rate of re-invasion by microorganisms and invertebrates, the longevity of the newly formed soil system, and the development of plant root systems.

QUESTIONS AND COMMENTS FROM THE FLOOR:

Donald B. Aulenbach, Rensselaer Polytechnic Institute: Was the limit of PCB's in sludge for land disposal 10 ppm or 10 ppb?

Lee Sommers: We developed a general statement concerning the PCB limitations for the U.S. EPA regulations published in 1979. This regulation states that sludges containing >10 mg PCB/kg sludge (i.e., 10 ppm) must be incorporated into the soil rather than surface applied.

Donald Aulenbach: That is adequate, we shouldn't put very strict limits on PCB's in sludge until we have more positive proof that PCB's are harmful to our health.

Lee Sommers: Fortunately, the majority of recent sludge composition surveys indicate that most sludges contain significantly <10 mg PCB/kg.

Leon Chesnin, University of Nebraska: What level of zinc applied as sludge was considered to be toxic? Research at the Panhandle Experiment Station at Mitchell, Nebraska indicated that 1250 lbs/acre of Zn banded as zinc sulfate on irrigated corn was not toxic.

Lee Sommers: We tried to address general issues in our workshop, there was not sufficient time to address items such as specific limits on metals. With respect to specific limits, I refer you to individual members of our group and also to regulations or guidelines that have been developed in states or regions of the United States. Most workshop members expressed a belief that metal limits should be related to regional differences in soil characteristics. Other studies will support your conclusion.

Dale Baker, Pennsylvania State University: Monitoring requirements for disturbed lands need to be refined and related to desirable sludge rates and properties. The future use of the site should relate to cumulative metal loadings.

Lee Sommers: I only briefly described the recommendations. The need for monitoring requirements during reclamation of disturbed sites with sewage sludge will be included in the discussion of management techniques.

Susan Young, INCOG, Tulsa, OK: (1) Did your group discuss a total management program, including pre-collection system treatment (industrial pre-treat), and/or bulking all sludges that we cannot now land apply, and other sludge disposal options. Do you have any recommendations for these non-agricultural management aspects? (2) Did the group address non-mined disturbed lands (construction areas, slope stabilization, oil field/brine damaged lands)? Were application methodologies, rates, or special problems discussed? Are there any recommendations for further efforts in this area? Are there recommendations to take in the problems for disturbed/marginallands (low pH, high salts, shallow soils, etc.)?

Lee Sommers: Part 1. We only addressed land application of "acceptable" sludges. Pretreatment programs and other sludge disposal options were not discussed. Part 2. A fairly small number of people were concerned with disturbed lands. If anybody in our group would like to respond to this question, please feel free to do so. The research needs stated were for disturbed lands in general.

Lee Jacobs, Michigan State University: Suggestions on research needs for disturbed lands would be welcomed.

Gerald Stern, EPA Cincinnati: In the management workshop - was sludge disposal on dedicated land sites considered?

Lee Sommers: No. We mainly discussed systems where sewage sludge is used as part of an agronomic fertilization program. In general, whenever sludge is applied in excess of agronomic rates, more extensive monitoring such as nitrate in groundwater and metals in crops are needed. Specific management practices needed on a dedicated site were not discussed.

Stephen B. Smith, Black & Veatch, Denver, CO: Lee, you made a comment to the effect that we need to establish species of plants for disturbed land reclamation. This may be the case for several parts of the country, however, this is not the case for the Colorado and Wyoming area. The Mine Land Reclamation Board of Colorado, and the Land Quality Division of Wyoming Department of Environmental Quality dictate the plant species that have to be used. Therefore, the issue becomes how well will these plants germinate and grow on sludge-amended soils, and whether or not they can sustain the vegetation into the future. I would submit to you that a research need exists to determine the effects of sludge application on these native semi-arid species for use in the western United States.

Lee Sommers: We appreciate your comment.

Terry Logan, Ohio State University: Should the same metal accumulation criteria be used for disturbed land and agricultural land? This question relates to currently disturbed land that is reclaimed through sludge applications and then, at some point in the future, is used for agricultural crop production.

Lee Sommers: This is a very important consideration. Unfortunately, the management group had insufficient time to address this topic.

Melvin Webber, Environment Canada, Burlington, Ontario: (1) Did the group concerned with sludge management consider defining maximum permissible heavy metal contents in sludge for agricultural use? (2) Would the chairman of the sludge management group elaborate on the groups' discussion concerning soil CEC in relation to heavy metal loadings? (3) Did the sludge management group define the heavy metal contents of a median sludge? I realize that Logan and Chaney in their paper presented at this conference define median sludge, but does the group agree with their definition?

Lee Sommers: Part 1: There was basic agreement that we definitely minimize problems by applying low metal or low organic sludges, i.e., the "median sludges" that have been discussed in several of the papers at this workshop. We did not specifically address upper limits for sludge constituents such as metals. The general consensus of the group was that current metal guidelines are appropriate in the future. That is, we should limit the amount of metal added to a soil rather than limiting the metal concentration in the sludge. We did not try to evaluate specific limits for metal additions to soils.

Melvin Webber: In this regard, I might add that most other countries, who have guidelines for sludge use in agriculture, have defined maximum permissible heavy metal contents of agriculturally acceptable sludges. However, considerable variation exists in the maximum acceptable levels for sludge constituents. Using zinc as an example, the maximum for different nations varies from 2,000 mg/kg to 10,000 mg/kg.

Lee Sommers: As a general comment, some data suggest that less metal uptake by plants occurs with low metal sludge than with high metal sludge, when both are applied at rates to add equal amounts of a metal to soil. Perhaps, a combination of the two approaches is needed.

Part 2 - The majority of workshop participants indicated that soil properties are important relative to limits for total metal additions to soils. Cation exchange capacity was originally selected for use in setting limits because (a) it is an easily measured soil parameter, and (b) it is positively correlated with the soil components (i.e., organic matter, iron oxides, and clay) that tend to minimize the solubility and plant availability of metals in soils. It was never intended to imply that metals added to soils in sludges are retained through a cation exchange mechanism. Metals added to soils in sludges are essentially 100% insoluble and are not present as exchangeable cations. It is now recognized that the CEC approach fails in some regions of the U.S. Soil texture may be more appropriate in some areas. The CEC does not have any relationship whatsoever to potential metal problems in calcareous soils. The CEC approach was developed in the mid-70's for the north central region of the U.S. where organic matter is the major contributor to cation exchange. It is now realized that pH is more important than CEC in controlling metal uptake by plants in all regions of the U.S.

