National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment

May 19 – 20, 2000
St. Louis, Missouri

Sponsored by

U.S. Environmental Protection Agency

Electric Power Research Institute

National Decentralized Water Resources Capacity Development Project

National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment

1001446

Final Report, March 2001

EPRI Project Manager
K. Carns

EPRI • 3412 Hillview Avenue, Palo Alto, California 94304 • PO Box 10412, Palo Alto, California 94303 • USA
800.313.3774 • 650.855.2121 • askepri@epri.com • www.epri.com

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EPA ASSISTANCE AGREEMENT NUMBERS CR825922-01-0 AND CR827881-01-0

CITATIONS

This report was prepared by

EPRI Community Environmental Center

This report describes research sponsored by EPRI, U.S. Environmental Protection Agency, and National Decentralized Water Resources Capacity Development Project under EPA Assistance Agreement Numbers CR825922-01-0 and CR827881-01-0.

The report is a corporate document that should be cited in the literature in the following manner:

National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment, EPRI, Palo Alto, CA, U.S. Environmental Protection Agency, and National Decentralized Water Resources Capacity Development Project: 2001. 1001446.

ACKNOWLEDGMENTS

U.S. EPA
Project Officer:
Jim Kreissl

National Decentralized Water Resources Capacity Development Project
Project Coordinator:
Andrea L. Arenovski

Steering Committee:
Jay Turner; Jean Tribull; Jerry Stonebridge; Patricia Miller; Raymond Ehrhard; Richard Otis; Steven Lindenberg; Ted Loudon; Tom Yeager; Valerie Nelson; Kevin Sherman

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INTRODUCTION

On May 19-20, 2000, the Research Needs Conference for "Risk-Based Decision Making for Onsite Wastewater Treatment" was convened in St. Louis, Missouri. This conference, funded by the U.S. Environmental Protection Agency (EPA), was the culmination of an eighteen-month long effort by the National Decentralized Water Resources Capacity Development Project (NDWRCDP) to assist onsite wastewater leadership in identifying critical research gaps in the field. The five "White Papers" included in this Proceedings, along with the reviewer comments for four of these papers, provided the basis for extended discussion. Topics for the papers had been determined from input at three prior daylong forums in Tampa, Florida; Kingston, Rhode Island; and Seattle, Washington.

Background

In 1997, EPA issued a landmark report, "Response to Congress on the Use of Decentralized Wastewater Treatment Systems" (EPA # 8332-R-97-001b), which stated that "adequately managed decentralized wastewater systems are a cost-effective and long-term option for meeting public health and water quality goals, particularly in less densely populated areas." However, EPA cited a number of critical barriers to expanded use of decentralized wastewater systems, including:

- Lack of knowledge and public misconception
- Legislative and regulatory constraints
- Lack of management programs
- Liability and engineering fees
- Financial barriers

The EPA-funded NDWRCDP is currently addressing a number of these barriers. Formed in 1996, the mission of the Capacity Development Project is to advance the state of management, technology, and practice in decentralized wastewater treatment through support of training, research, and development activities. Member institutions in the Project Steering Committee are the Electric Power Research Institute, the Water Environment Research Foundation, the National Rural Electric Cooperative Association, the Consortium of Institutes for Decentralized Wastewater Management, the Coalition for Alternative Wastewater Treatment, the Water Environment Federation, and the National Onsite Wastewater Recycling Association. The Project is administered through Washington University in St. Louis.

One of the most important priorities of the Capacity Development Project has been the development of a coordinated research strategy to resolve scientific uncertainties and strengthen the foundations of practice in the decentralized wastewater field. In the past, research on onsite wastewater treatment has received comparatively little federal funding and minimal attention by leading academic and other researchers. This relative inattention can be explained by both the lack of a strong federal role in onsite treatment, and by the longstanding view of septic systems as a temporary wastewater solution on the way to permanent sewers. Septic systems were also, for many years, considered primarily as a simple approach to "dispose" of wastewater effluent below the surface of the soil and away from the home, rather than as a more complex "treatment" system.

As EPA reported to Congress, future wastewater needs of the nation can increasingly be met by properly managed onsite and cluster treatment systems. However, state and local regulators and other policymakers must be convinced by scientific findings that decentralized wastewater systems adequately protect public health and the environment and will be maintained. And, homeowners, engineers, designers, installers, and managers of onsite systems must have greater confidence in the long-term performance and cost-effectiveness of traditional soil-based systems and new technologies.

Regional Forums

While there had been earlier efforts to discuss a research agenda for the onsite wastewater field, such as an EPA-WERF sponsored meeting in 1993,[i] the Capacity Development Project considered it vital to take the time necessary to identify research needs through a systematic, inclusive process of discussion and feedback throughout the country. Beginning with the November NOWRA meetings in Fort Mitchell, Kentucky in 1998, the community of decentralized wastewater experts has been given the opportunity to submit questions and concerns they consider important to advancement of the field.

Research needs forums were convened in three different areas of the country, in order to uncover the differences in regulatory concerns, treatment capacity of soils, climate, etc. These meetings were held at the University of South Florida in Tampa in February 1999; University of Rhode Island in June 1999; and the University of Washington in September 1999. At each of these sessions, regulators, industry leaders, academics, planners, and others were asked to make presentations or otherwise participate in breakout sessions. Approximately two hundred experts participated in one or more of these meetings. Written comments were also submitted.

Risk-Based Decisionmaking

The ultimate goal of the research needs project has been to identify and prioritize critical research gaps in the decentralized wastewater field. Early on in the project, it was determined that this process would be greatly facilitated by adoption of a risk-based decisionmaking framework. In 1996, EPA's Office of Research and Development (ORD) had analyzed and reorganized the Agency's research priorities and strategies using risk assessment and risk management principles and criteria, and, in general, EPA has been attempting to rationalize national policies and practices in the context of relative risk.

The EPA-ORD report, "Strategic Plan for the Office of Research and Development",[ii] was therefore used as a template for this project as well. EPA defined risk assessment in that report as a process to understand and evaluate the magnitude and probability of risk posed to human health and ecosystems by environmental stressors. Risk management combines these risk characterizations with statutory, legal, social, economic, and political factors in assessing regulatory or other options to manage risks.

While the precise methods of risk assessment and risk management are unfamiliar to most participants in the decentralized wastewater field, it was agreed in the various regional forum discussions that rough risk analysis was implicit in existing state or county onsite regulations, for example in requirements for groundwater separation or setback distances to allow for basic treatment in the soils. However, innovations in pre-treatment and soil-based systems, emerging concerns for nutrient and pathogen contaminants, proposals for performance-based standards, and more complex forms of management challenge the decentralized wastewater field for a much more complex and rigorous documentation of treatment processes and risks.

Initially, Glen Suter, and later Dan Jones, both from Oak Ridge National Laboratory, were contracted to guide the Capacity Development Project Steering Committee through the principles and methods of risk assessment and risk management. At each of the regional forums, Dan Jones described risk-based decisionmaking, as well as examples of risk assessments in other fields. This risk-based framework, while initially new to most of the audience, proved to be a helpful organizing framework for discussion. While there were regional differences in some areas of concern, eventually a consensus emerged about the major critical research areas that were important to practitioners.

"White Papers" for the Research Needs Conference

Four major research areas were defined at the conclusion of the regional meetings. They are:

- Fate and transport of nutrients
- Fate and transport of pathogens
- Long-term performance of soil-absorption systems
- Economics of decentralized wastewater systems.

National leaders were then identified to prepare White Papers in each of these areas, and two reviewers were also selected to critique each of the papers at the research needs conference. Dan Jones was asked to prepare a White Paper on risk assessment and risk management, and to incorporate specific onsite wastewater examples that had been cited in the regional meetings, as well.

The White Paper authors were asked to summarize the literature, both published and unpublished, and to identify gaps relevant for rigorous risk-based decisionmaking. They were also asked to follow the format of the earlier work of EPA's Office of Research and Development in several key respects, specifically in focusing the papers on the most serious risks to human health or the environment, and in covering needs for risk management tools, as well as methods and models of analysis. Authors were also asked to follow the EPA-ORD example, by providing tables identifying priority research needs and the types of studies that could be formulated to address those needs.

Additionally the authors were instructed to address issues at both the individual site ("micro" level) and at the watershed ("macro" level).

In the first white paper, "Integrated Risk Assessment/Risk Management as Applied to Decentralized Wastewater Treatment: A High-Level Framework", Dan Jones of the Environmental Sciences Division of Oak Ridge National Laboratory presents the basic principles of risk assessment. He then describes the framework for defining engineering, ecological, public health, and socioeconomic aspects of a decentralized wastewater problem and management options. The final section of the paper describes the method of using risk assessment and risk management criteria to develop as research needs agenda.

The second paper, "Design and Performance of Onsite Wastewater Soil Absorption Systems", was co-authored by Robert L. Siegrist, Environmental Science and Engineering Division, Colorado School of Mines; E. Jerry Tyler, Soil Science Department, University of Wisconsin; and Petter D. Jenssen, Agricultural Engineering Department, Agricultural University of Norway. This paper reviews design and performance issues for both a classic, modern soil absorption system, and for alternative, modified technologies. Reviewers for this paper are Aziz Amoozegar of North Carolina State University and James Converse of the University of Wisconsin, Madison.

The third paper, "Research Needs in Decentralized Wastewater Treatment and Management: Fate and Transport of Pathogens" was prepared by Dean O. Cliver, Department of Population Health and Reproduction, School of Veterinary Medicine, University of California, Davis. This paper describes pathogens of concern in domestic wastewater, treatment in standard and alternative onsite systems, and uncertainties about pathogen risks at the micro and macro scale. Reviewers are Chuck Gerba of the University of Arizona and Marylynn Yates of the University of California, Riverside.

The fourth paper, "Research Needs in Decentralized Wastewater Treatment and Management: A Risk-Based Approach to Nutrient Contamination", was co-authored by Arthur J. Gold, Department of Natural Resource Sciences, University of Rhode Island and J. T. Sims, Department of Plant and Soil Sciences, University of Delaware. This paper describes nitrogen and phosphorous pathways and risks through treatment systems and into broader ecosystems. Reviewers are Ray Reneau, Jr., Virginia Technical Institute and Will Robertson, University of Waterloo.

The final paper, "Economics of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits" was co-authored by Carl Etnier, Agricultural Engineering Department, Agricultural University of Norway; Valerie I. Nelson, Coalition for Alternative Wastewater Treatment; and Richard Pinkham, Rocky Mountain Institute. This paper describes important direct and indirect costs and benefits to be considered in decentralized wastewater treatment decisionmaking, as well as decisionmaking structures that in the future would integrate public health, environmental, engineering, and socioeconomic risks. Reviewers are Chris English, USDA Rural Development (MN), and John Herring, Coastal Program, Department of State, New York State.

Looking Ahead

In coming years, the Capacity Development Project will use the research needs analysis in the five White Papers as a template for funding of research projects. Participants in the May 19-20, 2000 conference will be sent surveys and asked to rank research needs identified by the White Paper authors. These national rankings will be used to develop Requests for Proposals (RFP's) in the highest-priority areas. Initially, Congress has appropriated $1.5 million for support of Capacity Development Project-sponsored research projects, and it is anticipated that additional funds will be available in the future.

Appreciation

Numerous individuals have contributed time to the research needs identification project, and their participation is much appreciated. Members of the Project Steering Committee attended all the regional meetings, as well as the final conference. Several people deserve particular thanks for their substantial efforts in organizing the regional forums and the final research needs conference. These people include Kevin Sherman, formerly of the Florida Department of Health and now Director of the Florida Onsite Wastewater Association; George Loomis and David Dow of the Rhode Island On-site Training Center at URI; Jerry Stonebridge from the Washington Onsite Wastewater Association; and Ray Ehrhard of the EPRI-Community Environmental Center at Washington University in St. Louis. Finally, our Project Officer, James Kreissl at EPA's National Risk Management Research Laboratory in Cincinnati, Ohio, has provided vision and leadership throughout the project, and his contributions have been greatly appreciated.

Special Acknowledgment

A special acknowledgement is given to Valerie Nelson. Her leadership, dedication, and perseverance drove this project, helped ensure its success, and has led to ongoing work under the National Capacity Development Project.

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[i]Water Environment Research Foundation. 1993. Small Wastewater Treatment Systems Workshop: A Strategic System for Identifying and Prioritizing Research Concepts. Alexandria, VA. 30 pp.

[ii]EPA. 1996. Strategic Plan for the Office of Research and Development. EPA/600/R-96/059. Office of Research and Development, Washington, DC. 49 pp.

Integrated Risk Assessment/Risk Management as Applied to Decentralized Wastewater Treatment:
A High-Level Framework

Prepared

by

Daniel S. Jones
Environmental Sciences Division

Anthony Q. Armstrong
Life Sciences Division

Michael D. Muhlheim
Engineering Technology Division

Barbara V. Sorensen
Energy Division

of

Oak Ridge National Laboratory
Oak Ridge, Tennessee

July 2000

1 OBJECTIVES

This paper is part of a process intended to guide the research and development of tools and information needed to evaluate and advance the field of decentralized wastewater treatment. This paper introduces the risk assessment paradigm and broadly tailors the integrated risk assessment/risk management approach to the field of decentralized wastewater treatment. Specifically, it:

- introduces the concept and terminology of integrated risk assessment as it applies to decentralized wastewater treatment;

- provides a context within which issues can be identified, organized, and prioritized; and

- identifies several high-priority risk assessment issues for decentralized wastewater treatment, based primarily on input from three regional forums, as a basis for further discussion at the national conference.

This high-level framework is not intended to be an exhaustive treatise on integrated risk assessment for decentralized wastewater treatment, answer the questions raised in the regional forums, or provide example applications of the framework to specific decentralized wastewater treatment scenarios.

2 INTRODUCTION

This paper is part of a process intended to guide the research and development of tools and information needed to evaluate and advance the field of decentralized wastewater treatment.

2.1 Background

A previous part of this process was the convening of three regional forums on risk-based decision making in decentralized wastewater treatment. They were attended by representatives of regulatory agencies, industry, and academia. These regional forums were held during 1999 in:

- Tampa, Florida
- Narraganset, Rhode Island
- Seattle; Washington

The purpose of these forums was to introduce the risk assessment/risk management approach to experts in the field of decentralized wastewater treatment and to gather from them a compendium of issues important to the application of decentralized wastewater treatment. The findings of those meetings were used to focus this paper and to identify issues of high priority to decentralized wastewater treatment experts and interested parties.

2.2 Scope and Approach

This paper supports the final component of this research and development process, which is a national research needs conference. This conference is a national meeting to identify and prioritize research needs in the field of decentralized wastewater treatment. It is organized around the principles of risk assessment and risk management. Research needs will be discussed and prioritized with respect to their ability to support high priority risk management objectives.

This paper is the first of five topical papers commissioned to support the national research needs conference. It introduces the general concepts of risk assessment and risk management, which are used to address specific technical issues in the subsequent papers. Those papers address:

- process, function, and performance of wastewater soil absorption systems;
- fate and transport of pathogens;
- fate and transport of nutrients; and
- direct and indirect costs and benefits.

This paper begins with a general introduction to risk assessment, presents a high-level framework for integrated risk assessments of decentralized wastewater treatment systems, and briefly describes of how the principles of risk assessment and risk management are applied to the prioritization of research needs.

3 PRINCIPLES OF RISK ASSESSMENT

This section provides a general introduction to the terminology and concept of integrated risk assessment as it applies to decentralized wastewater treatment systems. It includes a list of risk assessment definitions, many of which are adapted from EPA(, 1998).

3.1 Definitions

<u>Analysis</u> is the second element of the general risk assessment process in which the technical issues associated with estimating risks are addressed. It typically consists of an analysis of exposure and an analysis of effects.

<u>Assessment Endpoints</u> are an explicit expression of the value that is to be protected. They consist of an entity, a property of that entity that can be measured or estimated, and, whenever possible, a level of effect on that property that constitutes an unacceptable risk.

<u>Conceptual Model</u> is a visual depiction, with supporting text, of the relationships between the stressors and the endpoint entities.

<u>Integrated Risk Assessment</u> is the process of bringing various disciplines (e.g., engineering, ecology, public health, and socioeconomics) together to derive information and insights that would not otherwise be possible.

Measures of effects are measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it was exposed.

Measures of exposure are measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint.

Measures of ecosystem and receptor characteristics are measures of environmental attributes that influence the distribution of a stressor or receptor attributes that influence exposure and response.

Problem Formulation is the first element of the general risk assessment process in which the purpose of the assessment is clearly defined, the problem is clearly stated, and a plan for analyzing and characterizing risks is developed.

Receptor is an entity that is exposed to one or more stressors. It is the assessment endpoint entity or its surrogate. It may include human and non-human organisms, as well as systems of interest (e.g., ecosystems and social systems).

Risk is the likelihood that a course of action (or lack thereof) will result in an undesired event.

Risk Assessment is the scientific (objective) process of estimating the likelihood and magnitude of adverse effects.

Risk Characterization is the third element of the general risk assessment process in which the methods and models developed in the analysis phase are combined to produce qualitative or quantitative estimates of risk.

Risk Management is the subjective process of deciding which actions to take in response to a potential risk

Stressor is any physical, chemical, or biological entity that can induce an adverse response in a receptor.

Susceptibility is a measure of the exposure and sensitivity of the receptor to a stressor.

Sensitivity is a measure of how readily the receptor responds to the stressor.

3.2 Purpose of Risk Assessment

Risk is the likelihood that a course of action (or lack thereof) will result in an undesired event. We assess risks because we must choose between alternative courses of action, each of which has some degree of uncertainty associated with it. We informally assess risk when we make choices in our everyday lives. Formal, technical risk analysis is used when the likelihood or magnitude of the potential risks are perceived to be very high or very uncertain.

The outcome of complex environmental actions is rarely known with high certainty. The risks include squandering limited resources, failing to reduce the most significant impacts, and creating more significant problems than those already existing.

Because resources are limited, all risks cannot be eliminated. Tradeoffs must be made when choosing which risks to reduce and how much of a reduction is enough. For example, one cannot eliminate all risks of nutrient loading to an adjacent water body from waterfront housing without prohibiting development in the immediate vicinity. The risks can be dramatically reduced by mandating public sewer systems, but even that action entails risks. The most notable risk is the inappropriate expenditure of resources. An alternative is the use of decentralized wastewater treatment systems. The risks of nutrient loading to the water body may be higher than with a centralized system, but the risk of unnecessary expenses may be lower. The results of such decisions would have a tremendous impact on the local community and environment.

It is important for such decisions to be made from an objective frame of reference. The process of formal risk assessment provides that frame of reference.

3.3 Key Characteristics of Risk Assessment

The fundamental characteristic of risk assessment is that it provides transparency and objectivity to the decision making process. This is achieved by implementing a standard, logical structure that facilitates interactions between the technical experts and the decision makers. Key characteristics of that structure include :

- the separation of risk assessment from risk management,
- clearly defined assessment endpoints,
- rigorous technical analyses, and
- explicit characterization of uncertainty.

Each of these characteristics will be discussed below. However, separating the processes of assessment and management warrants further discussion here.

Risk assessment is the scientific process of estimating the likelihood and magnitude of adverse effects. This includes identifying the types of direct and indirect effects that may occur.

Risk management is the process of deciding which actions to take in response to a risk. It considers the results of the risk assessment along with other factors explicitly excluded from the assessment of risks (e.g., politics).

Maintaining the separation between risk assessment and risk management is critical to the integrity of the risk assessment process. That is, risk assessment is an objective process, whereas risk management is a subjective process in which value judgments are made. Injecting value judgements into the assessment process reduces the credibility of the results. That reduces the utility of the assessment in the decision making process.

This does not mean that risk assessors and risk managers must not work together. Indeed, it is equally important that the assessment address the needs of the risk managers. For that reason the process explicitly encourages interactions between the assessors and managers. However, the assessor must ensure that the estimates of risk are not improperly influenced by those interactions.

3.4 Types of Risk Assessment

There are several ways in which the principles of risk assessment can be applied to the field of decentralized wastewater treatment. These include comparative, discipline-specific, and integrated assessments.

Comparative assessments are used to choose among alternative courses of action. In the example above, one could compare the estimated risks of nutrient loading from decentralized treatment and centralized treatment.

The risk assessment process can also be applied to a very narrowly defined topic such that it only addresses issues within one technical discipline. For example, one could assess the risks of system dysfunction as part of an engineering risk assessment without estimating the risks of that dysfunction to human or ecological receptors.

Integrated risk assessments are more broadly targeted than discipline-specific assessments. They pull together disparate types of information into a cohesive and comprehensible format. This high-level framework is specifically intended to support integrated risk assessments of decentralized wastewater treatment systems. Such integrated assessments could be conducted for several alternative treatment systems, thus resulting in a comparative integrated assessment.

Integrated risk assessment is commonly thought of as a model building effort. However, integrated assessment is better defined as a process for bringing various disciplines together to derive information and insights that would not otherwise be possible. For example, engineering risks can be used to define the stressors to be evaluated in health, ecological, and socioeconomic risk assessments.

However, important feedback loops are developed to ensure that necessary decision making information is obtained. For example, the socioeconomic assessment may determine that costs and dysfunction rates of a particular pretreatment technology are critical to the assessment. This may lead to a more detailed engineering assessment for that technology.

Integrated risk assessment also integrates:

- space (e.g., across a watershed),
- time (e.g., across the life of a system),
- sources of risk (e.g., other activities in a watershed),
- results (e.g., direct effects causing indirect effects), and
- multiple endpoints (e.g. engineering costs and social impacts).

3.5 Standard Structure

There are three general elements in the standard risk assessment paradigm:

- problem formulation,
- analysis, and
- risk characterization.

These are depicted in Figure 1. The order of conduct is roughly as presented, but the overall process is often iterative. For example, one may conduct a screening assessment and then a definitive assessment. The results of the screening assessment are used to refine the problem formulation for the definitive assessment.

Problem formulation is a process for generating and evaluating preliminary theories about what effects might occur. It is the first step in developing a sound assessment. This component requires the input of the risk manager to ensure that the final results will support the decision making process. Key steps include defining the:

- characteristics of the stressors,
- spatial and temporal scope of the assessment, and
- functional relationships of the stressors and receptors.

Products of this step include clearly defined endpoints, a conceptual model of the interactions between the stressors and the receptors, and a plan for conducting the assessment. The details of these products should be agreed to by the decision makers.

Analysis typically includes analysis of exposure and analysis of effects. These analyses occur concurrently but separately. Analysis of exposure is the technically rigorous evaluation of spatial and temporal characteristics of the stressors. Analysis of effects is the technically rigorous evaluation of the responses of receptors to the specified stressors.