Part 3 - It was my impression that our group was in agreement with the definition of a median sludge proposed by Logan and Chaney in this workshop. The group did not formally adopt the Logan and Chaney definition.

Harvey Luce, University of Connecticut: Did your group consider site criteria, site evaluation, site selection as a consideration to sludge management use?

Lee Sommers: The items discussed relative to site management would include soil pH control and practices to minimize surface runoff and soil erosion so sludge materials would not be transported to surface waters. We did not discuss specific limits on slope.

Harvey Luce: I was thinking site selection, especially as it relates to forest lands. These sites may have steep slopes or soils that are shallow to bedrock. If we are going to apply sludge to forest lands, it will be applied to sites that have one or more of these less than ideal conditions. This area appears to be a major research need.

Lee Sommers: One of our highest research needs was additional information on site selection and management practices for sludge use in forests. We support your statements.

Jack Cooper, National Food Processors Association: Did your group address the need for records on such things as: site locations where sludge was applied, and soil CEC data? Did you discuss the need for these data to be available to the public? Did you discuss the assignment

of responsibilities to make sure that all parties accomplish their assigned tasks? Did you discuss whether there should be a mechanism available to assign responsibilities?

Lee Sommers: The general feeling of our group was that records must be compiled and maintained to comply with current guidelines and regulations on cumulative additions of metals to soils. Records must be available to determine if a metal limit is being approached. We did not discuss liability.

A. L. Page, University of California, Riverside: I believe the question relates to the need to keep records where sludge has been applied and the need to keep records of the composition of the sludge applied there. Any efforts along those lines?

Lee Sommers: Since the guidelines currently used in most states are based on cumulative metal additions, records on sludge composition and sludge application rates at different sites must be kept.

Lee Jacobs, Michigan State University: Since philosophies are different between regions and approaches to technical matters (e.g., fertilizer recommendations for crops) are different between states, it is difficult to develop a national statement on sludge management. We felt that regional concerns and regional differences in soils are best handled by individual states. Record-keeping probably is needed in all regions of the U.S. because of the need for limits on total metal loadings.

Lee Sommers: In general, the group feels that we know how to manage the sludge utilization systems for cropland and that we are fine tuning the basic approach. The basic approach is the same from Maryland to California, but specific number may vary.

Dale E. Baker, Pennsylvania State University, Chairman, NE Work Group: For the management group, I would urge you to consider problems and solutions on a regional basis. Soils are different, people constraints are different, and the requirements in relation to total or available land change. For example, in the northeast a permit may involve a 20 acre field, compared with up to 2,000 acres at other locations. The geochemistry effects must receive more emphasis.

Public Health and Risk Assessment: Pathogens

Elmer Akin, Moderator
Wylie Burge, Secretary
Bernard Sagik, Co-Secretary

Parttcipants: Donald Aulenbach, Gerald Berg, Scott Clark, Robert C. Cooper, D. J. D'Alessio, C. R. Dorn, Samuel R. Farrah, Jane Forste, Paul Fitzgerald, Charles Gerba, Thomas L. Gleason, III, Charles Hagedorn, Tom Holm-Hansen, George Jackson, Barbara Ann Kerdolff, M. Dale Little, Ebba Lund, Gordon A. McFeters, Barbara Moore, Robert Northrop, R. T. O'Brien, Mark Sobsey, Charles Sorber, Gerald Stern, Richard Ward, Willis Whitfield, William Yanko

The Pathogen workshop participants support the concept of land application of domestic wastes. With proper management and reasonable safety allowances based on sound research data, we believe that land application can be a safe, beneficial, and acceptable disposal alternative. However, three potential routes of exposure to pathogens in wastes applied to land are recognized, i.e., (1) direct contact with waste-amended soils, (2) aerosols from spray application, and (3) ingestion of contaminated surface-runoff water and groundwater.

Currently, there are no known outbreaks of infectious disease attributable to land application of wastes. This is not to say that no cases of illness have occurred. Disease produced by a common source exposure are not typically obvious by observation. Systematic study by epidemiological investigation is generally required to "prove" the occurrence of a common source outbreak and to identify the likely source.

The documentation of sporadic cases of illness from a common source environmental exposure is a different question and no satisfactory investigative approach is available to obtain these incidence data. Therefore the documentation and quantitation of health risks from the three potential routes of exposure in domestic wastes applied to soil is beyond the current state-of-the-art.

Efforts to gain these data were obviously the objective of research needs identified by the 1973 workshop participants. The participants at the 1983 workshop believe that the current needs generally remain the same as those identified by the 1973 workshop with perhaps different

priorities and emphases. This is not to say that nothing has been accomplished in the past 10 years. On the contrary, much has been accomplished. Major epidemiological studies have been or are now being conducted to determine the microbial health hazards of wastewater spray irrigation and sludge application. Major improvements in monitoring methods, especially in the virus area, have been achieved. These methods, applied to field studies, have documented significant migration of pathogens in the soil subsurface and groundwater. Data generated during the decade since the 1973 workshop have provided a scientific basis for developing sludge disposal criteria at the federal level.

The considerable progress achieved will allow a more rapid rate of progress in monitoring for the occurrence of pathogens in a wide variety of environmental media, i.e., water, wastewater, sludge, and soil. Emphasis can now be placed on quality control and quality assurance aspects, on standardization of methodologies and on a general improvement in the comparability of data from one study to another. From these developments, the data generated will likely produce more definitive trends in predicting microbial behavior in the environment. Compared to 1973 capability, a more accurate indication of "what is out there" and "how many" in terms of treatment effectiveness and pathogen transport can more readily be determined.

In terms of current detection capability, the sludge disposal criteria (Federal Register, September 13, 1979) is considered adequate to protect the public health from infectious disease via this transmission route. Perhaps the most likely attack on these criteria will be from the standpoint that they are overly restrictive. Data must be generated that will give guidance to any consideration for the relaxation of these criteria.

Participants in this workshop believe that considerable research information is now available to conclude that site specific criteria and management guidances are warranted. Meteorological and soil-type variations yield significant differences in microbial survival and behavior when applied to land via domestic wastes.

Epidemiological studies were given a top priority research need by the 1973 workshop participants. Important studies have been conducted and the results are becoming available for making generalized hazard-assessment decisions. It is believed that the epidemiological approach to determining microbial health hazards from land application has been adequately applied to those exposure circumstances where statistical analysis of data can be applied. Some studies directed to high exposure populations (worst case situations) may still be appropriate. However, this workshop does not believe that epidemiological studies remain as the highest research need.

Workshop participants believe that major resource allocations should focus on work that would allow a more accurate prediction of pathogen exposure and illness outcome through modeling approaches. The influence of specific actions, e.g., various levels of pretreatment, on the level of risk could be evaluated and may indicate more cost-effective treatment /disposal options.

In order for this approach to be meaningful and to avoid the erroneous conclusion drawn from faulty modeling exercises, the following specific research needs are identified by the workshop in order of priority per microbiology discipline:

RESEARCH NEEDS

Bacteriology

1. Determine survival and regrowth of established human pathogens (e.g., Salmonella) and newly recognized waterborne pathogens (e.g., Campylobacter, Yersina) in sludge and in soils amended with sludge and wastewater.

2. Improve methods for detection, enumeration, and assessment of virulence of these pathogens in sludge and wastewater.

3. Determine effectiveness and mechanism of action of treatment processes in reducing pathogens in sludge and wastewater.

4. Determine relevance of existing indicator organisms as indicators of the presence of new pathogens.

5. Identify factors and determine survival and translocation of pathogens and indicator organisms in sludge-amended soils under different infiltration rates and in different soils.

Epidemiology

1. Conduct comprehensive studies of "worst case" exposed populations, e.g., sewage plant workers and families. Determine occurrence of protozoan, helminthic, bacterial and viral infections with a focus on gastrointestinal illness.

2. Conduct studies of compost-site workers for occurrence of Aspergillus fumigatus infections.

3. Determine infectious dose of pathogens, especially differentiation of respiratory and ingestive routes.

Parasitology

1. Determine pathological effects of repeated low level Ascaris infections.

2. Determine virulence of swine vs. human Ascaris and importance of cross-infectivity in alternate hosts.

3. Determine potential pathological significance of Naegleria and similar organisms in waste-treated soils.

4. Determine important factors affecting survival of Ascaris ova in sludge-treated soil and develop die-off curves under various meteorological conditions.

Virology

1. Conduct site monitoring studies of survial and movement of viruses applied to land under wide ranges of climatic, hydrologic and waste conditions. Quality assurance of field data should be given high priority.

2. Determine effectiveness of sewage treatment processes before land application, especially disinfection, for virus reduction to acceptable low levels.

3. Develop design and operational criteria, e.g., loading rates, hydrology, soil characteristics, groundwater use, and crops grown, for land application systems with respect to viruses.

4. Develop methods to detect those enteric viruses that are known to cause human illness from fecally-contaminated water and wastes but are presently difficult to cultivate or as yet not cultivatable, e.g., Hepatitus A virus, Norwalk-type viruses, and rotaviruses.

5. Develop methods of high and defined efficiency (quality assurance) for the more easily cultivate enteric viruses in wastewater, sludges, soil, and groundwater associated with land application systems.

QUESTIONS AND COMMENTS FROM THE FLOOR:

Charles A. Sorber, University of Texas: I sat in on that group and I want to say that your summary of the consensus was excellent. I don't think anybody is going to argue with your summary, but I know you inadvertently missed one of the items that I think is important as you ran down through the slides. Perhaps Woody could put that Virology Section slide up.... Note that item #3 dealt with development of criteria based on what we have learned in the last 10 years and I think that this is particularly important to this group. You might want to go over that particular topic. You just ran by it inadvertently, Elmer.

Elmer Akin: I'm sorry, I did miss that. Again, I think a major concern here, and I would like the virology group spokesman to comment on this is that there is a need for site-specific criteria. Dr. Wellings, from the State of Florida, Dr. Vaughan from Long Island and investigators from other places that have sandy soils, have produced a fair amount of data on the movement of agents through that type of soil and it is the specificity of these data that needs to be taken into consideration. I would like for the virology group to comment on that further.

Anne Giblin, Woods Hole Oceanographic Institution: Some studies in a marine environment have shown that bacteria isolated from sewage dumping areas have a higher incidence of antibiotic resistance than bacteria from other areas. Do you think this is a problem?

Elmer Akin: I didn't hear you very well. A higher incidence of antibiotic resistant bacteria? What source?

Ann Giblin: Yes, from sewage dumping areas.

Elmer Akin: I think it is becoming very well documented that sewage does provide a medium for the transfer of plasmid-mediated antibiotic resistance. It is somewhat debated among bacteriologists what the real significance of this observation is, it perhaps has been ongoing for a long time. Since antibiotics were introduced in this country, physicians have used them freely to treat most everything infectious. I don't know the health significance of this observation. We didn't really address that in our group. Anybody want to respond?

Bernard Sagik, Drexel University: Reluctantly. At the 1981 Plasmid Conference in Santo Domingo, the issue arose. Natural levels of antibiotic resistance, particularly in South and Central America and third world countries, was even greater than among the isolates that were seen in the United States and that was pretty darned high, as most of you know. Out of that came a statement by the group, circulated very widely at the time by all persons at the Santo Domingo Conference, urging research workers and physicians to begin to be cognizant of the potential health risk of letting all of these organisms, and the potential for plasmid exchange, loose in the environment. The concern was not because of the "bugs" as much as because of the loss of effectiveness of antibiotics which physicians had depended on for almost 30 years. More recently, Stuart Levy started a group which is soliciting support for control of and conservative use of antibiotics. You may even see Public Health and Preventive Medicine make a come back from this.