These analyses are interdependent. That is, the types of stressors determine which effects should be evaluated and the time and space over which the effects occur determine the kinds of estimates of exposure that are needed.

Risk characterization is the process of combining the estimates of exposure with the estimates of effects. This process also is technically rigorous and should result in estimates of the probability and magnitude of specific effects.

Risk management is the final component of risk-based decision making. Although functionally separated from the assessment process, it provides a critical point for feedback and refinement of the assessment process for future iterations. This is the stage at which other factors are weighed, including the various types and magnitudes of risks, the costs and benefits of potential actions, and the ethical and political considerations of each action.

3.6 Current Use of Risk Assessment

Risk is implicitly included in current permitting regulations for decentralized wastewater treatment systems. Permitting regulations typically include minimum separation distances between the drain field and the water table and minimum setbacks from property boundaries and potable water supplies. Such regulations vary among state and local jurisdictions and have been established through experience with standard decentralized systems in typical soil conditions. These regulations are implicitly based on risks.

However, the estimation of explicitly defined risks associated with these rules has not been accomplished. A major impediment to assessing the risks of standard systems is the lack of a comprehensive and consistent approach to defining the potential risks. Alternative treatment systems are typically used when the prescriptive permitting guidelines for standard septic systems are violated. Given that baseline risks have not been estimated, it is very difficult to establish permitting rules for alterative treatment technologies.

Performance-based permits are generally issued when prescriptive guidelines do not apply. Such permits require potentially expensive monitoring and maintenance and are viewed as a risk by both home-owners and regulators.

- Home-owners fear the potential costs if their system fails the tests.
- Regulators fear the impacts to the public and environment if the system fails.

The likelihood and magnitude of risks of these two types of dysfunctions are generally not well known. As a result, both parties will tend to minimize their perceived risks, potentially by avoiding alternative treatment systems altogether.

Standardized methods for explicit risk estimation under a variety of conditions is needed to enhance and improve performance-based permitting for alternative treatment systems. With such methods one could develop and test a set of permitting guidelines that account for the site-specific variables that drive risks (e.g., depth to groundwater, soil type, temperature) to a suite of receptors (e.g. various members of the public and ecological receptors). These may ultimately take the form of new prescriptive regulations that can be applied to a wider variety of systems and sites.

Efforts are underway within the field of decentralized wastewater treatment to develop risk-based approaches to decision making. These include modeling efforts and community demonstration projects in which stakeholders identify the issues of concern and help guide the decision-making process. These efforts are an excellent step towards explicit risk-based decision making.

A significant limitation of these approaches is the lack of a standardized method for integrating disparate risks into a comprehensive approach to risk-based decision making that can be applied at various sites and geographical scales. Additional limitations generally include a lack of explicit, risk-based endpoints (i.e., a specified level of effect on an important property of the entity to be protected) and a failure to address all major types of risks (i.e., engineering, ecological, public health, and socioeconomic).

4 HIGH-LEVEL FRAMEWORK

This high-level framework provides a blueprint for integrating different discipline-specific assessments into a single cohesive risk assessment framework for decentralized wastewater treatment systems. Four disciplines are addressed:

- engineering,
- ecological,
- public health, and
- socioeconomic.

The general framework for decentralized wastewater treatment systems is presented in Figure 2. It consists of a general problem formulation, a component framework for each discipline, a general risk characterization, and a brief discussion of risk management issues. The explanation of each of these components will be aided by including high-priority issues as examples.

4.1 General Problem Formulation

The general problem formulation defines the scope and objectives of the general framework. This entails:

- identifying the spatial and temporal bounds within which the framework will be applied;
- identifying the potential stressors and receptors;
- selecting assessment endpoints and ensuring that they can be addressed within the appropriate component assessments;
- developing a generalized conceptual model of the systems to be evaluated; and
- selecting appropriate measures of effects and exposure.

These steps occur in approximately the order they are presented, though the problem formulation process is often iterative. For example, additional indirect stressors may be identified as the conceptual model is developed. Receptors and assessment endpoints would then be selected for those indirect effects. Each of these five steps is discussed below.

4.1.1 Spatial and temporal bounds

Identifying the spatial and temporal bounds within which risks will be considered is important because those bounds will determine what types of stressors and receptors are appropriate. This, in turn, determines which assessment endpoints and which discipline-specific assessments should be included.

For purposes of discussion, we will consider two spatial scales, the micro-scale and the macro-scale. The micro-scale refers to an individual residential plot with an on-site drinking water well and a decentralized wastewater treatment system. The macro-scale refers to a watershed which contains many individual decentralized systems, as well as other point and non-point sources of pollution.

4.1.2 Stressors and receptors

Identifying the potential stressors and receptors entails listing all of the credible ways in which the treatment of wastewater can adversely affect people (individuals and communities) and the environment. Stressors include any physical, chemical, or biological entity that can induce an adverse response in a receptor. Receptors include human and non-human organisms and systems (e.g., ecosystems, communities, social systems).

Micro-scale

People are the primary receptors of concern at the micro-scale, because most non-human organisms and ecosystems are best addressed at larger spatial scales (the macro-scale). The stressors of greatest concern for humans are pathogens and nitrogen in drinking water, based on feedback from the regional forums and the existence of prescriptive permitting requirements addressing those stressors.

The prominent pathway for exposure at the micro-scale is through the consumption of contaminated drinking water from a down-gradient well. The well in question may be on the same site as the treatment system or on an adjacent site.

Surface break-though of raw sewage is another potential pathway for human exposure. Pathogens are the primary stressor. Noxious odors may help limit incidental contact with raw sewage. Therefore, one might identify noxious odors as a secondary stressor and the property owners as the receptors.

The most obvious ecological receptor at the micro-scale is aquatic vegetation along waterfront property. Nutrient loading in the form of nitrogen and phosphorous is the primary stressor of concern.

Macro-scale

Pathogens and nutrients are also the focus of concern at the macro-scale (e.g., watersheds), but the receptors and pathways are more varied. The notable pathways for exposure to pathogens include contamination of municipal water supplies and coastal shellfish beds, in addition to the contamination of private wells. Adverse effects and receptors at the macro-level include illness in the general population. Hence, there are also socioeconomic risks in the form of inconvenience and reduced quality of life.

Nutrient-loading is a significant problem for many watersheds, as reflected in the establishment of Total Maximum Daily Load (TMDL) limits for key nutrients. Adverse ecological effects include the loss of pollution-intolerant species of aquatic animals.

4.1.3 Assessment endpoints

Selecting assessment endpoints and ensuring that they can be addressed within the appropriate component assessments is arguably the most critical and commonly mishandled step of problem formulation. Assessment endpoints are an explicit expression of the value that is to be protected. They consist of:

- an entity,
- a property of that entity that can be measured or estimated, and
- a level of effect on that property that constitutes an unacceptable risk.

Therefore, in addition to stating “protection of public health” as an objective, one would specify an assessment endpoint against which success or failure can be measured. For example, a public health assessment endpoint at the micro-scale might be defined as a rate of exceedance of drinking water standards due to contamination of a private well by a decentralized wastewater treatment system.

At the macro-level, an example assessment endpoint for the protection of public health might be defined as a rate of exceedance of health standards in a shellfish bed contaminated by decentralized wastewater treatment systems. An alternative endpoint might be a closure rate for shellfish beds (e.g., days per year) due to contamination by decentralized wastewater treatment systems. Note that this endpoint might also be appropriate for the assessment of economic impacts.

Criteria for selecting assessment endpoints include:

- relevance to the value to be protected,
- susceptibility to the stressors of concern, and
- relevance to public policy and management goals.

Relevance depends on the value to be protected. If it is an ecological value (e.g., a “healthy” stream system in a macro-level assessment), then the endpoint must be ecologically relevant. In this example, one might select the fish community as an endpoint entity because fish are an important part of energy transfer within aquatic systems. Similarly, exceedance of drinking water standards is directly relevant to the health of the public at-large and loss of productive shellfish beds is directly relevant to the economic value provided by shell fishing.

Susceptibility to the stressors of concern is a function of exposure and sensitivity. An assessment endpoint entity must be exposed, or potentially exposed, to the stressor of concern. Exposure is typically defined as co-occurrence or contact of the receptor with the stressor. Therefore, one must consider the likely sources, transport, and fate of the stressors when selecting an assessment entity.

For example, residents with private drinking water wells (the assessment entity) are potentially exposed to pathogens (the stressor) from up-gradient decentralized wastewater treatment systems. The frequency and magnitude of exposure must also be considered. In this example, one might further define the assessment endpoint entity as permanent residents, because they are expected to be more frequently exposed to the stressor than seasonal residents. If overloading of the system by an influx of seasonal residents is a concern, then one might establish two complementary assessment endpoints, one for permanent residents and one for seasonal residents.

Sensitivity refers to how readily the endpoint entity responds to the stressor. Sensitivity is a function of the mode of action of the stressor and the characteristics of the receptor. Mode of action typically refers to the way in which physiological mechanisms are affected by the stressor.

Characteristics of the receptor that may influence sensitivity include behavior and life-stage. For example, people with some diseases and conditions are known to consume above average quantities of water. This behavior may increase their exposure to pathogens, nitrogen, or other contaminants in private drinking water wells.

An example of life-stage influencing sensitivity can be seen in the susceptibility of babies to infantile methemoglobinemia (blue-baby syndrome) from exposure to nitrates in drinking water. Their potential sensitivity is the driver for current limits on allowable levels of nitrates in drinking water. This makes them likely candidates as assessment endpoints.

Relevance to public policy and management goals is a measure of the degree to which the assessment endpoint addresses the issues of concern to decision makers and stakeholders. The relevance (and importance) of the assessment to the decision making process depends on the relevance of the assessment endpoints. Failure to produce a relevant assessment will result in misuse or dismissal of the assessment. Selection of endpoints that are relevant to the public policy and management goals increases the probability that:

- decision makers will use the assessment and
- that the assessment will increase the quality of the decisions.

For example, if the management goal is to keep a shellfish bed open to harvesting for human consumption, then the assessment should include endpoints consistent with the local regulations for shellfish beds (e.g., specific coliform levels, etc.). If the only endpoints are those related to productivity of shellfish, then the assessment is likely to be ignored.

Ensuring that the assessment endpoints can be addressed within the appropriate component assessments (e.g., the public health risk assessment) entails 1) developing a conceptual model of the systems to be evaluated and 2) selecting measures of effects and exposure the are appropriate for the assessment endpoints and type of assessment. Those issues are addressed below.

Ensuring consistency between the component assessments and the general problem formulation phase entails iterative definition of the assessment endpoints at both levels of organization. That is, as the component assessment is developed and more is learned about the modes of exposure and effects, it may become necessary to refine the assessment endpoints in the general problem formulation. This is particularly likely for novel assessments.

It is also likely that the measures of effects and exposure will need refinement during the course of the assessment. This is why it is critical to maintain communication between the risk assessor and risk manager throughout the assessment process.

4.1.4 Conceptual model

Developing a generalized conceptual model of the systems to be evaluated entails describing and visually depicting the relationships between the stressors and the receptors. The conceptual model includes the known and expected relationships among the

- stressors,
- exposure pathways, and
- receptors (assessment endpoints).

Only those relationships which are considered in the assessment are included in the model. The supporting text should describe the assumptions used to develop the conceptual model. Relationships that cannot or will not be addressed should be clearly identified. The supporting text should include a rationale for the exclusion of prominent relationships.

For example, one might suggest that airborne transmission of pathogens from surface break through of sewage could be excluded. The rationale might be that this has been shown to be a trivial pathway or that there is insufficient scientific understanding of the processes involved to conduct a credible assessment. This is an example only and may or may not be an acceptable assumption. But the conceptual model provides an explicit statement of the assumptions and knowledge of the relationships between the stressors and receptors. This enhances communication with decision makers and stakeholders and highlights issues that need further research.

Two example conceptual models are presented below, one for the micro-level and one for the macro-level. Both examples address the transport of nutrients and pathogens. They are in the form of standard box-and-arrow diagrams. This is not the only way to present a conceptual model. For example, one could also use a diagram of a typical site, pictures or icons of sources and receptors, or a table with check marks indicating which pathways and stressors are associated with each receptor (endpoint). Any number of models could be developed, but the models must be clear and concise depictions of the relationships included in the assessment.

<u>A Micro-level</u> conceptual model for transport of nutrients and pathogens at the individual treatment system scale is presented in Figure 3. Only one ultimate source is considered in this example: a manufactured wastewater treatment system. Nutrients and pathogens may be discharged directly to the soil surface where the receptor can be exposed to the raw product. The exposure pathways for residents of the property include direct contact and inhalation of pathogens. Inhalation is conservatively included in this model (not withstanding the aforementioned rationales for excluding inhalation). Pathogens are the stressors of concern for resident receptors, but nutrients are assumed to be similarly discharged.

Discharge of the wastestream to a drainfield is the other major route for pathogens and nutrients that is considered in this example. The wastewater plume may intersect a private water supply, resulting in direct exposure to residents using the drinking-water well. Both pathogens and nutrients (nitrogen) are stressors of concern. It is recognized that the fate and transport assumptions are somewhat different for each stressor. These intricacies are addressed in the analysis of exposure, rather than in the conceptual model.

The wastewater plume may also intersect an adjacent surface water body (e.g., a lake or estuary). It is assumed that residents use this area for recreational swimming and shell fishing. Both activities are direct exposure pathways for pathogens. Nutrients (nitrogen) are not considered a stressor of concern via swimming: it is assumed that the susceptible receptors (infants) will not be swimming for significant periods of time (if any). Shellfish consumption is also assumed to be an insignificant pathway for nitrogen and infants.

Nutrients (nitrogen and phosphorous) are the stressors of concern for aquatic plants in the adjacent surfacewater body. Discharges to this area may result in accelerated growth of rooted aquatic plants and algae (eutrophication). This can have adverse effects on the people and animals that use this area.

It should be noted that more than one conceptual model can be used. This is particularly true for complex interactions. For example, one could develop a separate conceptual model for the social and ecological impacts of eutrophication. Another likely model would be one depicting the individual components of a the decentralized wastewater treatment system. It could include stressors on the system and likely points of dysfunction.

A Macro-level conceptual model for transport of nutrients and pathogens at the watershed scale is presented in Figure 4. Four ultimate sources are considered in this example: decentralized wastewater treatment systems, agriculture, a centralized wastewater treatment system, and urban run-off. This reflects the relative complexity of watershed level assessments and the need to address cumulative impacts.

In this example, direct exposure to each of the ultimate sources is not considered to be relevant or appropriate at the watershed scale. Two proximate sources are contaminated by the ultimate sources: groundwater and surface water. These proximate sources are assumed to be interconnected. Nutrients and pathogens may pass from groundwater to surface or from surface water to groundwater. That is, decentralized wastewater treatment systems can be a significant source of nutrients to surface water, even though the model assumes there is no direct link between them.

Decentralized wastewater treatment systems are assumed to discharge to groundwater via drainfields. Agricultural sources include livestock and crop farms. These sources may contaminate groundwater and surfacewater via leeching and run-off, respectively. Urban run-off is, by definition, considered to be a direct source to surface water.

Centralized wastewater treatment systems are assumed to be direct sources to surface water via permitted discharges. Groundwater infiltration is assumed to be minimal for this sewer system. While this is likely to be an oversimplification of reality, estimating infiltration along the miles of sewer pipelines was deemed beyond scope of this (hypothetical) assessment.

Groundwater is assumed to be a potential source of contamination to private and public potable water supplies. Nutrients and pathogens are stressors of concern. At the macro-level it may be appropriate to look at impacts to the local economy from an outbreak a water-borne disease. The endpoint could be lost productivity, repair costs (treating the water, supplying bottled water, etc.), reduced property values, or any other socioeconomic endpoint of concern to the stakeholders.

Contaminated surface water may impact public health, the environment, and socioeconomic endpoints. The direct pathways are the same here as for the micro-level model. However, the increased geographic scale of the macro-level model means that additional receptors may be appropriate. In this example, fish are identified as an endpoint with habitat degradation the stressor. It is assumed that widespread eutrophication could reduce the number and types of fish in the surface water system. This could have an indirect impact on socioeconomic endpoints These could include reduced quality of life (loss of recreational fishing and swimming), lost tourism, reduced property values, or any other socioeconomic endpoint of concern to the stakeholders.

4.1.5 Measures

Measures are attributes that can be estimated or measured directly. Selecting appropriate measures is critical to establishing a link between the stressor and the assessment endpoint. EPA (1998) identifies three types of measures:

- Measures of effects are measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it was exposed,

- Measures of exposure are measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint, and

- Measures of ecosystem and receptor characteristics are measures of environmental attributes that influence the distribution of a stressor (e.g., soil temperature and depth to groundwater) or receptor attributes that influence exposure and response (e.g., age and behaviors).

These measures may be direct measurements of the assessment endpoint (e.g., occurrence of an illness in the endpoint entity). A surrogate measure may be needed when direct measurement is either impossible or impractical (e.g., fecal coliform counts in drinking water).

In the example of gastrointestinal illness in residents of an individual household due to contamination of a private well by a decentralized wastewater treatment system, which was presented above in the discussion of assessment endpoint definition, one might select the following measures of effects and exposure:

For example, one might select the following measures of effects and exposure for an assessment of contamination of a private well by a decentralized wastewater treatment system:

Effects
- Drinking water standards for the protection of human health
- occurrence of symptoms diagnostic of pathogens which are likely to be associated with water contamination by domestic wastewater sources
- results of medical tests (e.g., blood tests) that support or refute contamination by domestic wastewater sources

Exposure
- fecal coliform counts in well water samples
- hydrologic conditions consistent with treatment dysfunction
- treatment system history, including maintenance and usage patterns, which is associated with dysfunction
- exposure patterns for other potential sources of similar pathogens (e.g., contaminated food)

Characteristics

- soil temperature
- exposure of sensitive age groups (e.g., children and the elderly)

These measures provide evidence supporting or refuting the conclusion that the assessment endpoint was violated. The occurrence of symptoms likely to be attributed to pathogens from domestic sewage is circumstantial evidence. Exceedance of drinking water standards and clinical tests of the receptor are direct evidence.

Each of the measures of exposure provide circumstantial evidence of exposure. The presence of fecal coliform in the drinking water is the strongest line of evidence and may even be considered direct evidence by some. The first three lines of evidence for exposure can be used to show an increased likelihood of exposure due to treatment system dysfunctions. The fourth measure of exposure, can provide evidence for alternative causes of the observed effects.

These measures are intended as examples only. In the course of an actual assessment of the scenario presented here, one might determine that these measures are unreliable or impractical. That would require the selection of new assessment endpoints or the pursuit of solutions to the technical problems through research and development.

4.1.6 Summary

In summary, the general problem formulation results are used to identify the discipline-specific assessments that are needed and to develop discipline-specific problem formulations. All four discipline-specific assessments may not be needed for a particular assessment, especially at the micro-level (individual decentralized treatment systems).

Engineering assessments of some form are needed for all assessments, because they provide the source terms of exposure for the other assessments. One might choose to stop at the engineering assessment if system reliability is the only issue and it has been defined as particular dysfunction types and rates (e.g., surface breakthrough of sewage) without specific consideration of the risks to potential receptors.

Most assessments will also include a public health assessment and, possibly, an ecological assessment. These address the receptors commonly protected by local and state regulations. Standard structures and methods for public health and ecological assessments have been developed for use in other applications, most notably the assessment of hazardous waste sites. These can be adapted for use in the field of decentralized wastewater treatment.

Socioeconomic risks are probably the most important risks from the public perspective. However, the socioeconomic risk assessment process is not as well defined as the engineering, public health, and ecological assessment paradigms. This often leads to incomplete or relatively informal socioeconomic assessments. Standard assessment paradigms (EPA, 1998) include socioeconomic issues in the endpoint selection process and the risk management process, without formally estimating the risks. A strength of integrated assessment is that it includes socioeconomic issues as part of the formal risk assessment and risk management processes.

Issues associated with development of discipline-specific problem formulations, and other discipline-specific issues, are discussed in the following sections.

4.2 Engineering Framework

The standard engineering risk assessment framework is comprised of three subcomponents: problem formulation, analysis, and risk characterization. Issues and approaches to engineering risk assessment for decentralized wastewater treatment systems are generally the same as those discussed above. Issues particular to the engineering component are addressed below

4.2.1 Objectives

The primary objectives of an engineering assessment are 1) to provide a basis for evaluating the probability of dysfunction, 2) to determine the source terms resulting from routine operations and system dysfunction for the ecological, public health, and socioeconomic components, and 3) to provide the source terms to the other components for determining the consequences, and ultimately risk, from these sources.

4.2.2 Problem formulation

The engineering problem formulation defines the scope and objectives of the engineering framework. This entails:

- identifying the spatial and temporal bounds within which the framework will be applied;
- identifying the potential modes of dysfunction and the factors that are likely to contribute to those dysfunctions;
- selecting assessment endpoints and, if they constitute source terms for other endpoints, ensuring that they are consistent with the exposure and effects models to be used in the subsequent assessments; and
- developing a generalized conceptual model of the systems to be evaluated.

Issues and approaches to problem formulation for the engineering assessment are generally the same as those for the general problem formulation. Issues particular to the engineering component are addressed below.

<u>The spatial bounds</u> of the engineering assessment for decentralized wastewater treatment systems differs from a typical engineering assessment because the environment becomes part of the system. That is, a drainfield or constructed wetland is an integral component of the treatment system, not just a part the receiving environment. This may be disputed from a regulatory perspective, but from an engineering perspective, aeration, percolation, type of soils (e.g., clay, silts, or sand), water table levels, etc., can be as important to the engineering assessment as the actual dysfunction of the manufactured components of the treatment system.

Potential factors that are likely to contribute to system dysfunction are conceptually similar to the stressors of concern in the general problem formulation. That is, they stress the system in ways that may lead to a violation of the endpoint (i.e., system dysfunction). Examples include, exceedance of system capacity, failure to properly maintain the system, periodically elevated groundwater levels, and decreased ambient temperatures. Such factors should be identified in the problem formulation and incorporated into the engineering analysis, to the extent practicable.

For example, if low temperatures are identified as a potential factor leading to insufficient microbial degradation prior to contact of the groundwater plume with a drinking water source, then transport models (qualitative or quantitative) that account for temperature should be sought during the analysis phase. If such models are not found, then this factor can be highlighted in the discussion of uncertainties and identified as an area needing further research.

Typical assessment endpoints for the engineering assessment are the expected types of dysfunction. They should include a magnitude and frequency of dysfunction (i.e., a level of effect) and a probability of dysfunction. For example, one might specify a surface breakthrough endpoint as a five-percent probability of an occurrence of detectable discharge to the soil surface.

One might also consider non-dysfunction in a similar way. This is done to provide baseline source terms for normally operating treatment systems. That is, the system may be operating within the established norms, but the discharge may contain stressors at levels that pose a risk to public or ecological receptors. This is especially true when considering the aggregate source term from multiple treatment systems.

4.2.3 Analysis

The engineering analysis subcomponent identifies the technical issues associated with the modes of dysfunction. This entails calculating or qualitatively estimating the probability of each type of dysfunction. This also includes predicting the fate and transport of wastewater constituents in the environmental components of the treatment system (e.g., on-site soil, groundwater, and constructed wetlands).

Each item in the treatment system is considered in relation to its likely modes of dysfunction, the probability of occurrence, and the effects of the dysfunction. Each item, its function, and potential mode of dysfunction is listed, followed by other relevant information. Failure mode and failure cause include the physical or operational description of the manner in which a failure of the component occurs and an evaluation of the probable cause(s) of dysfunction. These causes include chance, over-stress, improper aeration or percolation in the soil, etc. Failure mode frequency provides a quantitative estimate of the frequency of each failure mode described. This may be limited to qualitative estimates (e.g., low, moderate, or high), because of a lack of data. Estimating fate and transport within the environmental components of the treatment system entails using qualitative and quantitative models that account for significant environmental factors. These may include groundwater transport models, viral transport models, and biodegradation models.

4.2.4 Risk characterization

The engineering risk characterization subcomponent is where the probabilities of dysfunction for each individual element of the treatment system are combined to provide an overall estimate of the magnitude and likelihood of dysfunction.

The output from this process provides the inputs (source terms) to the ecological, public health, and socioeconomic assessments. Therefore, care must be taken to ensure that the dimensions of the engineering risks are consistent with the exposure and effects models to be used in the subsequent assessments. For example, the best available public health exposure-effects model may require estimated viral counts, rather than fecal coliform counts. Or the ecological exposure-response models may require, at a minimum, seasonal nutrient loading rates, rather than total annual nutrient loading rates.

This is also where uncertainty in the predicted failures and source terms is captured. That may be as simple as identifying and describing the sources of uncertainty or as sophisticated as quantitatively estimating the variability and uncertainty via Monte Carlo analysis. However it is done, the purpose of this process is to ensure that uncertainties associated with the engineering assessment are not over-looked in the other discipline-specific assessments.

4.3 Ecological Framework

The standard ecological risk assessment framework is comprised of four subcomponents: problem formulation, exposure assessment, effects assessment, and risk characterization. Issues and approaches to ecological risk assessment for decentralized wastewater treatment systems are generally the same as those discussed above. Issues particular to the ecological component are addressed below

4.3.1 Objective

The primary objective of the ecological assessment is to aid environmental decision-makers faced with selecting among alternative treatment technologies to ensure that risks to ecological receptors are properly considered.

4.3.2 Problem formulation

As with the general problem formulation, ecological problem formulation includes identifying the spatial bounds of the assessment, identifying the stressors and receptors, selecting assessment endpoints, and developing a conceptual model of the ecosystem at risk.
The spatial extent of the assessment will largely determine the types of assessment endpoints that are appropriate. For very narrowly defined assessments, an ecological assessment may not be necessary. This may occur when evaluating a single system which does not discharge to susceptible environmental systems (e.g, a suburban site with no surface water resources in the immediate vicinity). An ecological assessment is more likely to be included in macro-level applications (e.g., watershed assessments).

Ecological assessment endpoints are typically limited to populations and higher levels of organization (e.g., communities, ecosystems, etc.). The relevant spatial scale for these endpoints is larger than the typical micro-level assessment (i.e., individual decentralized treatment systems). Exceptions may include impacts to natural wetlands and impacts to ecological entities that result in adverse effects to the public (e.g., excessive growth of noxious aquatic vegetation). Again, it is critical that the assessment endpoints and measures of effects and exposure be consistent with those identified in the general problem formulation.

4.3.3 Analysis

Ecological analysis is comprised of two parallel processes:

- exposure assessment identifies the technical issues associated with the pathways and mechanisms of exposure for the selected receptors;
- effects assessment identifies the technical issues associated with the exposure-response modes of the selected receptors.

Exposure and exposure-response models that address these issues are identified, evaluated for their ability to support the objectives of the ecological framework, and used to generate estimates of exposures and effects.

These assessments are performed in parallel with much effort devoted to ensuring consistency of the models and estimates. For example, the exposure model may estimate the concentrations of nutrients entering an aquatic system from one or more treatment systems and the exposure-response model may estimate the rate of growth of nuisance vegetation at various nutrient concentrations.

Each of these model types may generate uncertainty estimates (error terms) which can be used in the characterization of risks to quantify uncertainty in the predicted risks.

4.3.4 Risk characterization

The ecological risk characterization integrates the exposure models and effects models to provide an estimate of the magnitude and likelihood of adverse ecological effects (i.e., risk). This may be in the form of a qualitative estimate (e.g., low, medium, or high probability) or a quantitative estimate, depending on the data and models that are available.

Where practical, estimates of the risk should be in the form of a probability function for varying magnitudes of adverse effects. This may only be possible for stressors with extensive exposure-response data sets (i.e., chemical toxicants).

It is expected that multiple lines of evidence will be available for most assessment endpoints (e.g., measured stressor concentrations in water, measured habitat characteristics, and measured fish community characteristics). In these instances, it is generally not possible to calculate an overall estimate of risk. Instead, risks are characterized by a weight-of-evidence process, which involves determining whether or not the assessment endpoint was exceeded and what factors account for apparent discrepancies in the results, based on each of the available lines of evidence.

As for the engineering risk characterization, this is where uncertainty in the predicted ecological risks is captured. This is to ensure that the ecological uncertainties are not lost in the final assessment. That is, uncertainty characterization provides the context within which the estimates of risk are evaluated. As with the estimates of risk, uncertainty characterization may be qualitative or quantitative, depending on the available exposure and assessment data.

4.4 Public Health Framework

The public health assessment is comprised of four subcomponents: hazard identification, public exposure, health effects, and risk characterization. Many of the issues and approaches to public health assessment for decentralized wastewater treatment systems are similar to those discussed in the proceeding sections. Issues particular to the public health component are addressed below.

4.4.1 Objective

The primary objective of the public health assessment is to aid environmental and public health decision-makers, who must select among alternative treatment technologies to ensure microbial and chemical risks does not exceed public health protection standards.

4.4.2 Hazard identification

Public health hazard identification involves developing a conceptual model of potential exposure pathways associated with decentralized wastewater treatment systems.

Public health assessments are appropriate at both the micro-level (individual systems) and the macro-level (watersheds). This is because health protection is intended to extended to individual members of the public, not just to the overall population. Therefore, even micro-level assessments are relevant to the spatial requirements of the assessment endpoint entities (e.g., residents using private wells).

Assessment endpoints may also be defined as segments of the population that are highly exposed or highly sensitive to the stressors of concern. As in the examples presented in the general problem formulation, these may include persons with particular physiological conditions or those in a particularly sensitive stage of life (e.g., infants exposed to nitrates). Pathogens are likely stressors of concern, in addition to chemicals and nutrients, which are also stressors of potential ecological concern.

4.4.3 Analysis

The analysis phase of the public health assessment consists of a public exposure assessment and a health effects assessment. This process is conceptually similar to the ecological exposure and effects assessments:

- the public exposure assessment identifies the technical issues associated with the pathways and mechanisms of exposure for the selected receptors;
- the health effects assessment identifies the technical issues associated with estimating the impacts of microbes (i.e., protozoa, bacteria, viruses) and chemicals (e.g., nitrates) on humans; and
- both assessments are performed in parallel to ensure consistency of the models and estimates.

4.4.4 Risk characterization

The public health risk characterization integrates the public exposure and health effects models. The results are either qualitative (e.g., low, medium, or high probability) or quantitative estimates of risk, depending on the data and models that are available.

To the extent practical, estimates of the risk should be in the form of a probability function for varying magnitudes of adverse effects. This may only be possible for stressors with extensive exposure-response data sets (i.e., chemical toxicants).

This subcomponent should also identify and evaluate approaches for comparing chemical and microbial risks, balancing the respective risks, and assessing the relative risks of drinking water in the context of other risks to public health.

The risk characterization also should present the qualitative and quantitative estimates of uncertainty, as is practical for the available data and models.

4.5 Socioeconomic Framework

Socioeconomic risk assessment is the least well formalized of the four disciplines presented here. The approaches used to evaluate social and economic impacts are quite varied because of the wide variety of impacts that are evaluated. However, one can still organize the general process into problem formulation, analysis, and risk characterization components.

4.5.1 Objective

The primary objective of the socioeconomic assessment is to ensure that social and economic issues are properly considered when selecting among alternative wastewater treatment technologies.

4.5.2 Problem formulation

This discipline arguably addresses the widest range of issues associated with wastewater treatment. They are often the most important to the stakeholders (i.e., homeowners and local community). Potentially contentious issues include:

- costs of treatment systems,
- fairness of financial burdens,
- property rights,
- economic impacts to communities, and
- land-use planning.

Applying principles of risk assessment may be particularly helpful in assessing socioeconomic risks. Risk principles can be used to bring these issues forward so that they can be properly addressed in the assessment and management processes. That is, one should:

- clearly define the endpoints (i.e., include an entity, attribute, and level of effect);
- develop a conceptual model;
- select measures that link the stressor to the endpoints;
- ensure that these endpoints and measures are consistent with the management objectives;
- use models and estimates of effects that are consistent with the selected endpoints; and
- capture the uncertainties associated with the risk estimates.

These general principles are discussed in the Principles of Risk Assessment and High-Level Framework sections. Issues particular to socioeconomic risk assessments are discussed below.

<u>Assessment endpoints</u> for social and economic risks are conceptually similar to more traditional endpoints. They include an entity, attribute, and level of effect (when appropriate). The entity is often implied in socioeconomic evaluations. That is, the attribute may be specified but not the entity.

For example, one might evaluate the cost of treatment systems. Cost is an attribute. The entity is the group or individual that pays the costs. While this is obvious, specifying which group is paying can help clarify the goals and obstacles to the assessment. In this example, one might specify two endpoints:

- the costs to all property-owners in a watershed
- the costs to owners of newly developed property in a watershed.

Together these endpoints address the underlying issue of fairness of the financial burden. That is, are owners who are seeking new building permits forced to use more expensive systems because the existing systems in the drainage area are ineffective? This amounts to a subsidy of less effective and less expensive systems by the new property-owners.

Socioeconomic attributes are not limited to monetary costs. They also include non-monetary costs, such as convenience, aesthetics, and intrusiveness (e.g., maintenance personnel inspecting private treatment systems). Monetary and non-monetary costs are discussed in detail in the associated issue paper on socioeconomic issues.

One issue raised in the regional forums was the difference between risks and benefits. It was argued that risks could include the loss of benefits. This is implicit in many statements of risk (e.g., decreased aesthetics due to algal blooms).

However, this may be awkward for some benefits. For example, one potential benefit of decentralized systems is the ability to reuse grey water (non-septic waste water), especially in arid regions were water costs are high. Rephrasing this as "the risk of not decreasing the costs of water usage" is awkward and unclear. It is preferable to retain the positive (benefit) wording. This means one must specify in the problem formulation that both risks and benefits are considered in the assessment. However, complicated explanations as to the distinctions between risks and benefits are not especially helpful.

Measures in traditional assessments include measures of exposure and measures of effects. This terminology may be confusing for socioeconomic assessments. It seems more appropriate to think in terms of costs (risks) and benefits. As noted above, costs may be monetary or non-monetary in nature and benefits may be the reduction of risks. These terms are not conceptually parallel to exposure and effects. However, it is the costs and benefits that should be estimated or measured in a socioeconomic assessment.

4.5.3 Analysis

The socioeconomic analysis subcomponent identifies the technical issues associated with these risks and benefits. This entails calculating or qualitatively estimating each type of cost (risk) or benefit. The socioeconomic analysis draws heavily upon the engineering assessment. System costs are directly related to the types of treatment systems, types of failures, and the required amount of monitoring and maintenance.

Monetary costs (risks) are often the easiest to explain and quantify. Money is a scale to which all members of the public can relate. This is one reason economic issues are often the most important and contentious issues risk managers must confront.

Direct monetary costs are conceptually straight forward. They include items like the costs of installing and operating a treatment system. These costs can be estimated using project planning and evaluation programs (e.g., COSMO (ref)). Such programs allow one to specify the assumptions used to estimate costs in detail. For example, the assumed labor and material costs can be based on local and up-to-date estimates.

These programs make it relatively easy to generate estimates for alternative treatment systems. This can make the assessment process more transparent: all assumptions for each alternative can be presented and compared. However, transparency should not be confused with clarity. How clear these analysis are depends on how they are presented to the decision -makers.

Indirect monetary costs may be more difficult to quantify than direct monetary costs. These include costs (risks) such as the loss of economic growth due to restricted land use and loss of tourism dollars due to social and environmental changes.

For example, unrestricted development may diminish the appeal of a resort community. If tourism decreases, so do the economic benefits of tourism. However, there are also costs of tourism and benefits of general economic development. Estimating the likely changes in these areas can be very complicated. Transparency is the primary advantage of using risk assessment principles when evaluating these costs. All of the assumptions used to estimate the costs should be explicitly presented as part of the analysis of risks.

Non-monetary costs (risks) and benefits may be converted to a monetary scale. For example, one could establish the dollar value of a weed-free lake to lake-front residents. This could be the amount people are willing to pay to keep it weed-free or the amount they are willing to be compensated for not controlling weed growth. The amount is often determined by conducting surveys. These surveys may be of a specific community or a representative cross section of a larger community.

However, there are significant ethical and technical problems associated with any such valuation method (see Socioeconomic Issue Paper). These issues must be identified and discussed in the analysis phase. Any associated uncertainties should be carried forward into the characterization of risks.

Non-monetary risks and benefits can also be addressed directly, without converting them to a monetary scale. For example, one could rank the relative importance of each potential risk (cost) and benefit. Such relative rankings are not quantitative. That is, the issue ranked number one is not necessarily twice as important as the issue ranked number two. Other non-monetary approaches are presented in the Socioeconomic Issue Paper.

4.5.4 Risk characterization

Risk characterization entails combining the monetary and non-monetary costs (and benefits) estimated in the risk analysis section. This includes ensuring the compatibility of those estimates and presenting the major sources of uncertainty in those estimates.

Integrating monetary and non-monetary costs into a decision making framework can pose significant technical challenges. One must ensure that the estimated risks and benefits are comparable. That is, the risks and benefits must be for the same:

- endpoint entities
- spatial scales, and
- temporal scales.

For example, the costs of individual treatment systems may be borne by a relatively small group of homeowners. The benefits may be realized by a much larger group of stakeholders. In this case, all members of the community might benefit from preserving the aesthetic appeal of a small community. How this disparity is handled can significantly affect the decision-making process.

How uncertainties are addressed can also significantly affect the decision-making process. It is important that all major sources of uncertainty are clearly and concisely presented. This is particularly important for socioeconomic assessments, because this discipline:

- is the least well-developed of the four risk assessment disciplines presented here and
- addresses issues very familiar to most decision-makers.

The assumptions inherent to the assessment may be based on relatively little technical information (data). That leaves them open for discussion and many interested stakeholders will have strong opinions about those assumptions. The best way to support the decision -making process is to explicitly identify the assumptions used and any information that supports or refutes those assumptions.

4.6 General Risk Characterization

4.6.1 Objective

The primary objective of the general risk characterization is to integrate the results of each component assessment into a cohesive evaluation of the risks to all of the selected assessment endpoints.

4.6.2 Approach

Two general approaches are appropriate for integrating the risks from multiple component assessments:

- mathematically propagating risks across disciplines and
- logically weighing the evidence of risks from each discipline.

Mathematical propagation is possible when quantitative estimates of risk are calculated within each component assessment. For example, it is standard practice to calculate the probability of human health effects as a rate of incidence. It is also standard practice to calculate engineering failure rates. It is theoretically possible to combine these calculations to get the probability that system dysfunction will lead to a particular health effect. This mathematical approach will probably not apply to most endpoints for decentralized wastewater treatment systems.

The integration of risks for most endpoints will be based on a variation of the weight-of-evidence process. The standard weight-of-evidence process entails logically evaluating several independent lines of evidence for a given endpoint (EPA, 1998; Suter, Efroymson et al., 2000).

A line of evidence is any model, test, or observation that can be used to estimate the magnitude or likelihood of risks. Examples include models yielding system failure probabilities, tests indicating the response of aquatic plants to increased nutrient loading, and observations indicating changes in aquatic communities. These lines of evidence may be used to characterize the risks of decentralized wastewater treatment systems damaging aquatic ecosystems.

However, one could not simply multiply the probabilities for each line of evidence to get the overall probability of impacts. Instead, one must logically evaluate each line of evidence to see how it supports or refutes the theory that decentralized wastewater treatment systems pose a risk to aquatic ecosystems. This entails weighing each line of evidence based on one or more of the following criteria, which are adapted from Suter et al. (2000):

- Relevance -Is the available information (models, data, observations, etc.) relevant to the types of treatment systems, environmental settings, and receptors being evaluated?

- Exposure/Response - Does an increase in the stressor of concern lead to an increase in the response of the endpoint (e.g., increasing installation costs results in increased aversion to alternative systems by home-owners)?

- Temporal scope - Does the information address important variations with time (e.g., depth to water table during wet season, seasonal use of vacation homes)?

- Spatial scope - Does the information adequately address the area(s) to be evaluated (e.g., includes sensitive water resources, macro-level models not used for micro-level evaluations)?

- Quality - Was the information generated using appropriate quality assurance and control procedures (e.g., appropriate analytical procedures were used, personnel installing the system were properly trained)?

- Quantity - How much information is available for a given system or circumstance (e.g., number of treatment systems tested)?

- Uncertainty - How reliable is the information in terms of estimating risks (e.g., estimated viral densities varying by several orders of magnitude, estimated costs within twenty percent of actual)?

After each line of evidence is evaluated, they are compared to each other and explanations are sought for any apparent inconsistencies. For example, public reaction to a proposed plan for using decentralized wastewater treatment systems might be more negative than was expected, based on the estimated costs. One possible explanation is that important costs (monetary or non–monetary) were not included.

Risk characterization is also the stage at which the effects of multiple stressors are evaluated. This can be done to varying degrees, depending on the types of stressors, effects, and data. One of the strengths of the weight-of-evidence process is that comparing the results for multiple lines of evidence can help elucidate the relative importance of multiple stressors. However, distinguishing the effects attributable to individual stressors, or the combined effects of multiple stressors, is complicated and difficult. This issue continues to be a focus of research in the environmental sciences.

The degree of quantification that is possible in the general risk characterization will be limited by the degrees of quantification for each discipline-specific risk characterization for a particular assessment endpoint. For example, the public health component might estimate a risk of 1 x 10-4 for a particular exposure, but the engineering component may only be able to estimate the probability of dysfunction as high.

The degree of quantification of risks in each component framework will also determine how uncertainty can be addressed in the general characterization of risks. For example, if the component assessments provide qualitative risk estimates (e.g., low, medium, or high), then the uncertainties should be identified and discussed in the form of an uncertainty narrative. Quantitative methods of risk estimation will yield quantitative estimates of uncertainty.

Sensitivity analysis can be used to identify the factors and issues that have the greatest effect on the final conclusions of the assessment, provided the models and data are of sufficient quality. This process entails systematically manipulating input variables and measuring the changes in the final estimates of risks. The results may be used to refine and improve the models and data sets for the most sensitive factors. This may entail collecting more or higher quality data or conducting additional model research and development.

Sensitivity analysis is most appropriate where quantitative models are used to predict exposure or effects (e.g., fate and transport models). It is possible to apply this method to qualitative estimates of risks by changing the assumptions and re-evaluating the information. However, this can be very time consuming and may be limited to only a few iterations for a given endpoint.

4.7 Risk Management

Risk management is the process of deciding which actions to take in response to a risk. It entails considering the results of the risk assessment along with other factors explicitly excluded from the assessment of risks.

4.7.1 Objective

The primary objective of risk management is to balance the risks to all endpoints and parties of concern.

4.7.2 Approach

Risk management is the final component of risk-based decision making. This is the stage at which all potentially important issues are considered, including the:

- magnitude and uncertainty of the estimated risks (including monetary costs and social impacts),
- risks of not taking any action,
- benefits of each potential action, and
- ethical and political considerations of each action

Risk management is a subjective process: it includes the personal opinions of the decision-makers and stakeholders. Although risk assessment is often based (in part) on professional opinions, these must be justified based on the available evidence. Personal opinions require no formal justification. Thus, risk management is subject to the will of the decision-makers.

Risk management entails setting management goals and using risk management methods to make decisions that achieve those goals.

Risk management goals are specific management objectives. They typically consist of a risk (or benefit) and the desired result of managing that risk (or benefit). For example, one might set a goal of reducing the risk of eutrophication to levels that are acceptable to the public (including regulatory agencies and the affected communities). Another goal might be to balance the risks of adverse impacts (e.g., risks to public health and the environment) against the risk of intrusion onto personal property (e.g., professional maintenance and operation of treatment systems).

Two related issues raised at the National Research Needs Conference are worth noting. The first is that one general management goal should be the development of treatment systems that require “reasonable” levels of maintenance at a “reasonable” cost, rather than “cheap” systems that require “little” maintenance. The second issue is that we as a society need to consider how much risk we’re willing to accept in order to have the freedom to chose a less-expensive system. These are both risk management issues that affect the selection of assessment endpoints and methods and research goals and objectives.

Management objectives and goals should be established prior to conducting the risk assessment (i.e., during the general problem formulation phase). It is these goals and objectives that the assessment is intended to support. If they are not clearly defined, then it is unlikely that the assessment will provided the information needed by the decision-makers.

Risk management methods are the tools and techniques used to balance the perceived risks and benefits. Public decision making is still generally done ad hoc, rather than via sophisticated decision analysis tools. Examples of rigorous decision-making tools include the WARMF and MANAGE models. Such tools compel one to carefully evaluate each piece of information independently and consistently.

In the absence of formal decision support tools, care must be taken to ensure that the decision-makers are provided with a clear and concise list of issues that were critical to the conclusions of the assessment. These include an explicit list of endpoints that were and were not at risk and the degree of certainty of those conclusions.

The risk assessors should participate in the risk management process as technical advisors. They should provide assistance in interpreting the results of the assessment and developing plans to acquire new information or conduct additional assessments. This includes identifying critical data gaps and indicating how likely it is that additional research could fill those gaps.

5 EVALUATION OF RESEARCH NEEDS

The National Research Needs Conference will identify and evaluate the key issues for the field of decentralized wastewater treatment. This conference is analogous to the problem formulation phase of an effort to assess the risks and benefits of decentralized wastewater treatment systems. It entails identifying and defining the:

- problems,
- scope of the research efforts,
- objectives,
- assessment endpoints, and
- management goals.