Elmer Akin: Just one quick comment. I suppose we can have some confidence that even though the transfer of antibiotic resistance does not occur, we still have disinfectants that don't recognize this resistance.

Lee Sommers, Purdue University: Did your group address relative risks? Currently, wastewater treatment plants are discharging effluents directly into rivers and other surface waters that may serve as drinking water supplies. What is the relative risk of having a few virus percolating into a groundwater from a wastewater utilization site as opposed to putting all viruses directly into the surface water?

Elmer Akin: We talked about it, and we've talked it before. In fact we almost started at that point because the discussion of every meeting of microbiologists, particularly virologists, comes down to this point. How do we determine relative risk and how do we obtain the evidence that is really needed to prove or disprove a health hazard from these agents in the environment. We basically have two approaches: epidemiological investigation and some type of modeling based on agent transport and infective dose data. That's a more indirect approach to the question. The epidemiology approach does have limitations. It is difficult to separate risks at low levels of common-source exposure to microorganisms for the background risk from all other exposure routes. I don't know that I can go much farther in responding to your question other than to say that microbiologists would like for you out there to agree that if we show that the agents are commonly present in the environment, distant from pollution sources that this indicates some level of risk. The risk may be acceptable but the data should not be swept under the carpet. The more agents that are out there, and we are finding more because our methods now are better, the greater the risk to some degree--some undetermined degree. The more the environment becomes "saturated" with organisms the greater the possibility of exposure. The

acceptability of risk is a subjective question--I don't know if we can approach it through objective research studies. At least, we haven't done a very good job of it to this point.

Herbert Pahren, EPA, Cincinnati: Elmer, I would like to point out that several years ago, Metcalf and Eddy made an attempt to determine the relative risk and perhaps either Crites or Pound are those in the room, I believe, might want to comment on the results of their study on that question.

Ron Crites, George Nolte and Associates: What we did was to compare the relative risks from activated sludge with stream discharge to a slow rate land treatment system. Dr. Bob Cooper was involved in the analysis with us. His conclusion was that in terms of pathogens, the relative risk of exposure for the public is about the same from these two systems.

Charles Gerba, University of Arizona: You made a comment that epidemiological studies on plant workers, which you considered the worse case, would give us an idea of risk to the general public. I really can't agree with that because I am not aware of wastewater operators and sludge operators eating or drinking wastewater and sludge. I would consider direct ingestion the worse case example of what we are trying to protect the public from. I don't think we have evidence to indicate that the results from those studies can be used to indicate the risk to the general public. What it does tell us is the risks to the plant workers. I believe they are aware of potential hazards they are working with but I don't think we can make that type of statement or use the evidence to indicate risk to the general public.

Elmer Akin: Scott, you are surely going to respond to that.

Scott Clark, UC Ohio (Kettering Laboratory, Cincinnati): I think there are ways where the exposure is comparable. There are several studies besides ours that have shown an increased risk of illness in wastewater workers as indicated by excess gastrointestinal illness such as diarrhea. Now, some of this disease could be due to the hygenic habits at the plant or in the collection system--lack of good wash rooms, lunch rooms, etc. Workers could be, in effect, eating some of the wastes attached to contaminated food, or by smoking. Airborne aerosols and dust in some of the wastewater and sludge treatment environments that I have seen are at high enough concentrations to literally amount to "eating" the waste. We thought that this population was the worse case in many respects. There is difficulty in doing the studies at land application sites where the exposure isn't usually as great. Although, we heard last night from Bob Northrop that some of the families in the farm studies, where the kids get on the field right after they are spraying, did

experience some illness. So, maybe there are some possibilities for further study there. There have been several studies recently, one in New York city and one each in France and Germany, that do show an excess of infections from the parasites.

Elmer Akin: Your understanding that we're talking about where we may find a worse case situation. We don't always know what the worst case situation is and Chuck takes exception to the fact that sewage treatment plant workers are a worse case example for those living around a sewage treatment plant or drinking water that might come from such land application sites. You know, Chuck, that is an open question to some degree.

Jack Cooper, National Food Processors Association: I have a comment or perhaps a question concerning the relative risk of viruses discharged into surface water versus those that may get into groundwater. When surface water is used as drinking water it goes through a filtration and chlorination system before it is drunk whereas groundwater often is used straight out of the ground without any treatment whatsoever and is considered to be very clean good water. I'm not sure that you are equating apples with apples here--you've got apples and oranges.

Elmer Akin: Well, that's true to some degree. I think perhaps we are moving in the direction of less confidence that groundwater is microbiologically safe. The data that some of the virologists are generating indicate that viruses do move through certain soil types further than we had anticipated and that perhaps groundwaters are in some jeopardy from surface microbe contamination. This concern is addressed several ways. One is to try to identify sole source drinking water aquifers and not allow any contamination or at least be more restrictive in potential risk of contaminating those aquifers and another is to concede that some groundwater sources are contaminated and anyone drinking water out of those are advised to treat that water--at least disinfection. I think if our group had to prioritize, groundwater contamination is perhaps one of the greatest areas of research need and perhaps the greatest potential health risk from land application of wastewater.

Paul Fitzgerald, University of Illinois: Near the end of our session last night, we had a lively discussion about what does risk really mean. I think this might be an interesting area to bring up to this general group. Does risk imply "disease" in itself or does risk simply imply "infection" with any particular organism. It seems to me that this is a point that needs to be further defined. It has been kicked around for a long time, trying to decide what risk assessment is. If we don't have a definition of what risk really implies, I don't see how we can ever come up with an idea of determining what risk assessment is. I would kind of like to hear some comments from the general group.

Elmer Akin: Anyone care to address that? Paul, I'm not sure you're going to get comments.

Herbert Pahren, EPA, Cincinnati: I'll make a comment. In my briefcase I brought along a memo from Elizabeth Anderson, the Director of the Office of Health and Environmental Assessment, Research and Development, Environmental Protection Agency in Washington, entitled "Suggested Definitions for Risk Assessment and Related Terms" that includes a definition of health risk assessment in lay terms. It might be of value.

> *Health risk assessment: Risk assessment may be used to determine (1) the likelihood that an environmental agent may cause human disease (e.g., potential to cause human cancer), and (2) on the assumption the agent causes a particular disease, the magnitude of the likely impact on public health in quantitative terms, given current and projected exposure levels.*

I don't know whether that helps, Paul or not.

Elmer Akin: I'll just make one comment on that. I think that at times we've been criticized for doing epidemiological studies for which the endpoint is infection rather than illness. Some would say, so what. Are infections really risk? Is that really documenting a concern to health? I think this endpoint gives the epidemiologists and microbiologists perhaps a more objective way of looking at risks. Because the infection rate is going to be higher then the illness rate, we are more likely to find a result. When we obtain these data we can likely correlate the ratio of infection to illness, e.g., one illness to 100 infections. Infection data are available for many of the agents. Perhaps the infection itself is not the risk but it allows us to project the illness risk in a more objective manner.

Cecil Lue-Hing: Metro Sanitary District, Chicago: First, I must thank the workshop participants for at least listening to my comments of yesterday. I would like to stress, however, that it was never my intention to put down the research, the researchers, or particularly those involved in microbiological research. I do represent a constituency that has always supported research. Particularly, I have, if you will, served as your envoy in places like Washington, D.C., appeared before groups such as the Appropriations Committee, and tried forcefully to encourage Congress to put more money into research. I hope that I can continue to do that. The caution that I would like to urge, is this: in today's environment, we're in a tremendous competition for funding. In any corporate budget any item appearing under research gets severe scrutiny. There is a perception being developed, perhaps more so in the constituents that I represent, that we may not be getting much benefit from the research we are paying for. It would be helpful for the people I represent who are payers of all sorts of bills to know that we are facing some

problems. It would be especially beneficial if the research could serve as medium to improve the conditions under which we are operating. Particularly, in keeping the environment protected, perhaps at a lower cost. If as researchers we are unable to do that, the perception will grow that perhaps we should not invest more in research because if we are as protected now as we can ever be, how much more protection can research offer us. If the perception further develops, it could very well be that the influential part of the constituency, that researchers need for their support, may view additional requests for research bluntly as self-serving. I hope we never get that far. And I hope that I can still serve as an envoy of my constituency to continue to support it. I intend to do that as long as I'm permitted to do so. I do believe strongly in the need for research both long-term and near-term. The difficult part about research is justification. And unfortunately, that is the area in which most of you are focused. We need help from you the researchers, to continue to support you, and I hope you can understand that and hope you can take these comments seriously.

Elmer Akin: Thank you, Cecil. I knew you would have to respond to my comment, and I was afraid you had left the room since you weren't the first to the microphone. I do appreciate your position and I know that a lot of sludge is generated every day in Chicago and your organization must dispose of it and you want to do so in a safe manner. Microbiologists have gone on record here as saying that we support the land application of wastes and we feel that it can be done appropriately. I think I can say quite confidently that we aren't here to undermine or create artifical risk but realizing that there is a large amount of waste and that controlled managerial design activities are not perfect in all cases. We feel the obligation of stating what needs to be done to protect the public's health from microbiological contamination and also state that we have not had perfect tools to find out. This last decade has seen tremendous progress in giving us some of those tools. We aren't there yet but I believe in the next decade, Cecil, there will be a lot more usable data coming forth. The data may very well be on the side of relaxing some of these regulations, but we want to believe that we have the support of other interested parties to allow us to continue that work and candidly, I don't know that we felt we were hearing that here. The idea that all the answers are not in seems to be somewhat of a common belief, now. This may sound like a real self-serving statement but and I don't mean it to be that. Man's relationship with microorganisms is a very dynamic one--it changes constantly because both are living, evolving beings. We find microbiological agents a problem today that were not a problem 10-20 years ago. Giardia lamblia was almost considered as normal flora in this country 20 years ago. Now we know it causes a major waterborne disease problem, giardiasis. Now we are concerned about the waterborne transmission of that disease--not only in the outbreak setting but the background level as well. We will never be done with research on the relationship between man and microorganisms and I would hope that there is not the perception that one day all answers will be in and that microbiological research will no longer be needed. The evolution of

relationships between these two life forms means that we will continually need to study those interactions if we are to protect the public health.

Bernard Sagik, Drexel University: I certainly endorse that statement, Elmer and I would like to state very briefly something that Dr. Lue-Hing brought out. When Cecil said that some of the questions raised 10 yrs. ago are long-term problems, and we are still asking them, I think perhaps he has forgotten that 10 and 12 years ago we did not know, in many cases, how to isolate viruses from the environment. When they were first isolated and reported at meetings, people said, how do you even know that they're viruses. How do you know that they're not toxic agents on those cell sheets; can you pass them; can you identify them? We've come a long way since that point. There were questions as to whether you could ever find viruses percolating through the soil because everyone told us that that didn't happen. I think we accept the fact that it does happen now. The questions that Cecil has raised are legitimate but I don't think it is the duty of the person doing the research necessarily to lower the cost.

Cecil Lue-Hing: I understand where researchers comes from, and I hope you never misunderstand the statements I used in reference to lowering the cost or making the bureaucratic burden any less. This is a perview of another group. For example, if by lowering the cost you interpret that to mean making the regulations more workable that is not your job. I am not here to ask you to do that. What I am here to ask you to do is condense the work of the last 10 years and objectively and honestly say to us did you learn anything and if you did, does it give us an opportunity to revisit some of the decisions you have made, some of which may have gotten into the regulations. The issue of regulations is not your concern, and for God's sake don't interpret my comments to mean that. I am asking you to look at your data and tell us if your findings indicate to you that the risks are there, or is there need for change, revisiting, re-evaluation, that's all. The end result of that is not your concern. You continue to do the science.

Elmer Akin: Cecil, I can clearly say that the last 10 year have produced certain progress in our capability to study that question. I don't know that it has helped us to determine the risk if you mean by that, can we give you a number or give you some tangible risk evaluation. Hopefully that will come, but it is going to be very difficult, to really pull a number out of the air or even perhaps to develop a standard. Take virus exposure as an example. To say that that exposure below a number is safe, and exposure above that number represents considerable risk for a large percent of the population is a difficult concept. I will ask you to bear with us. I think we are generating some very valuable data on, if nothing else, occurrence of organisms in the environment.