It also includes a general plan identifying which issues will be pursued and how they will be pursued. This plan is analogous to the development of an analysis plan.

5.1 Objective

The ultimate objective of this conference is to identify the most important research and development issues so that they can be addressed by the National Capacity Development Project (NCDP) or other organizations.

5.2 Approach

The risk-based approach was selected to avoid the pitfalls of undirected research and development. The most notable pitfall is identifying projects based solely on academic interest or tractability of the problem. The undirected approach may lead to understanding of the processes and issues. However, it does not necessarily lead to better management of the field of decentralized wastewater treatment. This includes management of research and development resources and management of the risks and benefits.

The risk-based approach to management of NCDP research and development is depicted in Figure 5. It consists of seven steps, which are discussed below. The risk-based approach will be used to guide the prioritization and review process. Each potential research topic will be evaluated regarding the degree to which it supports high-priority objectives and goals.

This approach is inherently an iterative process. As research needs are identified and met, it will be necessary to re-evaluate the research and development needs for distributed wastewater treatment. For example, the results of a particular research project may lead to a better definition of the acceptable risk associated with a given issue. This process will also help ensure that risk assessment is incorporated into the interpretation of the research results.

5.2.1 Identify objectives

The first step is the identification of objectives. Objectives are general statements, such as protection of public health from consumption of contaminated water. The identification of objectives is based the results of the regional forums and professional judgement.

5.2.2 Select endpoints and goals

The second step is selecting a set of assessment endpoints and management goals that support each general objective. Endpoints and goals are more specific than objectives.

Each objective has at least one assessment endpoint or management goal. The maximum number of endpoints and goals per objective is limited only by the feasibility of adequately addressing them. More than four or five per discipline for each objective is probably not practical.

Assessment endpoints include a specific entity and its attributes. Endpoints represent the values to be protected by an objective. The endpoints may be associated with any of the four disciplines that are part of the integrated assessment:

- engineering,
- ecology,
- public health, and
- socioeconomics.

Selection of assessment endpoints is discussed in more detail above in the high-level framework problem formulation section.

Risk management goals are similar to risk assessment endpoints: they are more specific than objectives. They typically consist of a risk (or benefit) and the desired result of managing that risk (or benefit). For example, balancing the impacts on future land use, the costs of installation and management, and the risks of water-borne illnesses.

5.2.3 Select measures and methods

The third step is selecting measures for each assessment endpoint and methods for each management goal. It is these measure and methods that are the focus of research and development efforts.

Measures are attributes of the endpoint that can be observed or estimated. This may require selecting surrogates for endpoints that can not be measured directly. Measures include: measures of exposure, measures of effects, and measures of endpoint characteristics (see the high-level framework problem formulation section).

Management methods are conceptually similar to measures for the field of risk management. They are the specific processes that would improve risk management for decentralized wastewater treatment systems (e.g., geographic information systems that improve the decision-making process).

5.2.4 Prioritize measures and methods

The fourth step entails classifying each measure and method based on three general criteria:

- importance to risk assessment or management,
- uncertainty of our current understanding of the measure or methods, and
- tractability of the associated research and development needs.

Prioritization is based on collective professional judgement. This consists largely of the considered opinions of selected experts in key fields of study. These opinions are presented in the other invited issue papers. The issues raised in those papers will be peer reviewed by the participants of the National Research Needs Conference.

Specific criteria for prioritizing the potential research topics were adapted from EPA (1996). They consist of the following questions, which are asked of each measure or method, as appropriate.

- What type of effect would be investigated?
- How severely might this effect impact the treatment of wastewater?
- How severely might this effect impact public health, ecosystems, or social systems?
- How severe are the potential economic impacts?
- Is this effect an immediate or long-term concern?
- How easily can the effect be reversed?
- What level of human, ecological, social, or economic organization would be impacted?
- How geographically extensive are the potential impacts ?
- How broadly applicable is the proposed model or method?
- To what extent will the proposed model or method facilitate or improve risk assessment or risk management?
- How large is the proposed user community for the proposed model or method?
- If risk management options currently exist, are they acceptable to stakeholders, implementable, reliable, and cost-effective?

- Could new or improved technical solutions prevent or mitigate the risks ways that are acceptable to stakeholders, implementable, reliable, and cost-effective?

- Are other research organizations currently investigating this issue or are they interested in working in partnership with the NCDP on this issue?

The final task of the prioritization step is to summarize answers to the above questions and provide a rank for each measure and method. To the extent possible, this summarization should be captured in a "quantitative" scoring system, in which each measure or method is assigned a score for each criterion (e.g., from 1 to 5) and the scores are summed across all criteria for each measure or method. This may require that answers to the 14 questions above be consolidated into a more manageable number of criteria. This concise list of criteria should include, at a minimum, the three general criteria presented above (i.e., importance, uncertainty, and tractability). Although the assigned scores are still subjective, this approach helps ensure that each criterion is applied consistently to each method or measure. The transparency of this approach will also lead to greater consistency among individuals assigning scores for each criterion.

5.2.5 Recommend for consideration

The fifth step entails recommending which measures warrant further research and development at this time. These recommendations will be based on professional judgement and may include nominations by the participants of the National Research Needs Conference. If a measure is not recommended for further research and development, then it should be identified as being of low priority.

5.2.6 Recommend for consideration by NCDP

The sixth step entails recommending which of the measures recommended for further research and development should be pursued by the NCDP. These recommendations also will be based on professional judgement and may include nominations by the participants of the National Research Needs Conference.

If a measure is not recommended for further research and development by the NCDP, then it should be identified as warranting further consideration by other organizations or institutions. For example, it was determined in the regional forums that the NCDP would not consider the appropriateness of existing effects thresholds for viruses and other pathogens. If it is decided that this issue warrants further investigation, then it should be identified as such and passed on to appropriate agencies.

5.2.7 Define research goals

The seventh and final step entails defining the research and development goals for the measures and methods recommended for further consideration by the NCDP. This step is intended to ensure that research and development efforts are focused on goals that support the risk-based decision-making process. Doing this for all selected measures may be beyond the scope of the National Research Needs Conference. Efforts should be focused on those measures considered by the NCDP to be of the highest priority.

The development and explanation of these goals are presented in the other four issue papers. They are captured as the rationales and justifications for considering each issue to be of high-priority.

5.3 High-Priority Issues

One of the objectives of this paper is to identify several high-priority issues for decentralized wastewater treatment system. These are issues that were consistently raised in the three regional forums. The prioritization criteria mentioned above were not explicitly used in the forums. However, many of those criteria were implicitly considered in the topical discussions from which these issues were drawn. Commonly identified high-priority issues include:

- Baseline efficacy of standard and alternative treatment systems
- Baseline failure rates (standard and alternative systems)
- Impact of maintenance on performance (standard and alternative systems)
- Waste-stream characterization (constituents, loading rate, waste strength, seasonal usage)
- Impact of constituents (household chemicals, antibiotics) on treatment systems
- Disinfection by alternative systems
- Pathogen transport in soil systems (especially viruses)
- Nutrient transport in soil systems (nitrogen and phosphorous)
- Minimum depth to watertable (variation with soil type and seasonal fluctuations)
- Impact of soil temperature on treatment efficacy

- Direct costs of alternative treatment systems (installation and maintenance)
- Non-monetary costs (standard and alternative systems)
- Equity of financial burden among users
- Acceptability of performance-based permitting
- Political barriers (current system is defacto zoning)

- Cumulative impacts at the macro-level (multiple sources and stressors)
- Land use (urban sprawl, invasion of pristine environments)

6 CONCLUSIONS

Risk assessment is more formal than other approaches to evaluation and decision making. It ensures that prescribed decisive effects are estimated and that uncertainties are considered.

Thus, the resulting products are more defensible and compelling. The bottom line is that the results will be more influential because they will be relevant to the concerns of decision makers and stakeholders.

If a risk-based approach to decision making is to be successful, the risk assessment process must be taken seriously. Unguided research or monitoring will not produce useful risk estimates. That is, any program intended to provide risk estimates must be guided by a risk assessment process consisting of:

- clear and acceptable endpoints,
- a logical structure,
- useful output, and
- estimates of effects and uncertainty.

Of particular note is the need for well-defined assessment endpoints. This requires all involved to clearly define the problems of interest. That is, the assessment endpoint must specify the property (e.g., entity and attribute) and a level of effect to be considered. Nebulous statements like protecting the quality of the environment or the quality of life are insufficient. Also, selected assessment endpoint properties must be measurable, either directly or by proxy (e.g., measures of exposure and effects).

Integrated risk assessment for the field of decentralized wastewater treatment can be accomplished. A prototype framework was presented. It consists of separate risk assessment and risk management sections. The risk assessment section includes subcomponents for each of for major disciplines: engineering, ecology, public health, and socioeconomics. Including a socioeconomics subcomponent means that costs (monetary and non-monetary) are explicitly included in the risk assessment section. These and other factors (politics, ethics, etc.) are also addressed in the risk management section.

A process for prioritizing research and development issues was presented. It is results in a risk-based program, rather than an unguided research and development program. It includes:

- clear and acceptable objectives,
- explicit assessment endpoints and management goals,
- explicit measures and management methods, and
- a consistent basis for prioritizing issues of concern.

The principles captured in this process were used to identify several high-priority issues. This was accomplished during the regional forums and in the issue papers commissioned for the research needs conference.

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FIGURES

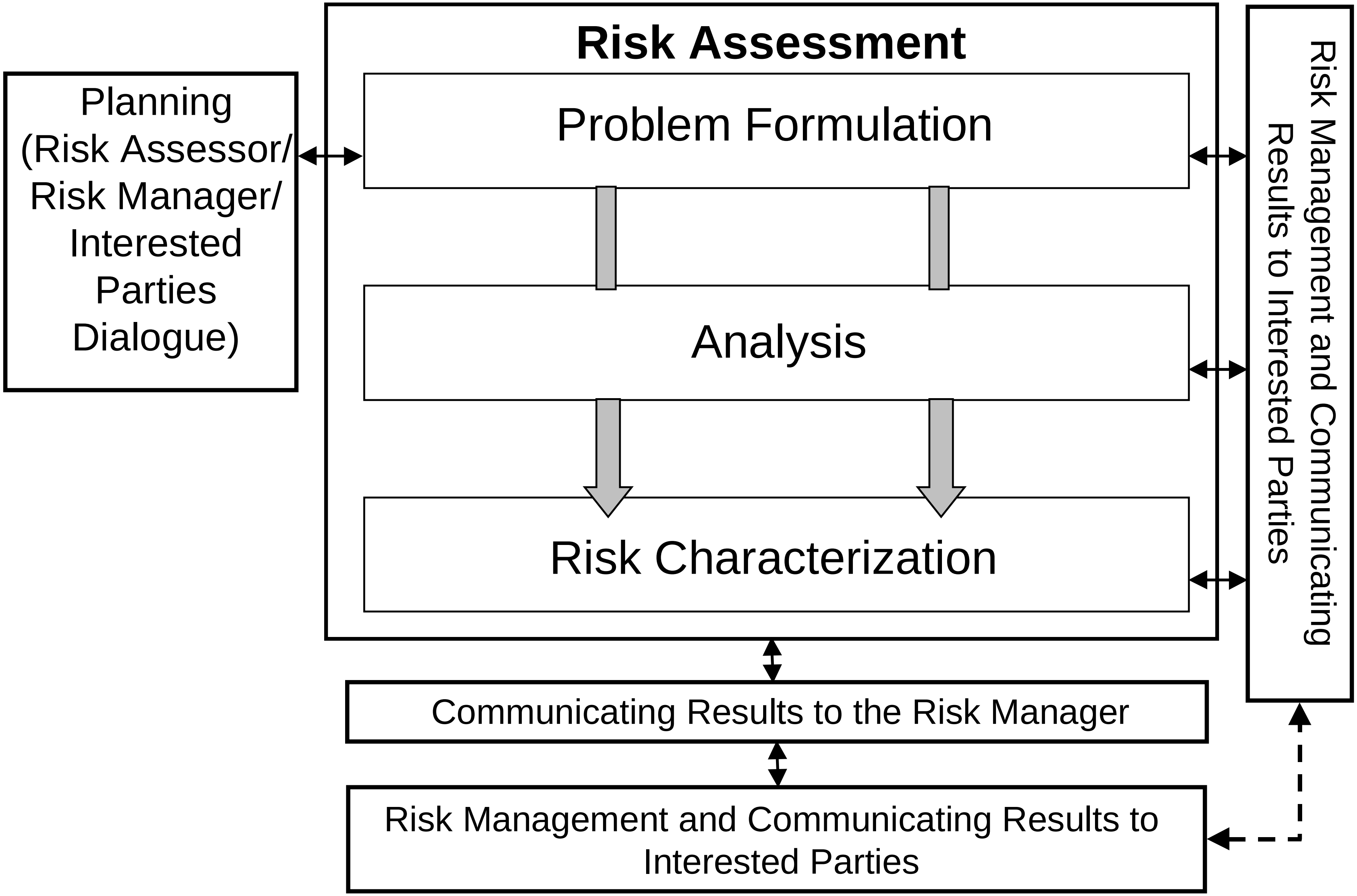

Figure 1. Standard risk assessment framework

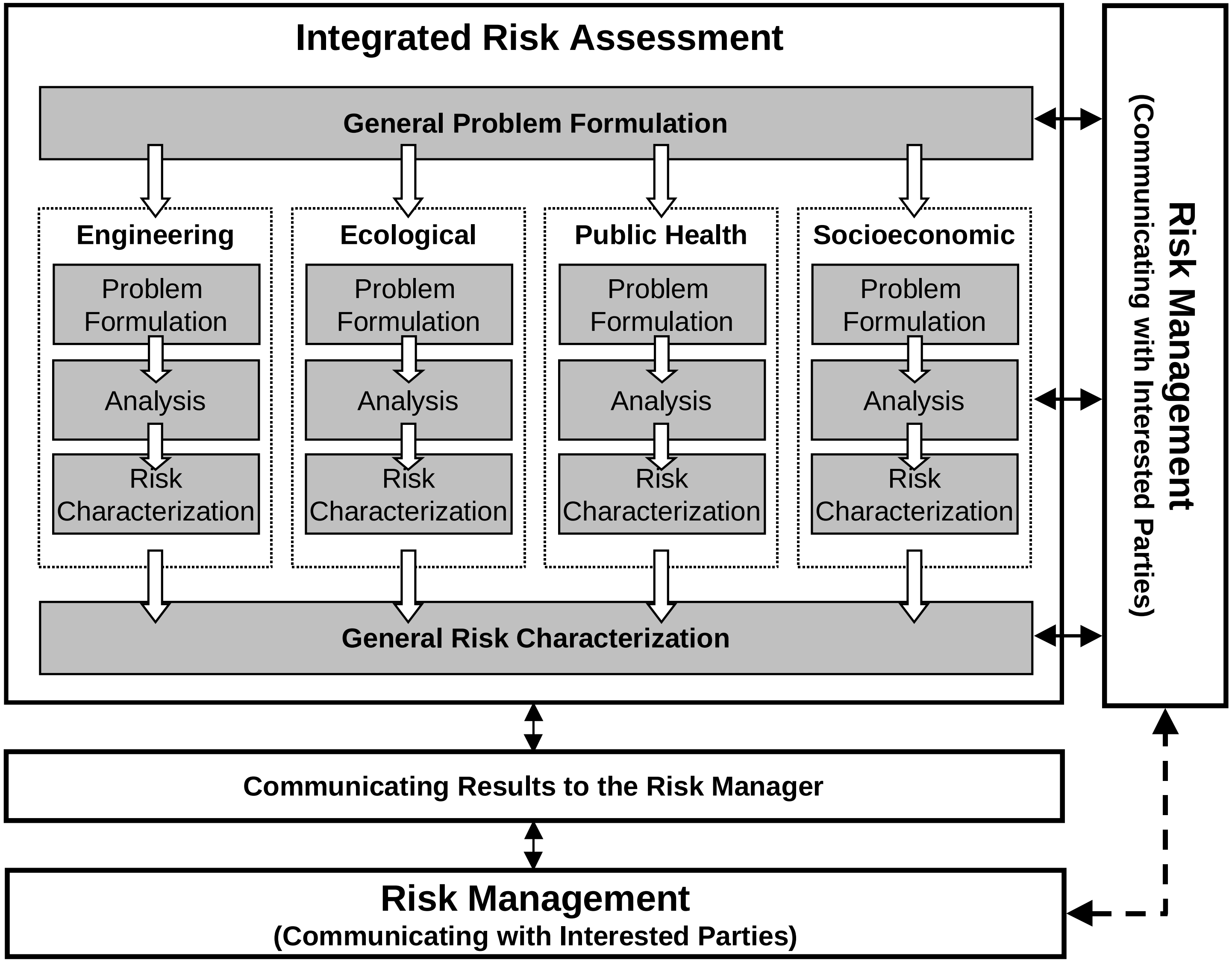

Figure 2. Integrated risk assessment framework for decentralized wastewater

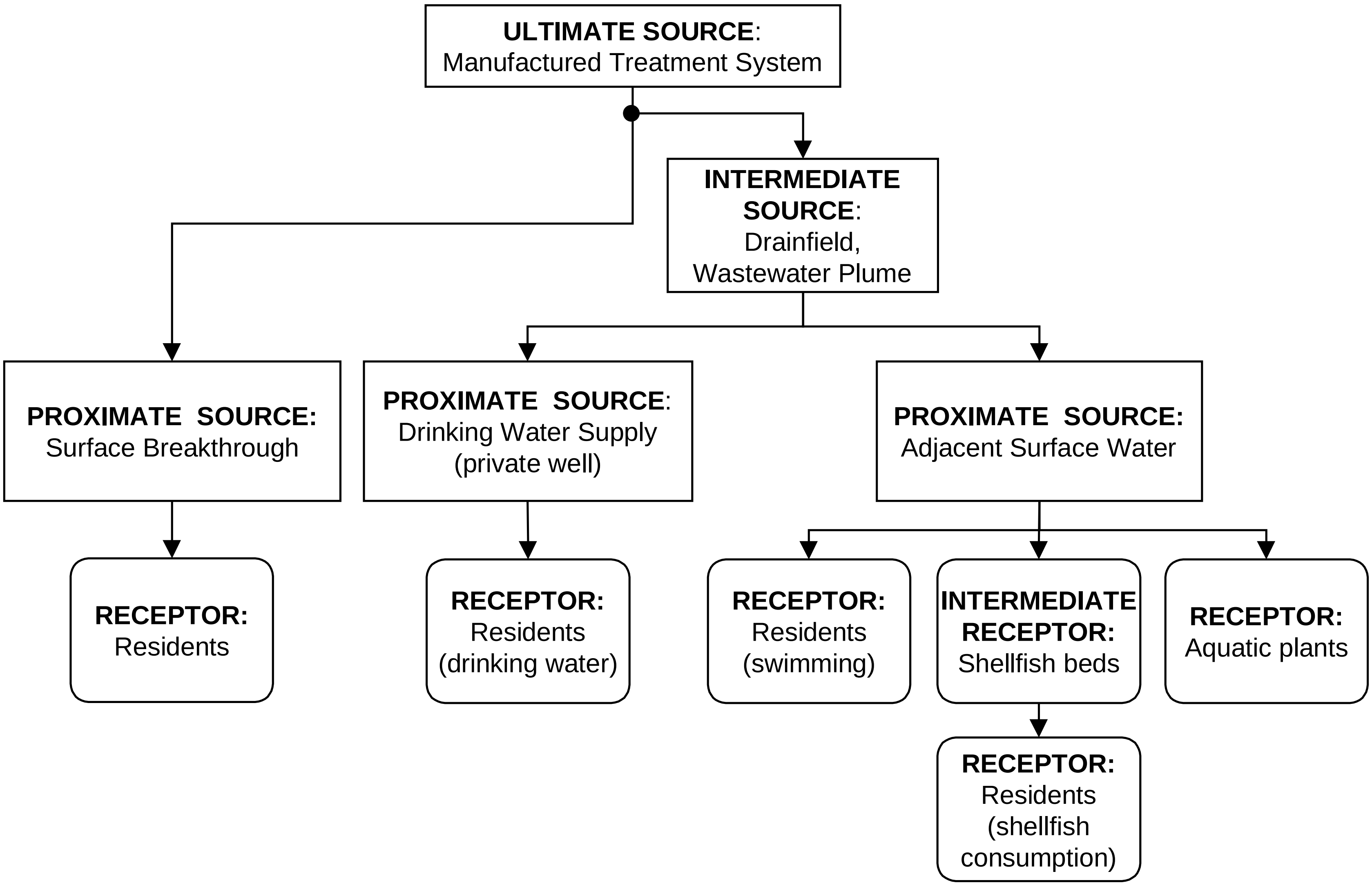

Figure 3. Example conceptual model for transport of nutrients and pathogens at the micro-level (individual treatment system)