Mark Sobsey, University of North Carolina: I would like to respond to this recent dialogue and to the desire by some for the microbiologists to try and summarize the work of the last 10 years or so and to put that into some context that will then allow others, perhaps, to make further decisions and judgments. I think one of the reasons that we are not able to do that there are several reasons some of which have already been mentioned, is that at least with respect to viruses, the ones that appear to be the most important in terms of environmental transmission of disease, are viruses such as hepatitis A, the Norwalk type of viruses and possibly rotaviruses that we did not even have the capability to work with 10 years ago. We couldn't even do serological work with them in the context of environmental studies. We are still not able to cultivate some of these viruses and can cultivate hepatitis A and rotaviruses only with great difficulty. So, these are the viruses for which we need information with respect to their transport, movement, in land application systems and in other environmental areas. That's why I think we, in part, made the research recommendations that additional studies need to be done, many of which are along the same lines that we have been trying to do over the last 10 years or so. The studies must be done with these important viruses and we cannot necessarily presume that their behavior is going to be the same in the environment as the other viruses that we have been studying for the last 10 years. It has become quite clear that the other virsues that we have studied the ones that we have been using as models, are apparently not the most important one in terms of disease transmission from environmental sources.

William Davis, City of Long Beach: I'm certainly no expert on virus, but I have a question. Is it not the general consensus of this group that the lack of information is not sufficiently large that we should oppose land application of municipal effluents and sludges but rather that we should move ahead in spite of your fears and concerns?

Elmer Akin: I think that was my original statement--that we do feel with current knowledge there is no reason we cannot go ahead with land application.

William Davis: That is a very important statement.

Elmer Akin: That was really the reason for our opening statement because we do feel strongly that there's no reason not to go ahead with land application, from a microbiological standpoint. There have to be caveats of course, such as the application of good management practices and the continuation of research to better quantitate and identify the risks. We don't know quite what the risks are. We feel that they are perhaps minimal but they have not been quantitated and I think we need to determine what they are.

William Davis: I think that's the point that when you are a user of this information and having to deal with it, those fundamentals aren't getting out.

Susan Young, INCOG, Tulsa, Oklahoma: I think we ought to have some follow-up on this. Again, I'm a user for the information. I need comparative risk assessments to explain things in the matter of what is the risk of exposure to viruses via land application of waste as compared to a child's exposure to virus in the classroom. We need that kind of information. It is not necessarily a research need, but it is a need we definitely have.

Elmer Akin: We could take a vote but I think most microbiologists-epidemiologists feel pretty confident in stating that the risk is greater in the classroom. We do feel that person-to-person transmission is a major route of transmission in this country. Does that mean that we aren't concerned about the other route? I don't think so. We talked about this in our session and I'll be very brief. Let's say arbitrarily that 90% of infectious disease transmission is from person-to-person, and only 10% from environmental exposure. What then. Do we ignore that 10%, or do we try to minimize that further? I don't think the American people are ready to accept that level of risk if it is controllable. Certainly they are not in regard to drinking water. They want their drinking water to be "risk free" or as close to it as can be obtained within reasonable economic constraints. I think that is the key word, "reasonable". That's what we work out in sessions like this, and in state legislatures, I suppose.

Public Health and Risk Assessment: Organics and Inorganics

Herbert Pahren, Moderator
Delbert Hemphill, Co-Moderator
James Ryan, Secretary
Mary Beth Kirkham, Co-Secretary

Participants: D. C. Adriano, Dale E. Baker, Thomas Bick, Garfield N. Biddle, George Braude, Richard Bull, Rufus Chaney, Michael Connor, R. B. Dean, Gregory Diachenko, W. R. Dunlop, Gayle E. Edmisten, Michael Flynn, Ernest C. Foulkes, George Fries, David Graveland, Philip Hopke, Robert Horvath, Charles F. Jelinek, Roma P. Jenkins, Norman Kowal, Terry J. Logan, John Meier, Raymond W. Miller, L. M. Naylor, Michael Overcash, Magnus Piscator, Michael Plewa, Raymond F. Shipp, James E. Smith Jr., Richard T. Sprague, Robert Swartz, M. A. Tabatabai, Wade Talbot, John M. Walker, M. D. Webber

In 1973 there was little discussion on the impact of human health or organic and inorganic contaminants in municipal wastewater and sludge when applied to land. Shortly thereafter, the health impact was recognized and various applied research studies were carried out at an accelerated pace.

The state of knowledge today is such that there no longer are serious gaps in knowledge. Research still required may be considered fine tuning the information presently at hand.

Three position papers relating to this subject, i.e., *An Overview of Public Health Effects*, *Specific Organic Compounds*, and *Metals* accurately summarize what we know today. This information is of great value to the designer of land treatment systems and the regulator.

Key points developed in this workshop may be summarized as follows:

1. At annual application rates of less than 15 Mt/ha, of sludge of median composition where a reasonable drainage and cyclic establishment of sustained aerobic soil conditions occur, and groundwater remains deeper than 0.3 to 0.7 meters from the soil surface, leaching of metals and organics should pose little threat to groundwater resources.

However, adjustments may be necessary because of site specific conditions. Concerns were the lack of detailed information on the composition of the sludge and the fate of these materials. Similar concerns were expressed for effluents.

2. It was the consensus that wastewaters should not be chlorinated prior to high rate application to land, since precursors to halogenated organics may be present.

3. The factors that affect the migration of organics through various soils and decomposition in groundwater are research needs.

4. There appears to be no health effects associated with surface water contamination if runoff is controlled.

5. It was the consensus that additional epidemiological studies with exposure of high risk groups to cadmium need not be initiated until the present studies have been completed.

6. It was the strong recommendation of the group that an international panel should re-evaluate the 1972 recommendations of the WHO/FAO on limits for cadmium intake in view of the substantially better data base today.