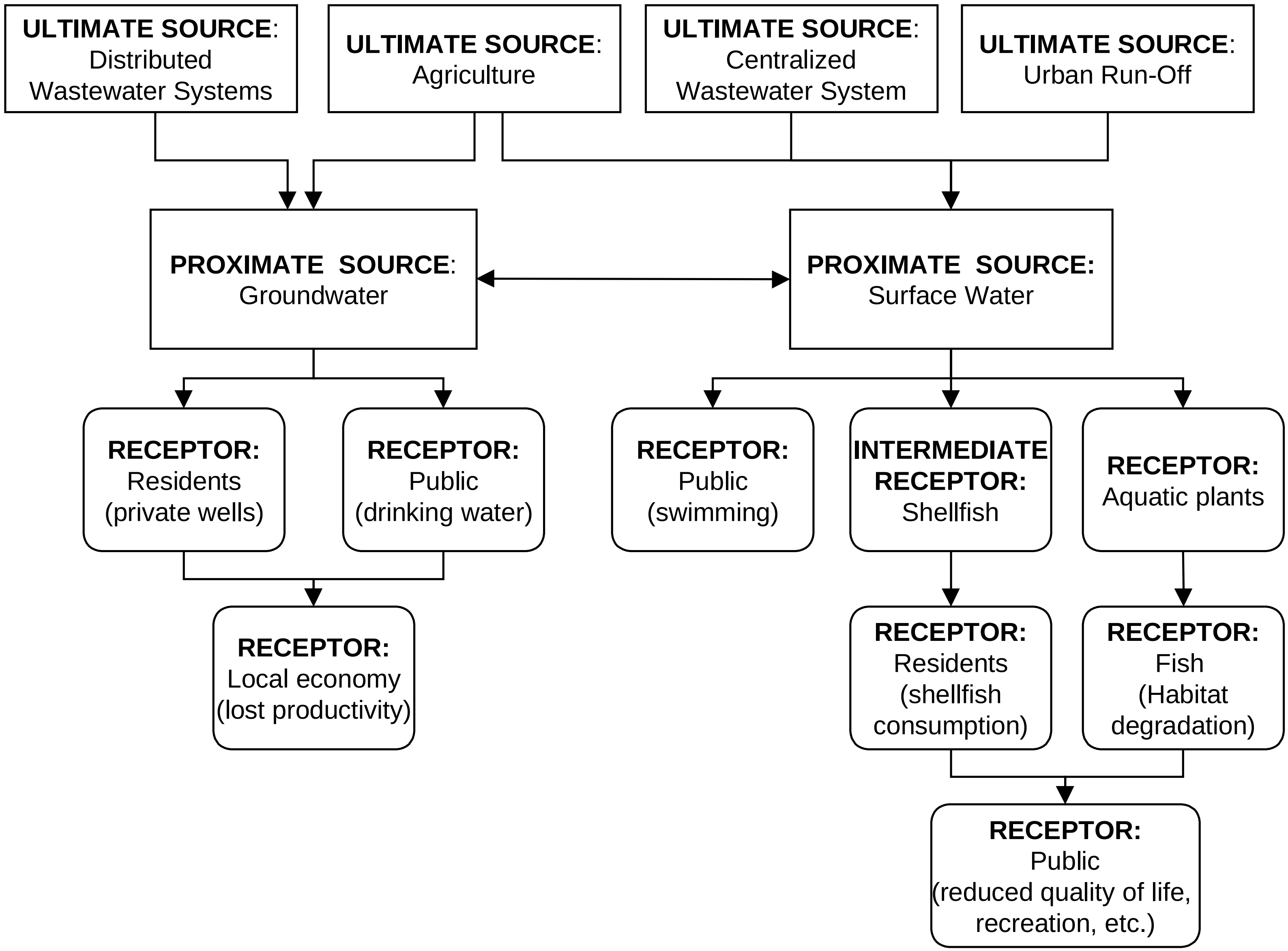

Figure 4. Example conceptual model for transport of nutrients and pathogens at the macro-level (watershed scale)

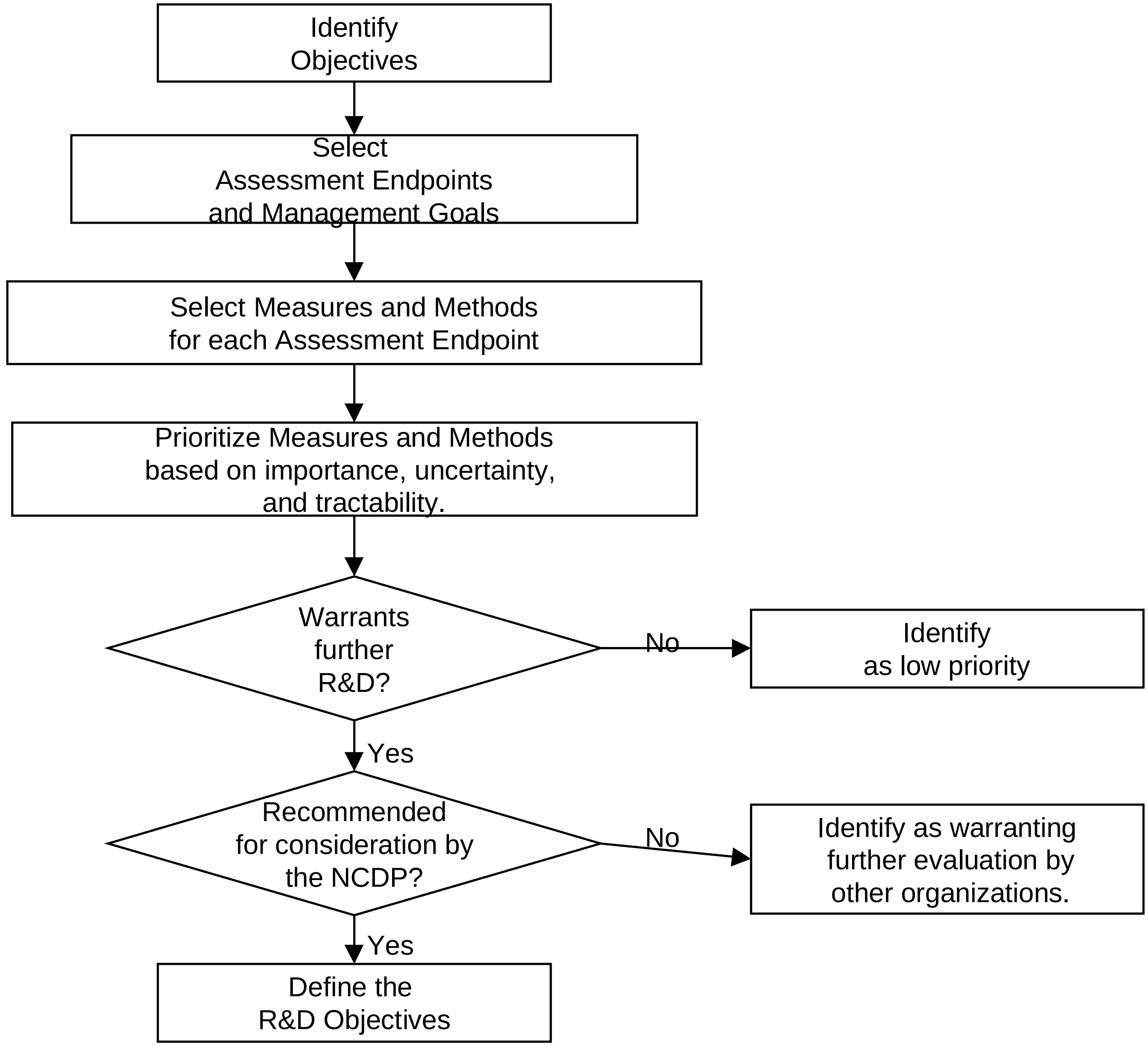

Figure 5
Risk-Based approach to management of the NCDP research and development efforts

Design and Performance of Onsite Wastewater Soil Absorption Systems

Prepared by

Robert L. Siegrist[1]
Colorado School of Mines
Golden, Colorado

E. Jerry Tyler[2]
University of Wisconsin
Madison, Wisconsin

Petter D. Jenssen[3]
Agricultural University of Norway
Aas, Norway

Prepared for

National Research Needs Conference
Risk-Based Decision Making for Onsite Wastewater Treatment
Washington University
St. Louis, Missouri
19-20 May 2000

Sponsored by

U.S. Environmental Protection Agency

Electric Power Research Institute's
Community Environmental Center

National Decentralized Water Resources
Capacity Development Project

Draft version, v2.0, May 2000
Final version, v3.5, September 2000

[1] Environmental Science & Engineering Division, Colorado School of Mines, Golden, Colorado. 90401-1887.
Tel. 303.273.3490, Fax. 303.273.3413, email: siegrist@mines.edu http://www.mines.edu/~rsiegris

[2] Soil Science Department, University of Wisconsin, Madison, Wisconsin. 53706.
Tel. 608.262.0853, Fax. 608.262. 265.2595, email: ejtyler@facstaff.wisc.edu

[3] Agricultural Engineering Department, Agricultural University of Norway, Aas, Norway.
Tel. 47.6494.8685, Fax. 47.6494.8810, email: petter.jenssen@itf.nlh.no

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Design and Performance of Onsite Wastewater Soil Absorption Systems

Robert L. Siegrist[1] E. Jerry Tyler[2] Petter D. Jenssen[3]

1. Abstract

The primary system for onsite and decentralized wastewater treatment in the U.S. includes septic tank pretreatment followed by subsurface infiltration and percolation through the vadose zone prior to recharge of the underlying ground water. These wastewater soil absorption systems (WSAS) have the potential to achieve high treatment efficiencies over a long service life at low cost, and be protective of public health and environmental quality. Favorable results from lab and field studies as well as an absence of documented adverse effects suggest that system design and performance are generally satisfactory. However, the understanding and predictability of performance as a function of design, installation/operation, and environmental factors, as well as the risk of inadequate function and its effects, have not been fully elucidated. This has been due to the complex and dynamic relationships between hydraulic and purification processes and the factors that control their behaviors. As a result, the current state-of-knowledge and standard-of-practice have gaps and shortcomings that can preclude rational system design to predictably and reliably achieve specific performance goals. Moreover, the quantitative analysis of long-term treatment efficacy on a site-scale up to watershed scale is difficult, as is any formal assessment of risks and selection of appropriate management actions. This white paper describes the process function and performance of WSAS. The system performance capabilities and predictability as well as reasonably conceivable system dysfunctions are described within a risk assessment and management framework. Issues applicable to the single-site scale and to the multiple-site to watershed scales are addressed. Based on an analysis of the current state-of-knowledge, critical research needs are identified and prioritized. As described herein, critical questions and current gaps in knowledge generally relate to the absence of fundamental process understanding that enables system performance relationships to be quantified and modeled for predictive purposes. High and very high priority research needs include those that support: (1) fundamental understanding of clogging zone genesis and unsaturated zone dynamics and their effects on treatment efficiency, particularly for pathogens, (2) development of modeling tools for predicting WSAS function and performance as affected by design and environmental conditions, (3) identification of indicators of performance and methods of cost-effective monitoring, and (4) development of valid accelerated testing methods for evaluating long-term WSAS performance.

2. Introduction

Wastewater infrastructure in the U.S. includes a continuum of technologies designed for scales of application that span from small decentralized systems serving individual homes in rural and suburban areas, to large centralized systems serving municipalities in densely populated urban areas. In the past, the decentralized or onsite systems were viewed by some as a means of providing temporary service until city sewers and a centralized treatment plant became available to provide permanent service. Early versions of onsite wastewater systems (e.g., pit privy, cesspool) were often designed with simple and short-term goals of waste disposal to prevent direct human contact and to achieve basic public health and environmental protection.

[1] Environmental Science & Engineering Division, Colorado School of Mines, Golden, Colorado. 90401-1887.
Tel. 303.273.3490, Fax. 303.273.3413, email: siegrist@mines.edu http://www.mines.edu/~rsiegris
[2] Soil Science Department, University of Wisconsin, Madison, Wisconsin. 53706.
Tel. 608.262.0853, Fax. 608.262. 265.2595, email: ejtyler@facstaff.wisc.edu
[3] Agricultural Engineering Department, Agricultural University of Norway, Aas, Norway.
Tel. 47.6494.8685, Fax. 47.6494.8810, email: petter.jenssen@itf.nlh.no

In the early 1900's, some system designs evolved to include raw wastewater pretreatment in a tank-based unit (e.g., septic tank) followed by disposal through a soil drainfield, and extension bulletins and guidance materials began to appear. As modern appliances became more commonplace, high water-use plumbing fixtures resulted in increased wastewater flows and a need for more careful siting and design of onsite wastewater soil absorption systems. For many designers and regulatory officials, the systems were still often viewed temporary with relatively simple waste disposal goals. During the 1990's the rapid movement toward centralization of wastewater treatment faded for a number of reasons, including the end of construction grants funding for treatment plants and a realization that large centralized solutions were not appropriate for all situations. Continuing to evolve, classic and alternative WSAS have been increasingly viewed as *treatment systems* and they have been designed and implemented to achieve purification as well as disposal, and even considered for beneficial reuse. Recently, increasing concerns over ground water quality and the effects of hazardous chemicals and waste pollutants have elevated the attention given to proper design and performance of WSAS. Today, nearly 25% of the U.S. population is served by onsite and decentralized wastewater systems and approximately one-third of new development is supported by such systems (USEPA, 1997). This amounts to roughly 25 million existing systems with 0.2 million new systems being installed each year. These onsite systems are now viewed as a necessary and permanent component of sustainable wastewater infrastructure in the U.S. and abroad.

The most common WSAS includes intermittent delivery (by gravity or pressurized dosing) of primary treated wastewater into the subsurface with infiltration and percolation through the vadose zone and into the underlying ground water (Fig. 1). Successful application of WSAS is based on engineering design that is compatible with the environmental conditions as determined through a site evaluation (Fig. 1). In properly implemented WSAS, advanced treatment is expected and can be achieved for many wastewater constituents of concern (COC's) through removal (e.g., filtration of suspended solids or sorption of phosphorus), transformation (e.g., nitrification of ammonium or biodegradation or organic matter), and destruction processes (e.g., die-off of bacteria or inactivation of virus) (Fig. 2). For the purposes of this discussion, the boundaries of the *WSAS treatment system* include the inlet to the soil absorption unit through the lower limit of the underlying vadose zone (see Figs. 1 and 2). In these WSAS, the conditions imposed by the WSAS process design (e.g., applied effluent quality and hydraulic loading rate) in a given environmental setting (e.g., soil type, moisture and temperature) must be such that key treatment processes occur at a rate and to an extent such that advanced treatment is reliably achieved before ground water recharge occurs (see Fig. 1). This is critical since the percolate released from most WSAS enters the underlying ground water, which can migrate under natural gradients toward points of exposure for receptors of concern (e.g., humans and drinking water supplies). Depending on local and regional conditions, ground water transport/fate processes may or may not reduce percolate COC concentrations, which would be of concern if exposure occurred at the point of percolate entry to the ground water, to lower levels that are not of concern at a remote point of exposure (Fig. 3).

In contrast to the modern WSAS simply illustrated in Figure 1, the large population of onsite systems in the U.S. today is extremely heterogeneous, including an array of old and new system designs, located in varied site conditions with different environmental sensitivities, and used to treat wastewaters from residential, commercial, and institutional sources (Table 1). Moreover, this population of systems includes those that are properly designed, installed and operated as well as those that are poorly designed, incorrectly sited, and/or improperly operated and maintained. Thus, characterization of performance capability and reliability for modern WSAS (e.g., Fig. 1) that are properly implemented in a given application must not be skewed based on the performance observed for older systems (e.g., disposal-based designs) and/or inappropriate applications (e.g., poorly sited systems).

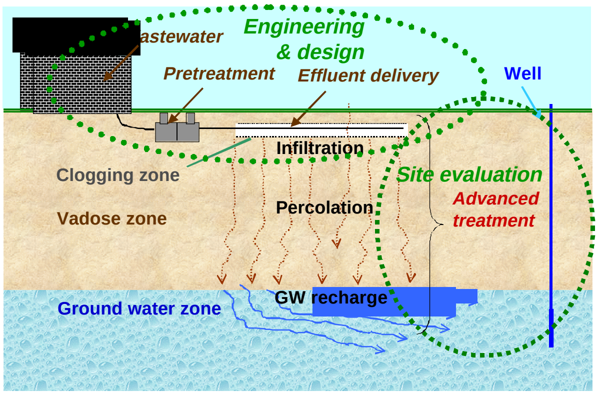


Fig. 1. Schematic of a modern wastewater soil absorption system and the engineering and design vs. site evaluation facets of system implementation.

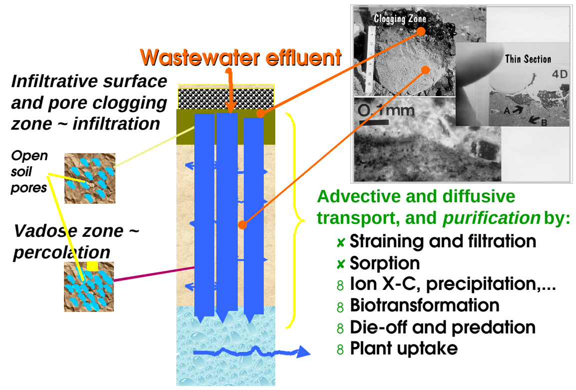


Fig. 2. Illustration of wastewater soil absorption system treatment processes.

For example, a very old system (e.g., 50-yr. old cesspool) might function effectively for hydraulics and disposal, yet accomplish limited purification, and thus its performance with respect to modern goals of treatment would be viewed as inadequate. This is in contrast to more modern systems designed to exploit physical, chemical, and biological processes to achieve highly efficient hydraulic and purification performance (e.g., 5-yr. old WSAS with pressure dosing of septic tank effluent into a network of shallow (e.g., 30 to 60 cm), narrow (e.g., 15 to 30 cm) trenches). In this paper, the emphasis will be on modern WSAS that have been designed, installed, and operated since about 1980 when contemporary understanding of onsite and decentralized systems was well documented and information was widely available (e.g., see USEPA, 1978; 1980).

Wastewater poses inherent risks due to its microbial and chemical constituents. The challenge with its management is to assess the magnitude of the risks in a given situation and decide on the most appropriate method to manage those risks (Fig. 3). For example, pathogenic bacteria, virus, and protozoa are present in wastewater, and disease could result if they are not removed or inactivated before an effluent reaches a receiving environment where humans can contact and ingest the water (e.g., drinking water, bathing beaches, shellfish beds). Also, if excessive levels of nitrogen and phosphorus in wastewater are input to sensitive surface waters (e.g., pristine lakes, estuaries), this could result in undesirable ecosystem changes (e.g., increased productivity and eutrophication). While simply stated, risk-based design and application of onsite WSAS is quite difficult to implement. For wastewater treatment, one could state the ultimate goal as being WSAS design and implementation so that (1) there is no infectious disease attributable to an onsite wastewater system, and (2) there is no measurable change in an ecosystem attributable to wastewater system inputs. Clearly, in a given setting, an onsite system that provides no treatment at all may present the highest risk, while increasing levels of reliable treatment effectiveness yield reduced levels of risk. However, since risk management requires consideration of nontechnical issues, such as socioeconomic factors, the most advanced treatment system will generally not be the best overall risk management solution.

Older systems that were designed and implemented to achieve disposal may represent an unacceptable risk to public health and environmental quality and need upgrading or replacement. A clear example of such a situation would include cesspools constructed in the ground water and with limited travel distances to drinking water supplies or sensitive surface waters. Other older systems are not so easily identified as inadequate and in need of upgrade or replacement. Modern onsite and decentralized systems are increasingly being designed and implemented as permanent and sustainable solutions for wastewater *treatment* rather than just disposal. In this context, *treatment* embodies goals associated with effective hydraulic and purification performance that can be sustained over a long service life at an affordable cost.

If these goals are sought and achieved, onsite systems can effectively manage public health and environmental quality risks that are inherent with microbial and chemical constituents normally found in domestic wastewater.

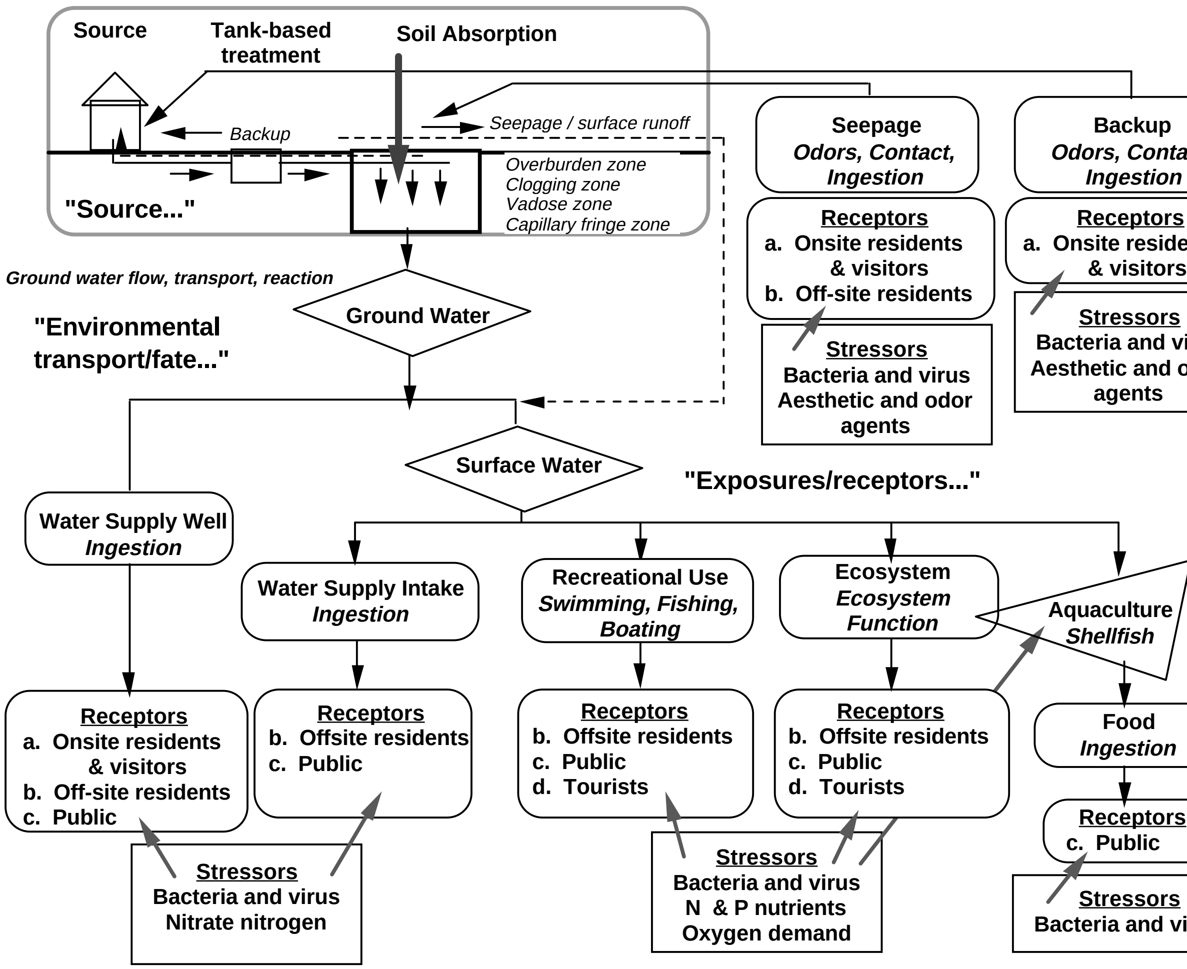


Fig. 3. Conceptual framework for risk assessment/management of wastewater soil absorption systems.

The design of wastewater systems for risk management necessarily requires the implicit or explicit setting of treatment goals. Only recently have attempts been made to explicitly establish performance goals or standards for WSAS (e.g., Otis and Anderson, 1994; Hoover et al., 1998b), in large part because this has been difficult for soil absorption systems. Explicitly establishing treatment goals and assessing their achievement requires that specific COC's are identified and the assessment methods are clearly defined. For a tank-based unit operation, it is straightforward to identify a common characteristic such as BOD_5 as a COC and to set the treatment goal as a certain average effluent concentration (e.g., 30 mg/L) and/or a % reduction in the influent concentration (e.g., 90%). The performance assessment could be made using 24-hr flow composited samples collected once each week with statistical analysis of the resulting dataset on a quarterly or annual basis. For WSAS, this is much more complicated based on both the variety of COC's that might be present (e.g., organics, nutrients, pathogens) in a given environmental setting and the absence of an "end-of-pipe" point of assessment. For example, one could assign the equivalent end-of-pipe assessment point to any one of the following: (1) soil solution in the lower limit of the vadose zone under the infiltration system under the seasonally highest water table elevation, (2) ground water at the downgradient edge of the footprint of the infiltration unit, (3) the ground water at the property boundary, or (4) the water at some proximal receptor that would be sensitive to potential wastewater system inputs (e.g., drinking water, shellfish waters). As of this writing, standardized performance goals have been advocated (e.g., Otis and Anderson, 1994; Hoover et al., 1998a), but there appears to be no consensus as to the COC's, the performance to be achieved within a prescribed space-time domain, or the methods to be used to measure and assess compliance. In this paper, the primary COC's are defined to include measures of oxygen consuming materials (e.g., BOD_5 and COD), nutrients (e.g., N and P), and human pathogens (e.g., bacteria and virus) based on their prevalence and potential adverse effects on human health and environmental quality. The primary treatment unit boundaries for a WSAS are defined to include the influent from a tank-based treatment unit (e.g., septic tank) through infiltration and percolation of the vadose zone and capillary fringe before discharge to a receiving ground water environment (Fig. 3). However, the method of performance assessment is not defined as it is application specific and many factors need to be considered such as system type, size, and the sensitivity of the primary receiving environments (e.g., ground water) and secondary recipients (e.g., surface waters).

3. Soil Absorption System Features and Design Basis

3.1. Features and Design of Modern WSAS

While old and new WSAS vary widely in their design and implementation (see Table 1), the vast majority of systems are based on discharge of partially treated wastewater effluent to subsurface soils with recharge to ground water underlying the site. The classic onsite system of modern design involves a wastewater source (e.g., dwelling unit), tank-based treatment unit (e.g., septic tank), and an infiltration unit (e.g., subsurface trench or bed) (Fig. 1). In this system type, water use from all fixtures and activities generates a combined raw sewage (solid plus liquid wastes) which flows into a septic tank buried outside but adjacent to the home or establishment. The principal treatment processes in a septic tank include sedimentation, flotation, and some anaerobic digestion. Septic tank effluent (STE) still contains high concentrations of organic matter, total suspended solids (SS), nutrients, and microorganisms and is not suitable for discharge to a receiving environment without further treatment (see Table 2). Requisite further treatment is achieved by discharging STE into a subsurface trench or bed filled with gravel aggregate or outfitted with a chamber, from which infiltration and percolation occur through an underlying unsaturated zone with recharge to ground water under the site (see Figs. 4 and 5). When a partially treated effluent such as STE, is applied to soil, infiltration and percolation through the unsaturated porous media involve a complex set of hydraulic and purification processes that can interact to reliably and sustainably achieve advanced treatment efficiencies (Table 2). These hydraulic and purification processes interact in a dynamic manner, evolving as a WSAS matures from startup through the first year(s) of operation.

Design of WSAS has historically been accomplished through a series of steps such as the following:

- Estimate the wastewater flow and composition with an implicit or explicit factor of safety,
- Characterize the site for landscape and land use features,
- Determine the subsurface lithology and hydraulic properties, and identify any limiting features,
- Select a design hydraulic loading rate, often based on a long-term acceptance rate for effluent,
- Specify geometry and placement of the infiltrative surface and its interface features,
- Select and size the pretreatment unit and the effluent delivery and distribution method,
- Determine what modifications, if any, are needed and appropriate for the site, and
- Select process controls and monitoring devices.

Table 1. Physical and operational features of historical, current, and emerging wastewater soil absorption system designs.[1]

Period of use	System type or operational feature	Motivation	Description of representative system features
1. Historical system designs	A. Cesspool	Disposal	Open or lined (e.g., brick or block) pit into which raw wastewater is discharged. Solids are retained in the pit while effluent infiltrates into the surrounding soil for disposal though some treatment can occur.
	B. Seepage pit	Disposal, some treatment	Open or lined (e.g., brick or block) pit into which pretreated wastewater is discharged. Effluent infiltrates into the surrounding soil for disposal though some treatment can occur.
	C. Leachfield	Disposal, more treatment	Network of trenches or beds filled with gravel or aggregate for disposal of pretreated wastewater by infiltration and percolation.
2. Current common system designs	A. Trench / bed WSAS	Disposal and treatment on favorable sites	Engineered network of trenches or beds filled with gravel or outfitted with chambers from which wastewater effluent (often from a septic tank [2]) infiltrates and percolates through 1 to 5 ft. or more of unsaturated soil before recharging ground water under the site.
	B. Shallow LPP WSAS	Disposal and treatment on difficult sites	Shallow, narrow trenches used for wastewater infiltration by intermittent delivery of wastewater effluent. Originally designed for sites with shallow, slowly permeable soils and seasonally high water table conditions.
	C. At-grade WSAS	Disposal and treatment on difficult sites	Trench or bed WSAS designed with the infiltration surface placed at the original ground level. Designed for sites with shallow depth to limiting conditions such as seasonally high water table or bedrock.
	D. Mound WSAS	Disposal and treatment on difficult sites	Trench or bed WSAS designed with the infiltration surface placed within a bed of imported sand fill above the original ground surface by 1 to 2 ft. Designed for sites with very shallow depth to limiting conditions such as seasonally high water table or bedrock.
2A. Current common installation or operational variants	A. Drainage	Increase vadose zone depth	Use of dewatering trenches or drains to lower the permanent or seasonal water table such that an adequate depth of unsaturated soil is maintained between the infiltrative surface.
	B. Over-excavation	Reduce particle sizes, increase media contact	Construction technique used wherein naturally occurring bedrock is excavated and crushed onsite and then placed back into the excavation. This creates a coarse grained fill into which a trench or bed WSAS can be installed.
	C. Dosing application	Cyclic loading, better distribution	Intermittent application of effluent to any WSAS with delivery in large draintile or small diameter pressure pipe.
	D. Pressurized dosing	Cyclic loading, uniform distribut.	Operational method of intermittent application of effluent into small diameter pressurized pipe to achieve more uniform distribution through the WSAS.
3. Emerging designs and operational variants	A. In-tank STE filters	SS removal	Filter cages installed into the effluent baffle from a treatment tank to capture suspended solids.
	B. Timed-pressure appl.	Cyclic loading, equalization	Design to include a pump vault and high/low switching gear with hourly bursts of STE discharged to a WSAS. Over a narrow range of liquid levels, the septic tank can provide some equalization capacity.
	C. Drip application	Treatment and reuse	Method of soil application where STE is further treated by optional methods before delivery to the shallow soil zone by timed pump application and drip emitter lines.
	D. Interm. sand filters	Adv. treatment	Design with single pass or recirculation through a 2 to 4 ft. packed bed of engineered sand media.
	E. Advanced treatment units (ATU's)	Adv. treatment	Tank based systems using biological treatment in suspended growth or packed bed systems, possibly incorporating biofilm supports of foam, textiles, or other materials.
	F. NO_3^- removal	Adv. treatment	Recirculation of STE through a packed bed and return to the influent end of the septic tank for nitrification-denitrification.
	G. UV irradiation	Disinfection	After advanced treatment, irradiation with UV light to kill/inactive pathogenic organisms in the effluent.

[1] The information is provided to represent typical characteristics for residential systems and it is recognized that all known or possible system designs or operational strategies are not included.
[2] In some locations, aerobic treatment units (e.g., extended aeration package plants) are conventionally used for pretreatment prior to wastewater application to soil.

Table 2. Wastewater COC's and representative concentrations in effluents applied to WSAS and percolates reaching ground water.

Constituents of concern (examples)	Example direct or indirect measures (Units) *Degree of explicit consideration in design or assessment*	Basis for concern over wastewater constituent	Relative degree of concern over treatment effectiveness of WSAS	Tank-based treatment unit effluent concentrations					WSAS percolate reaching ground water at 3 to 5 ft. depth (% reduction of effluent applied)
				Domestic septic tank effluent [1]	Domestic septic tank effluent with N-removal recycle [2]	Aerobic unit effluent	Sand filter effluent	Foam or textile filter effluent	
Oxygen demanding substances	BOD_5 (mg/L) *Common* [3]	(1) Create anoxic or anaerobic conditions and (2) stimulate clogging development	Low	140 to 200	80 to 120	5 to 50	2 to 15	5 to 15	>90%
Particulate solids	TSS (mg/L) *Common*	(1) Pore plugging and accelerated soil clogging	Low	50 to 100	50-80	5 to 100	5 to 20	5 to 10	>90%
Nitrogen	Total N (mg-N/L) *Common*	(1) Contributes to oxygen demand, (2) toxic via drinking water ingestion by sensitive receptors, (3) upset productivity in receiving waters.	High	40 to 100	10 to 30	25 to 60	10 to 50	30 to 60	10 to 20%
Phosphorus	Total P (mg-P/L) *Not common*	(1) causes increased productivity in surface waters.	Low	5 to 15	5 to 15	4 to 10	<1 to 10[4]	5 to 15[4]	100 to 0% [4]; highly variable due to soil's P sorption capacity
Bacteria (e.g., *Clostridium perfringens, Pseudomonas aeruginosa, Salmonella, Shigella*)	Fecal coli. (org./100 mL) *Common*	(1) infectious disease hazard to sensitive receptors by drinking water ingestion or contact with untreated seepage or via recreational water exposures.	Medium to high	10^6 to 10^8	10^6 to 10^8	10^3 to 10^4	10^1 to 10^3	10^1 to 10^3	>99.99%
Virus (e.g., enteric virus such as hepatitis, polio, echo, and coxsackie; coliphage)	Specific virus (pfu/mL) *Not common*	(1) infectious disease hazard to sensitive receptors by drinking water ingestion or contact with untreated seepage or via recreational water exposures.	High	0 to 10^5 (episodically present at high levels)	0 to 10^5 (episodically present at high levels)	0 to 10^5 (episodically present at high levels)	0 to 10^5 (episodically present at high levels)	0 to 10^5 (episodically present at high levels)	>99.9%
Organic chemicals (VOCs, endocrine disruptors)	Specific organics or total VOCs (ug/L) *Not common*	(1) potential carcinogens to humans by ingestion in drinking water or vapor inhalation during showering	Low at present	0 to trace levels (?)	0 to trace levels (?)	0 to trace levels (?)	0 to trace levels (?)	0 to trace levels (?)	>99%
Heavy metals (e.g., Pb, Cu, Ag, Hg)	Individual metals (ug/L) *Not common*	(1) potential toxicants to humans by ingestion in drinking water or (2) to ecosystem biota	Low at present	0 to trace levels	0 to trace levels	0 to trace levels	0 to trace levels	0 to trace levels	>99%

1 Note: concentrations given are for single family dwelling units. Multiple family units are probably quite similar. However, concentrations in restaurant STE are markedly higher particularly in BOD_5, COD and suspended solids (see Siegrist et al., 1985). Concentrations in graywater STE are noticeable lower in total nitrogen (see Siegrist and Boyle, 1982).
2 N-removal accomplished by recycling STE through a packed bed for nitrification with discharge into the influent end of the septic tank for denitrification.
3 None indicates characterization and monitoring not done and design basis limited with respect to these COC's.
4 P-removal by adsorption/precipitation is highly dependent on media sorption capacity and P loading rates and time of operation.

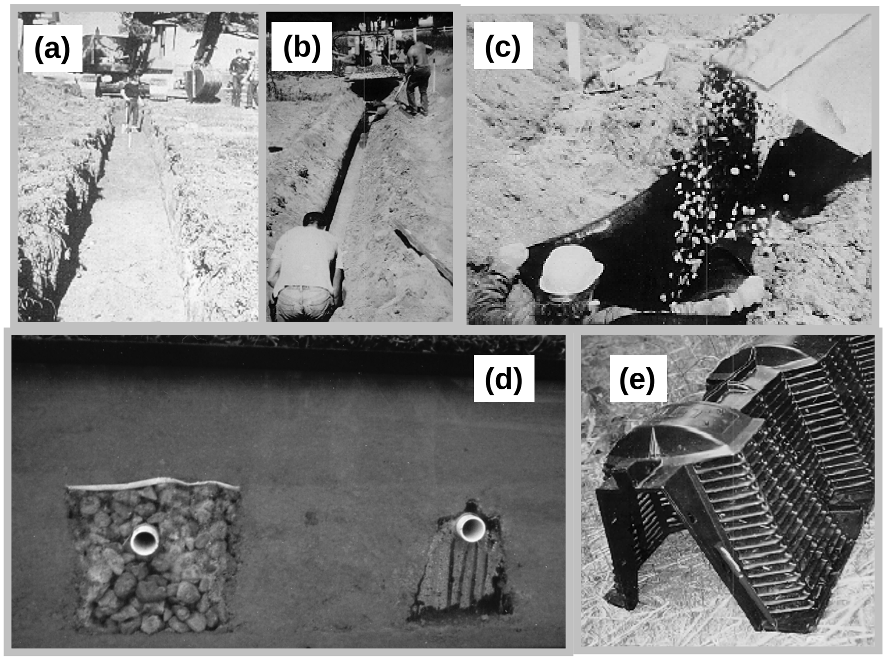


Fig. 4. Examples of soil absorption unit design approaches, including: (a) conventional trench excavation, (b) narrow trench excavation, (c) deep textile-lined narrow trench, (d) gravel-filled trench with geotextile overlay and 10-cm diameter STE delivery piping compared to a gravel-free chamber unit, (e) gravel-free chamber unit.

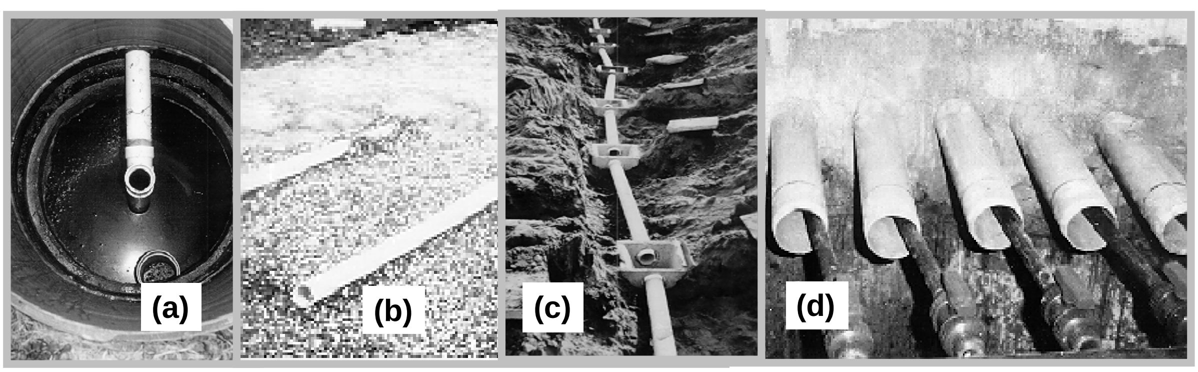


Fig. 5. Examples of effluent delivery methods for delivery of wastewater effluent into the soil absorption unit: (a) STE baffle and gravity outlet, (b) 10-cm diameter perforated drain tile, (c) drop-box serial distribution for sloping sites, and (d) hydrosplitter to equalize flow between trenches.

The design of WSAS has normally been completed with an over-riding conservatism in most all steps of the process, for example during selection of the (1) design flow, (2) septic tank size, and (3) application rate to the soil infiltrative surface (IS). The experiences and preferences of local designers and contractors, as well as the availability of materials and equipment, that lead to the lowest system costs often determine the WSAS designs that are most commonly used. For single-family home and other small WSAS, the design practices are often prescribed in state or local codes (e.g., Docken and Burkes, 1994; Briggs and Barranco, 1994) which can vary widely from state to state and even county to county within a given state. The codes themselves have evolved from local practices and perceptions, sometimes accounting for local conditions (e.g., topography, climate), but often not based on any fundamental understanding or even objective technical data. Recognizing the great range in actual past and current practices, the following remarks are made to illustrate the type of practices used for a classic WSAS serving a single-family dwelling unit.

Design flows are commonly calculated for a single-family home based on a per capita flow and a residency estimate (USEPA, 1980a; Crites and Tchobanoglous, 1998). Flow estimates for commercial and institutional sources are based on occupancy and/or event/activity water-use and are much more difficult to estimate accurately due to highly uncertain practices (Siegrist et al., 1985a). For a single-family dwelling unit with four bedrooms, the design flow might be estimated at 2.27 m^3/d (600 gallons per day, gpd) (assuming all four bedrooms are occupied by two persons each of whom produces 280 L per day (Lpd) of wastewater (75 gal per capita per day, gpcd). This represents an implicit factor of safety of over 500% compared to an average occupancy of 2.1 persons per home and average per capita flow of 170 Lpd (45 gpcd) (USEPA, 1978; 1980). This large factor of safety may be appropriate for one single dwelling unit to ensure that the actual flow generated at any individual dwelling within a large population of dwellings does not exceed the design flow. However, for a cluster of dwellings (e.g., five or more homes on a clustered system), the design flow can be reduced toward the average since the clustering attenuates the actual flow variations from home to home. For clusters of 5 or more dwelling units, the daily design flow can be based on average conditions with an explicit factor of safety (e.g., 1.5 to 2.0) applied to the base flow.

Septic tanks are normally sized based on the design flow and 2/3 of the tank volume set aside for sludge and scum accumulation and a 24-hr hydraulic detention time in the remaining 1/3 volume. This effectively yields a total tank volume equal to 3 times the daily flow volume (Baumann et al., 1978; USEPA, 1980a). The septic tank sizing and design features can affect the average STE output rate and quality as well as the raw wastewater source (see Tables 1 and 2). Baffles are provided on the inlet and outlet of a septic tank (see Fig. 5) to yield quiescent conditions within the tank and limit the disruption and re-entrainment of sludge and scum in the wastewater passing through the tank, thereby minimizing suspended solids concentrations in the STE. Sludge and scum accumulate over time in a septic tank and these solids must periodically be pumped out and properly managed (USEPA, 1995). The needed frequency of pumping and composition of the removed solids, referred to as septage, has been related to the type of usage (e.g., with garbage disposals) and environmental conditions (e.g., temperature)(USEPA, 1980a,b; 1995; Bounds, 1995a,b).

The soil absorption unit size is determined by selecting (1) a specific infiltrative surface geometry (e.g., sidewall vs. bottom area) and placement in the soil profile (e.g., *in situ* deep, *in situ* shallow, at grade, or mounded in fill), (2) infiltrative surface character (e.g., gravel-laden or gravel-free chamber units, and (3) estimating the steady-state hydraulic capacity (e.g., cm/d) of the IS once a system is fully mature and soil clogging has approached its maximum (see Fig. 5). While bed geometries permit more efficient use of landscape area, with increasing IS area per unit length of system, beds can experience diminished performance due to construction damage, high overburden pressures and gravel embedment, gas entrapment and anaerobiosis due to inhibited O_2 transfer, and potentially excessive ground water mounding and reduced unsaturated zone depth (Siegrist et al., 1984; 1986; Mahuta and Boyle, 1991). To mitigate the negative effects of beds, trench geometry's with shallow placement have been advocated to maximize IS area and exploit the most biogeochemically active zone of the soil profile. The required gross area of IS is based on the design flow divided by a long-term acceptance rate (LTAR) for the IS expressed in volume per area per time (e.g., 1 $cm^3/cm^2/d$ = 1 cm/d = 0.245 gpd/ft^2). This gross IS area may be increased by a factor of 1.5 to allow for extended resting (e.g., 6 mon.) of 1/3 of the absorption system to retard soil clogging development. The gross IS required is then converted to a length of trench of a prescribed width (e.g., 90 cm or 3 ft.) which then must be laid out on the landscape. Trench separation is prescribed (e.g., 1.8 m or 6 ft.) to enable a platform for construction equipment during installation. Modern installation methods can use specialized equipment (e.g., continuous trenchers) for which trench separation can be quite low based on equipment constraints alone.

For most systems, delivery and application of the STE to the absorption area is based on wastewater generation in the dwelling unit or establishment with gravity flow from the septic tank designed to be distributed to all of the operational absorption trenches or bed units. Attempts to distribute the flow equally between trenches or areas of a bed using distribution boxes and 10-cm (4-in) diameter perforated drain pipe are commonly made, but have been shown to be ineffective (Otis et al., 1978) (see Fig. 5). As described below, this has led to modifications in system design that incorporate dosing into larger gravity piping or dosing into small pressurized piping for more uniform delivery. For some systems, serial distribution is used whereby a portion of a system is hydraulically overloaded during system startup, but as clogging evolves, additional trenches are loaded based on overflow from an upslope trench (Otis et al., 1978).

The total infiltration area required in a WSAS is determined explicitly or implicitly based on a long-term acceptance rate concept that attempts to account for the loss in infiltration rate capacity that occurs in soils as a result of wastewater effluent infiltration (more discussion is given in Section 4). For most situations with individual onsite systems, the effective IS area (i.e., bottom area vs. sidewall vs. both) and the LTAR are incorporated but hidden within a code-prescribed system. For example, a prescribed sizing for a 4-bedroom home on sandy soil might be to provide 60-m of lineal 90-cm wide by 30-cm deep trench. As discussed in Section 4, several attempts have been made to estimate system infiltrative area requirements by selecting an LTAR based on correlation's between a LTAR and soil physical properties (e.g., Ryon, 1928; USPHS, 1967; Jones and Taylor, 1964; Bouma, 1975). Kiker (1948) proposed a fixed reduction factor based on the clean water infiltration rate. Ryon (1928) and later the U.S. Public Health Service (USPHS) (1967) based the assessment on a crude percolation test and a simple empirical relationship. Both of these methods are based on a strong soil dependence of the hydraulic design rate. Based on the imprecision and error of the test and a lack of any correlation between the test results and an LTAR (Bouma, 1971; Healy and Laak, 1974b; Jenssen, 1986; 1988), soil morphology evaluation was promoted as a better method to estimate infiltrative capacity as well as identify depths to limiting conditions in the soil profile (e.g., seasonal perched ground water, low permeability restrictive layers) (e.g., Tyler and Converse, 1994). However, the morphologic description may be best suited to eliminating applications to problem sites and thereby preventing failures as opposed to discriminating a LTAR based on subtleties in soil morphology. Research does suggest lesser dependence of a LTAR on soil properties such as soil texture and greater dependence on wastewater application rate and composition (Jenssen, 1986).