7. Except for cadmium and lead, metals are not expected to create a human health problem in sludge-amended soils. The direct deposition of lead on plants should be evaluated for passage through the food chain.

8. There was a concern by some over accurate information on present dietary exposure to cadmium. However, it was brought out that FDA is pursuing a better definition of dietary intake.

9. There was need for more research on how various nutritional factors, including other metals, influence the absorption of cadmium.

10. There was concern over the use of sludge containing high levels of lead on garden soil due to the potential pica problem. However, no specific research recommendation was made.

11. **Research is needed for the impact of several elements in sludge on grazing animals.** The principal elements of concern were Co, Mo, F, and Se.

12. **It was agreed that additional studies on the impacts of organics on the food chain through the animal-human pathway were not necessary for the near future.**

13. **More research is needed on the passage of organics through the plant-human pathway, including development of better methodology and bioassay systems.** Concern is not limited to priority pollutants.

14. **Research is needed on the use of treated wastewaters and sludges for aquaculture purposes.** Levels of organics and inorganics in ponds and the uptake by fishes are initial concerns.

15. **There was general agreement among workshop participants and a recommendation that the U.S. EPA re-evaluate its September 13, 1979 regulations on the limits for cadmium contained in solid waste applied to land.** The data base today is substantially better than that used at the time the regulations were promulgated.

QUESTIONS AND COMMENTS FROM THE FLOOR:

William Davis, City of Long Beach: There seems to me to be a consensus that much has been learned from the research over the last 10 years and what I thought I heard said here was that the information developed, at least by Herb, would justify a re-evaluation of existing regulations that now exist which govern the land application of effluents and sludges? Is that the general consensus I hear?

Herbert Pahren: We specifically referred to re-evaluation of the WHO/FAO guidelines that was one of the many considerations in developing regulations. In your comment you used the term regulations. The WHO/FAO numbers are not regulations per se. Our view is that whenever guidelines or regulations, are established for the public to follow, they should not be set in concrete, but from time to time they should be re-evaluated.

William Davis: As I look at the September 1979 criteria, for example, my recollection is that those were based in large measure on much of the concepts that you alluded to from WHO. Isn't it reasonable to conclude that if the basic concepts are now needing to be re-evaluated, so should not the regulations?

Herbert Pahren: As a good principle, yes.

A. L. Page: It might be advisable that this question be directed to each of the moderators. It might be helpful in terms of the proceedings of this particular session.

William Davis: The comment seems to me to be grouped into two categories. One is what is the impact on the farmers - phytotoxicity? The other is on the health side of it, particularly the point just made related to the cadmium issue and the metals issue.

Herbert Pahren: I might point out Bill, that the issue came up during our deliberations and it was the feeling that the workshop members did not want to be in a position to develop new regulations yesterday afternoon and last night, but they did agree on the principle that as new information is developed, it makes good sense to re-evaluate the guidelines and criteria we have.

William Davis: We knew that 10 years ago and, the other thing I would agree wholeheartedly with, it is beyond the scope of this group to define the new regulations but I think is well within the scope of this group to comment on the current regulations and the need to look at those existing regulations in the light of what has been learned in the past 10 years. That is something that is well within and should be, otherwise, the question again comes up, why continue it?

A. L. Page: Comments from other moderators on this question?

L. E. Sommers, Purdue University: (Representing the sludge group): I think it was the consensus of the sludge management group that the sludge regulations need to be re-evaluated. However, this is still a regional consideration because of soil differences and cropping patterns, availability of different management practices, and so forth. But I think the group basically felt that though there probably should be a national standard that should be set but it should be based on regional differences, and it does need to be re-evaluated.

Raymond Loehr, Cornell University,(representing the Engineering group): It is always hard to speak for a group, but from the discussion yesterday, I believe that our group would endorse that last comment. It is a natural extension of one of the recommendations I discussed earlier, the re-assessment of the heavy metal loading criteria.

Herbert Pahren: As one member of the panel pointed out, it is a matter of how much of a factor of safety one should select in setting these criteria. Some people want to cut it quite thin and others want to be very conservative and use a very high safety factor. It's a matter of where the line is drawn.

A. L. Page: Parker, would you care to comment on the discussion which transpired in the workshop which you moderated?

Parker F. Pratt, University of California, Riverside, (representing the Effluent group): The workshop I moderated did not discuss this question directly, but I am sure if we polled them, they would agree that any time there is new information related to guidelines or standards, whether they be regional, state, or federal we should have a re-evaluation.

(Question by A. L. Page restated for Elmer Akin, [representing the Pathogens group] temporarily out of the room): In light of the data that we have developed over the past 10 years is there a need at this particular time to re-assess or re-evaluate the criteria which govern the land application of municipal effluents and sludges. Correct me if I am wrong, Bill.

William Davis: I used the term regulations not criteria.

Elmer Akin: I am aware of only one regulation and that pertains to the land application of sludge only. It is premature to re-evaluate it now because the results of some major studies are pending. If, however, new informatin indicates that regulations can safely be relaxed, then I think the change should be made. I am quite confident the pathogen workshop participants would agree although this specific question was not put to them.

A. L. Page: Dick Thomas, were you going to make a comment?

Richard Thomas, EPA, Washington: From the standpoint of the land treatment of effluents, the only fixed regulation pertains to the protection of groundwater.

Robert Dean, Environmental Science and Technology, Copenhagen, Denmark: As a member of this group on metals, I think that we should put into the final document something out of the position papers by Logan and Chaney, and the base paper published by Ryan et al., that some of our original assumptions on cadmium were probably conservative, by a factor which has been variously estimated of between 3 and 10. I do think we should hear back again from people who presented or who worked on these position papers, and I think it should go into the document. I do feel personally, that we were overly cautious on cadmium and that the cadmium limits turned out to be right at the cutting edge which decided whether or not land application was going to be economical. We perhaps have thrown out possible applications that would have been the best way to go if we had a little more information. Now we've got the information, and I'd like to see it used.