Common practice continues to be that the design application rates for soil absorption systems (trenches or beds) are typically in the range of 1 to 5 cm/d (0.24 to 1.23 gpd/ft^2) (either explicitly or implicitly set) with the site-specific rate based on soil textural properties (e.g., 5 cm/d for a sand and 1 cm/d for a clay loam) and in some areas, percolation testing (e.g., 5 cm/d (1.23 gpd/ft^2) for a 10 minutes per inch (MPI) percolation rate and 1 cm/d (0.24 gpd/ft^2) for a 60 MPI rate). While these relatively low design rates, which are only minute fractions of the respective soil saturated hydraulic conductivities (Ksat), are speculated to represent an LTAR, there are continuing debates regarding the nature and magnitude of LTAR's. Some investigators have reported that an equilibrium or steady-state LTAR actually evolves (Healy and Laak, 1974a; Kropf et al., 1977; Anderson et al., 1982) while others have reported that a continuous, albeit slow, decrease in infiltration rate capacity occurs (Thomas et al., 1966; Okubu and Matsumoto, 1979; Jenssen, 1986). It is likely that an LTAR does not represent a steady-state infiltration rate capacity at which a wastewater absorption system will operate indefinitely when continuously used and in the absence of permeability restoring processes (e.g., soil biota penetration, freeze-thaw effects). Rather, most systems that are operated under continuous use with STE applied at a design rate of 1 to 5 cm/d (0.24 to 1.23 gpd/ft^2) will eventually clog to a degree where hydraulic failure can occur (i.e., the daily application rate exceeds the infiltration rate at time, t (IR_t). The wastewater-induced soil clogging development and hence IR_t is dependent on several factors such as soil morphology (Jones and Taylor, 1964; Healy and Laak, 1974a; Bouma, 1975; Jenssen, 1986), wastewater composition and loading rate (Laak, 1970; Siegrist, 1987a; Duncan et al., 1994; Loudon et al., 1998; Amoozegar and Niewoehmner, 1998; Loudon and Mokma, 1999) and application mode and continuity of use (McGauhey and Krone, 1967; Siegrist, 1987a; Hargett et al., 1982; Tyler et al., 1985). Hence, the clogging process is complex and difficult to model precisely. Most criteria for sizing of soil infiltration systems are therefore still based on empirical data regarding LTAR's (Ryon, 1928; USPHS, 1967; Anderson et al., 1982) with increases in area provided based on implicit or explicit factors of safety added (e.g., conservatively estimated design flows or increased areas for beds over trenches, respectively).

It is emphasized that practices as described above are applicable to domestic wastewater, often from single-family homes. During the 1970's and 1980's, applications began to occur that included different wastewater types and scales of development, such as multiple family dwelling units, restaurants and commercial facilities, and small communities. The WSAS designs for these facilities was initially based on a simple scale-up from that used for single-family homes with little or no adjustment for the performance effects of system size and/or wastewater composition. As a result, hydraulic and purification dysfunctions were reported (e.g., Siegrist et al., 1985a,b; Siegrist et al., 1986; Plews and DeWalle, 1985) which led to modifications in design practice to account for the performance effects of wastewater source type and landscape loading. On the contrary, more dilute wastewaters such as graywater STE, may permit different treatment approaches and equivalent or better public health protection (Siegrist and Boyle, 1982).

3.2. Modifications to Classic System Designs

There are a number of modifications to the classic WSAS as described above that have evolved to improve its performance capabilities and/or reliability (see Tables 1 and 2). Modifications of the wastewater source can be made to reduce the volume of wastewater to be treated and/or its pollutant load through (1) flow reduction (e.g., water conserving fixtures) (Siegrist et al., 1978), (2) waste segregation (e.g., no garbage disposal, urine separation, or graywater vs. black water separation) (Siegrist, 1978; Siegrist and Boyle, 1982: Jenssen and Skjelhaugen, 1994; Rasmussen et al., 1996), (3) in-house recycle (e.g., graywater for toilet flushing) (Anderson et al., 1981; Siegrist et al., 1981), and/or (4) point of use treatment (e.g., bag filter on laundry discharges) (Fig. 6). Modifications to septic tank designs have been targeted at STE quality, particularly with respect to reducing the STE suspended solids concentration and to a lesser extent the BOD_5, and thereby prevent accelerated clogging of soil absorption systems. Examples of these include the use of septic tank effluent biofilters units (see Fig. 7a).

Fig. 6. Examples of wastewater source modifications: (a) 3-L volume flush toilet, (b) compost toilet, (c) graywater recycle unit, and (d) point-of-use bag filter for laundry discharge.

Modifications or variants to the soil infiltrative surface have been directed at improving infiltration capacity through changes to the interface character and its geometry and placement in the soil profile. Research related to the rate and extent of soil clogging in WSAS with gravel aggregate on the infiltrative surface (aggregate-laden) led to the development and use of infiltration systems which have an open surface without a layer of aggregate on it (aggregate-free), the most common of which is a chamber system (see Fig. 4) (Tyler et al., 1991; Keys, 1996; May, 1996; Loudon et al., 1998; Van Cuyk et al., 2000). Based on the potential adverse effects of gravel on short- and long-term infiltration capacity (e.g., compaction, fines, embedment, and focused pollutant loading), these aggregate-free systems are designed with infiltration areas on the order of 40% to 50% less than required with gravel systems (see Fig. 4). Geometry and placement in the soil profile can be selected to maximize infiltrative surface area and enable delivery of effluent into the soil where the treatment potential is highest. Increasingly, the use of narrow trenches (e.g., 15- to 30-cm wide) that are placed shallow in the soil profile (e.g., 30- to 60-cm depth) is being promoted (see Fig. 5). Soil permeability is usually higher shallow in the profile and more importantly, narrow and shallow placement improves aeration potential. The use of at-grade and low pressure pipe (LPP) systems were designed to place the infiltration surface very near the land surface while mound systems place it in an imported layer of sand fill. These system types are intended to overcome site limitations associated with an inadequate unsaturated zone thickness beneath an IS (Converse et al., 1978; Tyler and Converse, 1985; Stewart and Reneau, 1988; Converse et al., 1991; Hoover and Amoozegar, 1989; Amoozegar et al., 1994). Such limitations are most often due to the presence of a low permeability layer, seasonal or permanent high water table, and/or porous or fractured bedrock.

Modifications to the classic WSAS also encompass the method of delivery and frequency of application of wastewater effluent to the soil (see Fig. 5) (Otis et al., 1978). With the addition of a pump or siphon to the system, intermittent dosing into conventional 10-cm (4-in.) diameter perforated pipe can enhance the delivery of STE to a soil absorption system (see Fig. 5). Compared to the normal, gravity delivery that results in a semicontinuous trickle flow that is randomly and non-uniformly distributed, dosing improves intermittent delivery of STE and improves the distribution somewhat. If small diameter (e.g., 2.5-cm) perforated (e.g., 3.2-mm orifices) pipe is used, a pump or siphon can produce pressurized distribution of the dosed effluent which can lead to more uniform application of the loading to the soil system. Early research led to guidance that dosing frequencies should be 1 to 4 times per day based on waste generation characteristics and pressurized dosing networks should be designed to achieve relatively equal headlosses and flow rates between orifices (Otis et al., 1978). Later, Hargett et al. (1982) showed that pressurized dosing offered little advantage over gravity fed application in a silt loam soil since, with both delivery methods, soil clogging evolved to the extent that the infiltrative surface was continuously ponded and fully utilized in both loading regimes. A recent innovation includes the concept of timed dosing through pressurized distribution networks where the septic tank provides equalization capacity to permit frequent dosing. This is thought to enable more uniform application and enable more unsaturated flow through the unsaturated zone beneath the infiltrative surface, thereby aiding treatment.

An important development which might be viewed as a modification to the classic WSAS, includes the array of devices and equipment that have evolved to enable process control and monitoring of system function and performance. For example, the addition of control panels with hydromechanical sensors and telemetry features have provided a means by which to control effluent application to a soil absorption unit, to record effluent loading rates, and to detect gross system dysfunction and correct it early. Control and monitoring of purification still relies on sampling and analysis, which is easy for end-of-pipe locations but difficult for soil solution and ground water.

3.3. Alternative Unit Operations to Classic System Designs

Alternatives to the classic system design involve major changes in the unit operations and treatment train within the system (Fig. 7) (see Tables 1 and 2). These alternatives can be categorized to include (A) add-on treatment units to a septic tank, (B) anaerobic unit operations as replacements for a septic tank, (C) aerobic package plants, and (D) engineered porous media biofilters (PMB's). The primary purpose of these alternatives has been to (1) provide a measurable improvement in BOD_5 and SS removal (group A); (2) remove nitrogen from STE before discharge to a WSAS thereby reducing nitrate contamination of ground water (group B), (3) markedly reduce the BOD_5 and SS concentrations in STE before discharge to a WSAS thereby retarding soil clogging (group C and D) and/or (4) produce an effluent suitable for disinfection and discharge to the land surface (disposal only or beneficial reuse such as landscape irrigation) or a receiving water (group C and D).

Add-on units (AOU's) are relatively simple in design and operation and include (1) specially designed effluent filters which support biomass growth and SS removal (Fig. 7a) and (2) submerged media filters with aeration and recirculation provided by a simple air-lift pump to provide some BOD_5 removal (Fig. 7b). The treatment efficiency of these AOU's remains somewhat speculative as there have been few if any experimental studies documenting performance.

Anaerobic upflow filters were envisioned as means of equal or better treatment with less susceptibility to upset and high concentrations of SS being released into the STE. These systems were comprised of rock-filled tanks with an upflow flow regime to aid in distribution through the media. Performance observations suggest the filter's performance is comparable to that of a well-designed septic tank, and possibly improved in some cases (Kennedy, 1982).

Aerobic package plants based on fixed film or suspended growth processes were down-scaled from traditional designs in an effort to produce an effluent quality suitable for infiltration in low permeability soils and/or for discharge with disinfection to the ground surface or a receiving water (see Fig. 7c). While these systems were shown to have the inherent ability to produce a higher quality effluent than STE, they were subject to mechanical malfunctions and process upsets (Hutzler et al., 1978; USEPA, 1978; 1980; NSF, 1996). Thus, to reliably achieve their system performance capabilities, operation and maintenance (O&M) must be provided.

Advanced treatment has been demonstrated with PMB's comprised of a bed of sand (Fig. 7d), peat, foam (Fig. 7e), textiles, or other granular media (see Fig. 7) that are intermittently loaded (e.g., 4 to 24 times per day) at hydraulic loading rates that are much higher than those for a soil WSAS (e.g., 5 to 20 gpd/ft^2 vs. $\leq$1 gpd/ft^2) (Anderson et al., 1985; Effort et al., 1985; Jowett and McMaster, 1994; Loomis and Dow, 1998; Crites and Tchobanoglous, 1998; Driscoll et al., 1998; Roy et al., 1998; Van Cuyk et al., 2000). The higher loading rates are enabled by coarse particle diameters and design, which allows easy access to the medium to clean and/or replace it if, needed. These PMB's are being advocated and used to provide higher quality effluents thereby reducing the purification that the soil absorption system must achieve as well as reducing soil clogging and enabling higher application rates. Most PMB systems can yield substantial reductions in BOD_5 and SS as well as complete nitrification and even some N-removal (Lamb et al., 1990; Loomis and Dow, 1998). Microbes can be reduced by a factor of 10 to more than 1000, but there still can be pathogenic bacteria, viruses and protozoa in the effluent (Emerick et al, 1997; Higgins et al., 1999; Loomis and Dow, 1998; Van Cuyk et al., 2000). While small-scale onsite disinfection units (e.g., chlorination, ultraviolet light irradiation) are available, they are rarely used prior to subsurface soil absorption.

PMB's have also been applied to achieve nitrogen removal in an otherwise classic WSAS. The nitrified effluent from a PMB (e.g., trickling filter, textile filter, or sand filter) is directed back to the influent end of the septic tank (Whitmayer et al., 1991). The anaerobic conditions in the tank combined with adequate carbon and nitrate as an electron acceptor have been shown to enable nitrogen removal, on the order of 50 to 80% or higher in some cases, thereby yielding STE concentrations of 20 mg-N/L or less (Shafer, 2000).

Beneficial reuse of wastewater effluent has been accomplished through graywater treatment systems producing effluent for flushing water carriage toilets and/or landscape irrigation (Anderson et al., 1981). A recent innovation involves the application of drip irrigation tubing and emitters to deliver STE to the shallow subsurface into the root zone (see Fig. 7f) (Sinclair, et al., 1999). To prevent emitter plugging, a spin-disk filter apparatus is used to remove suspended solids normally found in STE.

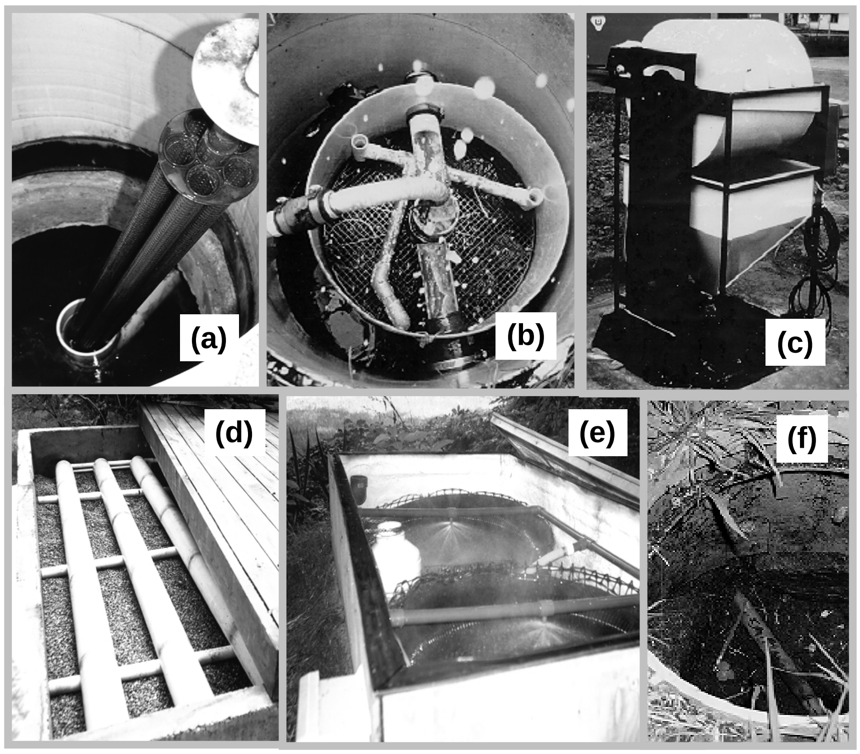


Fig. 7. Examples of alternative unit operations including: (a) effluent biofilter unit, (b) in-tank aeration unit, (c) rotating biological contactor, (d) sand filter, (e) foam filter, and (f) drip irrigation line.

4. Performance Capabilities, Predictability, and Reliability

4.1. General Performance Capabilities of WSAS

The performance achieved by a modern WSAS depends on a number of inter-related factors. Engineering design is completed for a given application based on a site evaluation. This leads to construction and startup, followed by system usage, and any requisite O&M. If all of these factors are properly addressed, and the actual conditions and usage are consistent with any assumptions made, then system performance should be as described below. However, if any of these factors are overlooked or inadequately addressed, or if actual conditions depart from assumptions made in design and implementation, then performance deficiencies can occur either early or late in the system's life. These deficiencies can manifest themselves as mechanical, hydraulic, and/or purification dysfunctions and all three can increase the risks of adverse public health and environmental effects (see Fig. 3). Purification dysfunctions that lead to ground water and surface water contamination and are of particular concern since they can be difficult to monitor for, detect, and mitigate.

The design of any WSAS inherently includes subsurface infiltration and percolation for advanced treatment and disposal of a partially treated effluent, most often STE. As noted earlier, these systems typically employ delivery of primary treated wastewater into a soil absorption trench or bed from which wastewater infiltrates and percolates through a depth of soil into underlying ground water (see Figs. 1 and 2) (Anderson et al., 1985; Brown et al., 1979; USEPA, 1978, 1980, 1981, 1992, 1997; Kristiansen, 1982, 1991; Jenssen and Siegrist, 1990; Crites and Tchobanoglous, 1998). Effective purification requires adequate hydraulic retention time (HRT) and suitable conditions for treatment processes to function (e.g., adequate biomass and bioactivity, aerobic conditions, favorable pH and temperature) such that processes occur at a rate and extent to achieve removal (e.g., sorption/precipitation), transformation (e.g., biodegradation) and die-off/inactivation before ground water recharge occurs (see Fig. 2). The percolate moving downward from the WSAS may be mixed with ambient ground water, which migrates under, natural gradients toward points of exposure to receptors of concern (see Fig. 3). Depending on local and regional conditions, transport/fate processes along the pathways to receptors may or may not reduce residual concentrations of COC's in the percolate from a WSAS that are above threshold concentrations to lower levels that are no longer of concern.

For effective purification of primary treated wastewater in natural soils, unsaturated flow in the porous medium can be critical since this controls contact between wastewater constituents and soil particles and associated biofilms, over an adequate period for treatment processes to occur (Bouma, 1975; USEPA 1978; Jenssen and Siegrist, 1990; Emerick et al., 1997; Schwager and Boller, 1997; Stevik et al., 1999; Van Cuyk et al., 2000; McCray et al., 2000). Unsaturated flow conditions can be achieved by application of limited daily loadings (e.g., 1 to 5 cm/d) which are usually a minute fraction of the medium's K_{sat} (e.g., 100 to 1000 cm/d). Intermittent dosing (e.g., 4 to 24 times per day) and pressurized uniform application can also be employed to help create an unsaturated flow regime. Also, in time, wastewater-induced soil clogging evolves due to an accumulation of inert particles and amorphous organic matter (like humic-substances) in a few cm-thick zone at the IS (see Fig. 2) (Otis, 1985; Jenssen and Krogstad, 1988; Siegrist, 1987a; Siegrist et al., 1991). This clogging leads to a reduced permeability and more uniform temporal and spatial infiltration with a concomitant unsaturated flow almost independent of wastewater loading. When soil-clogging is extensive, STE may continually pond on the horizontal infiltrative surface thereby causing vertical sidewalls to become available for infiltration. Soil clogging is an important, if not critical, process, which contributes to the advanced treatment potential of WSAS. Not only does it enhance infiltration surface utilization and yield an unsaturated flow regime in the vadose zone, it provides powerful treatment in the clogging zone. However, if soil clogging yields too great a reduction in permeability at the IS, it can be detrimental by causing hydraulic dysfunction (e.g. backup into a dwelling or seepage to the ground surface) or adversely affecting purification (e.g., anaerobic conditions and reduced biotransformation rates).

For the common wastewater chemical COC's such as BOD_5, COD, and SS, purification efficiencies of >90% can be sustainably achieved by filtration, sorption, and biodegradation processes in most WSAS and settings (see Fig. 2; Table 2) (USEPA, 1978; 1980; Jenssen and Siegrist, 1990; Van Cuyk et al., 2000). With dilution and dispersion in the ground water and any additional removal therein, these COC's seldom present any concern for adverse impacts to the receiving environment. However, nutrient removal (nitrogen and phosphorus) and any adverse impact on a receiving environment are much more sensitive to process design and site conditions (Gold and Sims, 2000). Microbial COC's commonly found in STE include pathogenic bacteria at sustained, high concentrations and virus and protozoa at highly variable and episodically released levels (see Table 2) (Bicki et al., 1984; Anderson et al., 1985; USEPA, 1978; Van Cuyk et al., 2000; Cliver, 2000). While WSAS performance has been documented for bacteria such as fecal coliforms, there is less information on purification with respect to pathogenic bacteria, viruses, and protozoa. Purification efficiencies in WSAS can be very high, yielding near complete removal of fecal coliform bacteria and 99.99% or higher reductions in virus (Emerick et al., 1997; Stevik et al., 1999; Van Cuyk et al., 2000). Despite excellent purification performance observed in controlled experiments (e.g., Van Cuyk et al., 2000) and field studies with properly implemented modern systems (e.g., Anderson et al., 1991, Higgins et al., 1999, Oakley et al., 1999), the transport of pathogens from WSAS to ground water, and in some cases, drinking water has been alleged (e.g., Rose et al., 1999). However, the factors causing the transport were often not documented, or the WSAS studied were of older disposal-based designs.

Apart from purification efficiency, the hydraulic function of a WSAS is often gauged by its service life. Service life is closely related to soil clogging and the daily loading rate vs. the long-term acceptance rate, which in turn are influenced by an operational loading factor (LF = ratio of actual loading to design loading) and continuity factor (CF = days of use divided by 365). At low LF's or for low CF's, service lives may be practically indefinite. Several studies of system service life have been completed during the past 25 yr. suggesting hydraulic service lives varying from 11 to >30 yr. (see Fig. 8) (Hill and Frink, 1980; Hoxie and Frick, 1984; Plews and de Walle, 1985; Gårderlokken, 1997; Sherman et al., 1998; Keys et al., 1998). Hill and Frink (1980) studied more than 3000 small systems and concluded that a service life of more than 30 years could be expected. Plews and de Walle (1985) studied 369 large, buried systems and found that more than 60% had a hydraulic service life of more than 20 years. For systems with an actual loading rate of < 4 cm/d only 3.8% had poor hydraulic performance. For Norwegian systems built after 1985 when new regulations and loading rates of 2.5 cm/d or less where applied to most systems, no reports of hydraulic failure of properly installed systems have been reported (Gårderløkken, 1997). This suggests that for standard domestic STE applied to gravel-filled infiltration trenches and loading rates <2.5 cm/d, a hydraulic service life of several decades can be expected for WSAS that are designed and constructed today and operated and maintained as needed based on the design specifications. On the contrary, Sherman et al. (1998) showed 18 years as the mean age of failure in three counties investigated in Florida while Keys et al. (1998) predicted that gravel-filled systems in sand soils have a predicted life of 11 years, even when loaded as low as 1.6 cm/d. For other wastewater effluent types and sorted soils, coarse sands and gravels, Jenssen and Siegrist (1991) suggested a conceptual framework for hydraulic loading rates for subsurface treatment systems. However, no data are available on service life for alternative system designs.