Charles Jelinek, FDA, Washington, D.C.: I am going to comment on what Bob Dean just said, but that wasn't why I came up here originally. I think we ought to recommend reassessment by an international body of acceptable daily dietary intake of cadmium in light of additional information that has been developed on toxicity of cadmium. I would agree with that. I definitely don't agree that there is enough information in to show that we've been too conservative. Indeed, there are gaps in knowledge in toxicity, and it might even be that when this international body reassesses it they might even come up with a lower acceptable

dietary intake. I don't think we can pre-judge them at all. Now, in light of conservatism, I am speaking now as a member of the Bureau of Foods, of the Food and Drug Administration, we're always going to be conservative when it comes to deciding what level of a given toxicant we can tolerate in our food supply. We would be remiss in our duties if we were not conservative. We would also be remiss if we are too conservative. So it gets around to a question of what safety factor do you allow. And I would point out to people here that in the case of cadmium in general, we can be more liberal in the safety factor that we allow than we do in the case of food additives. The reason being, that cadmium is a naturally occurring toxicant; it is harder to control and therefore, we have gone with lower safety factors than we would in judging food additives. Finally, I'd like to point out that I don't think we want to disregard safety factors too much. All we have to do is think of what happened in the Hyatt Hotel in Kansas City.

A. L. Page: In the way of attempting to rap up this particular issue, I sense that there is a general agreement amongst the moderators and among the group as a whole, that the time is right to re-evaluate and I believe that this can go as a recommendation from this particular workshop.

William Davis: Comments were made about existing regulations, relative to cadmium, were overly conservative. Was there a consensus about the conservatism?

A. L. Page: I don't believe it would be productive in this large a group to tend to either defend the current regulations as they pertain to cadmium or to suggest what they should be, if those in effect are not adequate.

William Davis: That was not the question. The question was, is it the consensus of the group regarding the degrees of freedom, the conservatism that is in those regulations, is it open for re-evaluation - is it too much, too little?

A. L. Page: I think the group agrees that this is open to evaluation but I don't think we should be presumptive relative to what the outcome will be.

Thomas Gleason, EPA, Washington, DC: I would just like to say there is an interagency group that is working on the consideration of re-evaluation of the regulations, and that the proceedings of this conference will play an important role in that accomplishment. As to whether there is an over-conservative viewpoint from a toxicological viewpoint, going

from a safety factor of 4 down to 1, how low can you go? I think we can adequately justify the safety factor.

Richard Bull, EPA, Cincinnati: I want to reinforce what Herb just said, in the sense that I think that in the data that's been presented to us in the past few days, there is no real indication that there is a problem with health risks associated with organics and associated with sludge application. But one thing I wanted to point out, and it get's back to some of the comments on cadmium, is that most of the conclusions that were arrived at were not arrived at by considerations of the health effects of the individual compounds. They were really going from that data which was available primarily from that offered by Mike Overcash in terms of how much of that material is likely to get into a situation where man was going to be exposed. We really did not deal in any case with the intrinsic health effects of the individual compounds. And that is the reason, even in the case of cadmium for which we did discuss the recommendation of WHO, I don't think this was the proper body to come to grips with that particular issue at all. Insofar as whether it's more conservative or less conservative, I just didn't see the cadmium experts. It was not something that could have been voted on in this particular situation.

Dale Baker, Pennsylvania State University: Under statement #1, there is a very serious omission, and that refers to 15 mt/ha of median composition of sludges. The intent was that sludges of median composition has to be entered in there because 15 mt/ha of any sludge would not be safe. #11 you have some non-metals listed in there as metals, like Se and F.

Herbert Pahren: OK, I'll make the adjustment.

Philip Hopke and Michael Plewa, University of Illinois: We feel compelled to add some additional comments after listening to your report on the workshop session in which we participated. It is our feeling that the issues discussed regarding organic contaminants were not fully represented in the summary report. Michael Overcash and others besides ourselves have raised unanswered questions regarding persistence, translocation, metabolism, sequestering, and biological effects. For example, the sludge-animal-human food chain results as described by the FDA looked at a very small portion of the spectrum of compounds that may be present in sludge. The examination of only PCB's and pesticides is hardly exhaustive. We can make some estimates of the uptake of lipophilic compounds into fat or milk through octanol/water partition coefficients but there is much yet to be learned as the health effects of such accumulated materials for many, many possible species.

The development of much more information about the fate of organic contaminants in soils, their ability to be incorporated in the food chain should be a high priority area of research during the next few years. Although it has been shown by Kirk Brown that some waste materials can decompose rather rapidly, other materials particularly polyhalogenated or multiple-branch compounds may have quite long lifetimes. It should be remembered that the dioxin contamination of Times Beach, MO, soils were not considered to be a problem in 1975 since it was felt that dioxin would rapidly decompose in the soil. Let us not make the same mistake with other contaminants in sludges or effluents.

We believe strongly that the direct biological activity of the sludge, particularly as expressed in whole plants such as our Zea mays wx-locus mutagenicity test, have potential ecological impacts that have not been considered. In addition, such tests indicate the potential for sludge-grown plants to act as a vector for introducing mutagens into the human food chain. It has been shown that plants can take up a compound, (e.g., atrazine), and metabolize it into a stable, extractable mutagen. Thus, plants can act to induce or enhance the biological effects of trace contaminants. As we understand it, there are groups who are raising the mutagenicity of sludge as an issue in land application. There is a published report of increased mutagenic activity in the urine of male Fischer rats fed beets grown in sludge-treated soils. The interpretation of these results are difficult because we do not know if any of the internal tissues of the rats have shown any effects. It would seem to us that in important research objective should then be evaluating the transport and fate of mutagens in the sludge and from those results, identifying the level of risk potential to the general public. Our work on the mutagenicity of various sludges indicates a substantial range of biological activity is found in different municipal sludges. We believe that it would be very simple to develop a good set of biological indicator organisms that could be grown in a demonstration plot before large scale operations with a given sludge are begun.

In summary, we believe that the potential for human health effects from trace organic contaminants in sludge and effluents were substantially underestimated in the report than was made to the closing workshop sessions and we urge that more emphasis be given to organics as research needs. We believe that the organics may well come to play as an important role in determining the suitability of sludges and effluents for land treatment and that further research now will result in fewer problems being discovered later.

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