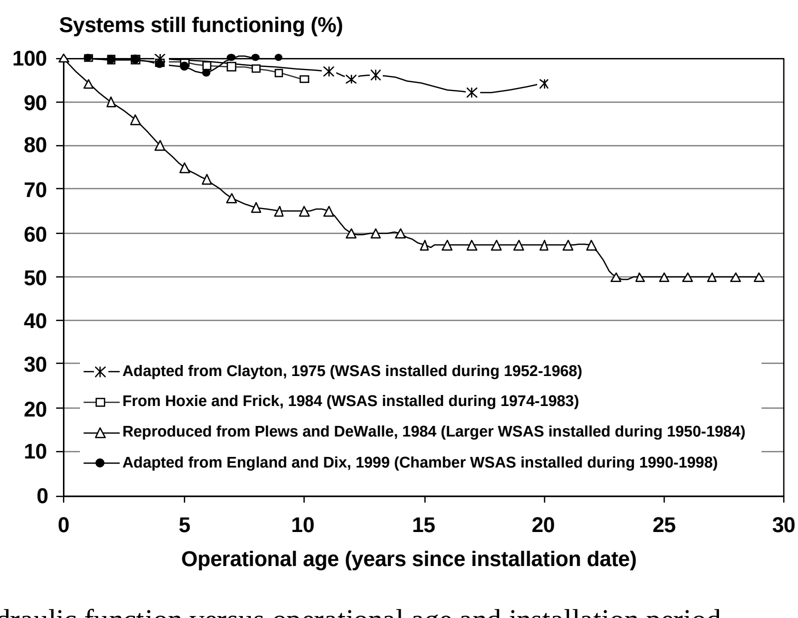


Fig. 8. System hydraulic function versus operational age and installation period.

4.2. WSAS Treatment Potential for Key Pollutant Groups

The common COC's in domestic wastewater and reasonable performance expectations for modern system designs that are properly implemented (i.e., proper siting, design, installation and operation/maintenance) are summarized in Table 2. Additional details regarding three key pollutant groups, (1) organics and suspended solids, (2) nutrients, and (3) pathogens, are given in this section.

4.2.1. Organics and Suspended Solids. Biodegradable organics in either dissolved or suspended form can be characterized by the BOD_5. Volatilization and adsorption, followed by microbial degradation are the main processes for removal of soluble biodegradable organics. Suspended solids, including organic and mineral matter, can be removed through a combination of physical straining and biological degradation processes (Reed et al., 1994). Most soils are effective porous media biofilters due to narrow pores and effective straining of wastewater particles. The large surface area of the soil particles also provides a great potential for biofilm development and infiltration systems are reported to attain maximum efficiency with respect to removal of organic matter as early as of 2 to 3 weeks from the onset of operation (Pell and Nyberg, 1989a,b). Others report a period of 2 to 3 months before the biological degradation potential is fully developed (Van Cuyk et al., 2000). In either case, the start-up phase may be of little consequence to overall public health and environmental protection given the service life of most WSAS is years in length. When viewed over their long service lives, most WSAS can be expected to reliably achieve very high removal of BOD_5 and SS (Hines and Favreau, 1975; Anderson et al., 1985; Effert et al., 1985; Soltman, 1990). Organic chemicals such as volatile organic compounds (e.g., benzene, trichloroethylene) and pesticides can be present in domestic wastewater (Greer, 1987; Kolega et al., 1987; Bicki and Lang, 1991; Sauer and Tyler, 1991; Sherman and Anderson, 1991). However, the concentrations appear to be at trace levels, which do not migrate and pose problems in most WSAS treating domestic wastewater.

4.2.2. Nutrients. In domestic wastewater, typically 70-90 % of the nitrogen is in the form of ammonium ion (NH_4^+) and 10 to 30 % is in organic form (Lance, 1972; Nilsson, 1990; Gold and Sims, 2000). The removal mechanisms for nitrogen in a WSAS include volatilization, ammonification, nitrification/denitrification and matrix adsorption. For a properly installed system, the predominant N-retention reaction would be ammonium adsorption while the predominant transformation reaction would be biological nitrification. The principal removal reactions include biological denitrification and leaching and under certain conditions also chemical denitrification in the ground water zone (Siegrist and Jenssen, 1989). Nitrogen removal in wastewater infiltration systems vary greatly. In general near complete nitrification is achieved in properly installed systems, and nitrification is normally very rapid occurring in the first 30 cm of soil below the infiltrative surface. However, 1-2 months are required from the onset of infiltration to generate a full population of nitrifiers (Pell and Nyberg 1989a,b; Zhu 1998; Van Cuyk et al., 2000). A removal of 10 - 20% of the total nitrogen applied can normally be achieved in conventional WSAS (Siegrist and Jenssen, 1989; Westby et al., 1997; Converse, 1999). Higher removal is possible in mound systems and those with cyclic loading/resting. Westby et al. (1998) found an average of >85% N-removal in dosed mound systems. In systems optimized for nitrogen removal, more than 50% removal can normally be achieved (Lance et al., 1976; Laak, 1982; Siegrist and Jenssen, 1989; Converse, 1999). Phosphorus is typically present in wastewater as orthophosphate, dehydrated orthophosphate and organic phosphorus. Biological oxidation results in conversion of most phosphorus to the orthophosphate forms (Cooper et al., 1996). The main processes for phosphorus removal from wastewater in porous media are adsorption, complexation and precipitation. Most models assume that phosphorus fixation in a PMB occurs in two consecutive kinetic reactions: rapid physical adsorption followed by a slower chemisorption (Tofflemire et al., 1973; Sikora and Corey, 1976; Gold and Sims, 2000). Calcium and oxidized compounds of Fe and Al are known to be important agents for P-sorption in soils. Stuanes and Nilsson (1987) documented that the Fe and Al pools were the most important P-sinks in soils receiving STE. The potential for P-sorption of a porous medium is dependent on the mineral composition and the degree of weathering of the particle surfaces which renders the metals in an oxide or hydrous oxide state where they are able to react with P compounds. In general, soils have variable P-sorption ability (e.g., 0.2 to 1.2 g-P/kg). In a quartz sand the P-sorption capacity of a wastewater infiltration system may become saturated after a few months whereas in weathered sand or fine grained soils (e.g., clays, silt, loam) the sorption capacity may hold for a period of ten years or more. Most studies of P removal have evaluated the sorption potential using equilibrium isotherms, often described by the Langmuir equation (Ellis and Ericson, 1969; Tofflemire et al., 1973; Johnson et al., 1979; Sommers et al., 1979). Experimental results often show that the P-sorption capacity of the PMB is actually much higher than estimated by an equilibrium isotherm (Stuanes and Nilsson, 1987). However, even though many studies assume sorption to be instantaneous, it has been shown by several researchers that this is not always the case (Haseman et al., 1950; Coleman et al., 1960; Davidson and Chang, 1972; Enfield, 1974; Kuo and Lotse, 1974). Overman et al. (1978) reported that the assumption of equilibrium between solution and adsorbed phases in wastewater PMB's was reasonable for lower wastewater flow velocities but less suitable for higher velocities, the latter of which might occur for shallow depths in coarse PMB's or at high loading rates.

4.2.3. Pathogens. Microbiologic COC's commonly found in STE include pathogenic bacteria at sustained high concentrations and virus and protozoa at highly variable and episodically released levels (Cliver, 2000). From a single family home, wastewater bacterial densities are typically quite high with values of 10^8 to 10^{12} organisms per L being commonly encountered (Bicki et al., 1984; Anderson et al., 1985; USEPA, 1978; Haas et al., 1999; Van Cuyk et al., 2000). Of the total bacterial density there can be prevalent, but highly variable, concentrations of pathogenic bacteria like *E. coli*, *Pseudomonas aeruginosa*, *Staphylococcus aureus*, and *Salmonella*. Other pathogens, such as virus and protozoa, are not continuously present at high densities, but rather are shed during disease events and thus the concentration in the wastewater stream at a given home can vary from non-existent levels to values on the order of 10^6 organisms per L or more. Even domestic graywater can contain appreciable levels of pathogens (Siegrist and Boyle, 1982; Rose et al., 1991). Multiple family homes or clusters of individual homes tend to attenuate the episodic nature of pathogen release, but increase the likelihood that the wastewater will contain pathogens at any given time. Pathogens may also be more prevalent in commercial and institutional sources and in some cases, at very high levels (e.g., highway rest areas).

Numerous investigations have studied the transport and fate of bacteria and viruses in soil and ground water under laboratory and field conditions (Romero, 1970; McCoy and Ziebell 1975, USEPA, 1978; Lewis et al., 1982; Harvey and Garabedian, 1991; Tuetsch et al., 1991; Yates and Ouyang, 1992; Harvey, 1997; Higgins, 1999; Oakley et al., 1999; Stevik et al., 1999; Van Cuyk et al., 2000). Studies of bacterial transport/fate have most often employed fecal indicator bacteria such as fecal coliforms or enteric bacteria such as *E. coli*. Studies of virus transport/fate have often been accomplished using bacteriophages such as MS-2 or PRD-1. In general it can be concluded that bacteria and viruses are transported only a few decimeters to meters in the unsaturated zone whereas in the ground water (saturated) zone, they can travel ten to hundreds of meters (Keswick and Gerba, 1980; Keswick et al., 1982; Lewis et al., 1982; Rose et al., 1999). In WSAS removal and inactivation/die-off of pathogens can be extremely effective during STE infiltration through the clogging zone and percolation through the unsaturated flow regime beneath it. The mechanisms for immobilization of bacteria and viruses in WSAS are a combination of straining and adsorption (Peckdeger and Matthess, 1983; Sharma et al., 1985). Straining of bacteria can occur if the pores of the filter are smaller than the bacteria. According to Updegraff (1983) straining becomes an effective mechanism when the average cell size is greater than the grain size d_5 of the soil (d_5 is the diameter where 5% of the particles in mass are smaller and 95% of them are larger). Bouwer (1984) reported that straining occurred when the diameter of the suspended particle was larger than 0.2 times the diameter of the particles constituting the porous medium. A more sophisticated criterion for filtration of bacteria under saturated conditions than the two mentioned above was suggested by Matthess and Peckdeger (1985). Results form straining experiments have shown that in addition to media grain size, straining is controlled by the amount of mechanical and biological clogging of the media, the degree of water saturation, and the hydraulic loading rate (Peckdeger and Matthess, 1983; Corapcioglu and Haridas, 1984). The work of several investigators (e.g., McCoy and Ziebell, 1975; Van Cuyk et al., 2000) has shown that clogging can be of essential importance in pathogen cell removal. This is partly due to reduction of pore size, which induces straining, but also to biotic factors and adsorption, which may increase in importance when clogging, is present.

When the pores are larger than the microorganism, adsorption becomes the dominant retention mechanism; adsorption is therefore of great importance to virus removal. Adsorption of cells to a porous media is dependent on several factors as the content of organic matter, degree of biofilm development, and electrostatic attraction due to ion strength of the solution or electrostatic charges of cell- and particle surfaces (Stevik et al., 1998). Coating of Fe-oxides on media surfaces is shown to enhance adsorption of bacteria and viruses (Keswick and Gerba, 1980). This is due to the Fe-oxides turning the surface charge more positive and thereby increasing the adsorption of bacteria that normally have a negative surface charge at neutral pH. Iron oxides also enhance phosphorus removal (Stuanes and Nilsson, 1987) and hence a positive correlation between phosphorus and bacteria/virus sorption can be expected. Adsorption of microbial cells is a two-step process that can be reversible or irreversible. Reversible adsorption is a weak interaction between the bacteria and the porous media, and the primary forces are electrostatic forces and van der Waals' forces (Mozes et al., 1986). Irreversible adsorption, or adhesion, is a permanent interaction that occurs when bacterial polymers connect the bacteria and the adsorbent (Griffin and Quail 1968; Marshall 1971; Elwood et al., 1982). Die-off of bacteria and inactivation of virus can occur in the adsorbed or in the liquid phases. These processes are affected by biotic and abiotic factors such as soil water content, pH, temperature, organic matter, bacterial species, predation, and antagonistic symbiosis between microorganisms in the system (Yates and Ouyang, 1992; Stevik et al., 1999).

4.3. Factors and Processes affecting Performance

The performance capabilities as noted above are influenced by various factors and their interactions. While it is impossible to isolate a single factor and describe its effect on WSAS performance, Table 3 lists some key factors and the following discussion is given to illustrate the nature and types of effects that can occur. Soil and site conditions are described first followed by system design, installation, and operation/maintenance. For the purposes of this discussion, the WSAS treatment system encompasses the inlet to the soil absorption unit through the lower limit of the underlying vadose zone (see Fig. 3).

4.3.1. Soil and Site Conditions. *Soil properties* such as grain size and pore size distribution, bulk density, porosity, water content, surface area, mineralogy, organic matter content, pH, and microbial biomass and diversity are very important to flow and transport processes in WSAS. Also important are any heterogeneities in these properties with horizontal and depth dimensions. The infiltration zone, unsaturated zone, and ground water zone are all important regions of interest. As noted earlier, long-term acceptance rates have historically been linked with soil texture (e.g., sand, silt loam) and a crude measure of hydraulic capacity (i.e., percolation rate). However, there is little research evidence that has established the relationship between LTAR's and soil properties and some evidence to the contrary. Jenssen (1986) conducted column experiments that revealed that for time periods less than 2 yr., the infiltration rate is dependent on the K_{sat} of the soil. However, for longer times and for soils with an initial saturated hydraulic conductivity below 2500 cm/d (sands range from 50 – 10000 cm/d) the clogging development seems to control the infiltration rate (Fig. 9). Jenssen concluded that there should be little need for differentiation of the loading rate based on soil type for standard gravel-filled soil absorption units receiving domestic STE in fine sands and soils of lower hydraulic conductivity (clayey, loamy and fine sandy soils). However, the actual flux rate through a soil clogging zone can be impacted by the moisture potential underneath it which is controlled by soil texture and unsaturated zone depth, as well as the head of ponded effluent above it (Bouma, 1975).

Table 3. Design and environmental conditions affecting WSAS performance.

Category	Factor/condition	Example performance effects
Soil and site conditions	Soil system properties and heterogeneity within the WSAS	Grain size and pore size distribution, bulk density, porosity, water content, surface area, mineralogy, organic matter content, pH, and microbial biomass can affect flow and transport processes.
	Unsaturated zone depth between the infiltrative surface and ground water	Distance can affect hydraulic function and in turn purification by influencing the soil water content, aeration status, media surface area, and hydraulic retention time.
	Soil temperatures	Can influence soil hydraulic conductivity properties based on viscosity effects; can affect the solubility of dissolved gases such as O_2 and the rate and extent of biological reactions; virus inactivation is highly temperature dependent with higher rates at higher temperatures.
Design features	Effluent application rate and composition	Can affect the rate and extent of clogging at the infiltrative surface, which in turn can affect the hydraulic flow regime and treatment within the WSAS.
	Method of effluent application	Can influence performance depending on the COC and the type and rate of reactions effecting its treatment.
	Depth and geometry of the infiltrative surface	Can affect moisture, temperature, and aeration regimes, and the degree to which diurnal and seasonal variations occur. Can affect degree of biogeochemical reactivity within the infiltration and vadose zones (shallower depth is typically more reactive).
	Infiltrative surface interface characteristics	Presence of an aggregate such as gravel on an infiltrative surface can reduce ISZ permeability by blocking pore entries, becoming embedded in the soil matrix, yielding fines that are deposited in pore entries, or by focusing BOD and TSS as a result of the reduced permeability.
	System size and density of application	As systems become larger and/or the density of application of small systems increases, there is increased potential for adverse hydrologic effects (e.g., ground water mounding) and cumulative pollutant effects.
Construction and operation	Construction practices	Can affect the hydraulic properties of the natural, undisturbed subsurface through compaction, smearing or puddling of the surface due to shear from a vehicular tire or track at the ISZ interface, or deposition of wind-blown materials.
	Age of installation and operational service life	The age of installation is important as the state-of-knowledge and standard of practice have evolved over the past 50 years and systems installed in 1950 are not the same as those installed in 1990. Operational service life combined with age of installation can affect system performance, primarily due to the rate and extent of clogging.
	Operation/maintenance	Changes in wastewater flow/composition from design assumptions can impact overall performance; maintenance through pumping and hydromechanical repair as needed is important to long-term function.

Unsaturated zone thickness beneath a soil absorption unit and the depth to ground water can affect hydraulic function and in turn purification by influencing the soil water content, aeration status, media surface area, and hydraulic retention time. In the U.S., the thickness of the unsaturated zone for WSAS range from 0.6 to 1.2 m and for intermittent sand filters, from 0.6 and 0.9 m (USEPA, 1980a; Anderson et al., 1985; Crites and Tchobanoglous, 1998). A high degree of treatment normally occurs in the infiltration zone as soil clogging develops. However, at higher hydraulic loading rates and with nonuniform distribution methods, constituents of concern that would normally be treated can be transported through the vadose zone to ground water.

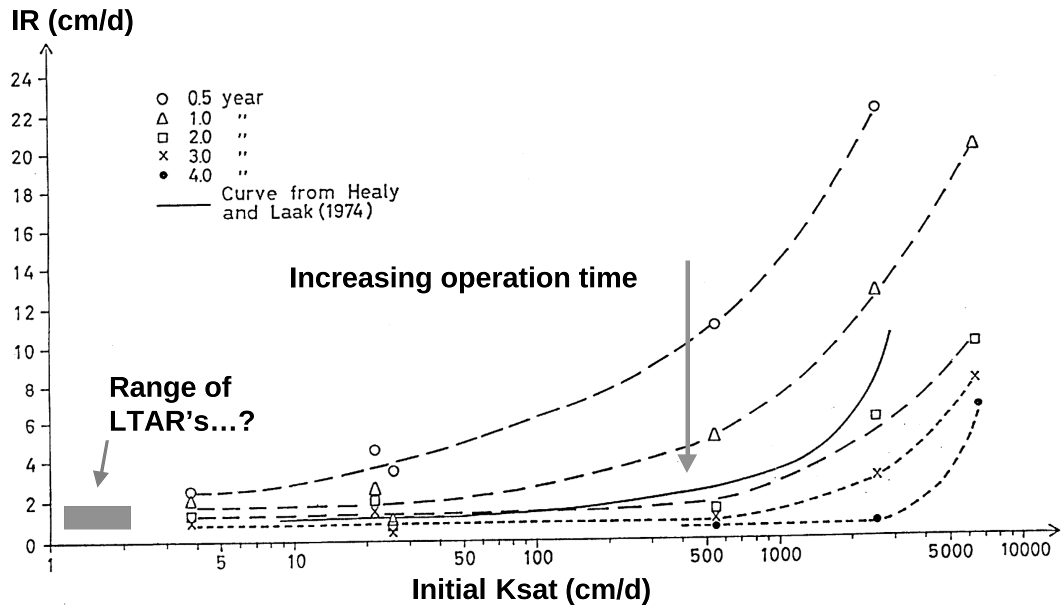
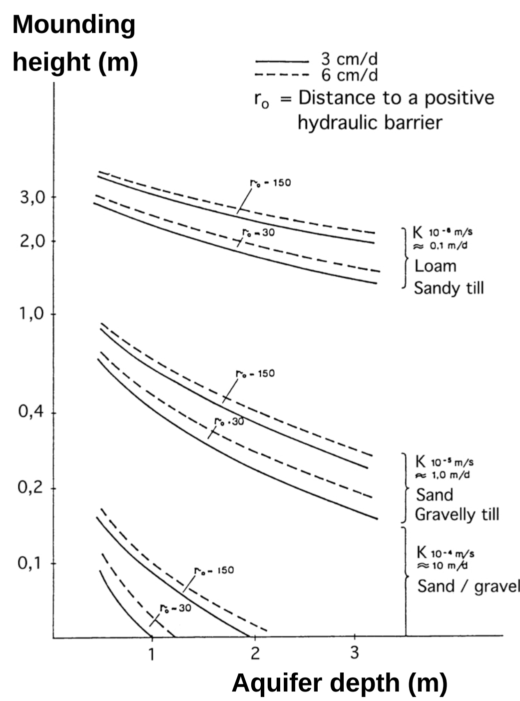


Fig. 9. Time-dependent infiltration rate changes during wastewater application related to the initial saturated hydraulic conductivity of the media for clean water (from Jenssen, 1986).

For example, many studies have shown that a large percentage of bacteria remain near the IS when effluents are applied to porous media (Brown et al., 1979; Kristiansen, 1981; Duncan et al., 1994; Smith et al., 1985; Huysman and Verstraete, 1993; Emerick et al., 1997; Stevik et al., 1999; Van Cuyk et al., 2000). Kristiansen (1981) found no fecal coliform bacteria at more than 30-cm depth below the infiltrative surface of an onsite soil absorption system with a mature clogging zone. Duncan et al. (1994) evaluated the relationship of pretreatment and soil depth on percolate composition and found high removals of fecal coliforms within the first 30 cm for all pretreatment methods. The authors concluded that higher levels of pretreatment could be substituted for increased soil depth. Stevik et al. (1999) found a significant reduction of *E.coli* with soil depth and observed that 99% of *E. coli* was removed in the top 12 cm of 80-cm long columns packed with sand media. Emerick et al. (1997) observed that intermittent sand filters as shallow as 38 cm were capable of removing 90 percent of coliform bacteria from wastewater with a high dosing frequency and a hydraulic loading rate of 4.0 cm/day. Van Cuyk et al. (2000) completed 3-D lysimeter studies and field monitoring of mature systems to quantify the fate of indigenous fecal bacteria as well as viral surrogates (MS-2 and PRD-1 bacteriophages). Lysimeter results revealed breakthrough of fecal coliforms and virus in sand regardless of depth to ground water (60 vs. 90 cm) or infiltrative surface/loading rate scenario (gravel-laden at 5.0 cm/d vs. gravel-free at 8.4 cm/d) (Masson, 1999; Van Cuyk et al., 2000). However, if hydraulic loading rates are too high or the dosing frequency is too low, some microbes can be transported to lower regions in a soil matrix, posing a purification concern in systems that are too shallow to ground water. Alternatively, at some point there is limited additional improvement in purification by increasing unsaturated zone thickness (Peeples et al., 1991).

The thickness of unsaturated soil beneath an IS is not fixed. Rather, it can be quite variable due to changing ground water table elevations associated with seasonal precipitation, or to excess infiltration due to wastewater application. When wastewater is applied to soil, the ground water recharge in the area of infiltration increases. This can result in a local increase in ground water level termed ground water mounding (Hantush, 1967; Fielding, 1982). The ground water mounding is dependent on several factors, such as the saturated hydraulic conductivity of the soil, distance to hydraulic barriers, and depth of the saturated layer (see Fig. 10). The loading rate and design of the system also will influence the height of the ground water mound. In some tills and fine grained soils, the ground water can rise several meters due to wastewater application to a WSAS. Jenssen (1988) therefore defined the hydraulic capacity of a site as "...the amount of liquid per unit time that can be continuously infiltrated without raising the ground water table above an acceptable level". The hydraulic capacity is most likely to be limited in soils of low hydraulic conductivity or shallow depth. In such soils the hydraulic capacity should not be overlooked even when designing small systems. Mound systems are built in shallow soils of low hydraulic conductivity and failure of mound systems (at least in Norway) is due to insufficient considerations of the hydraulic constraints at the site. Ground water mounding must be considered on sites with hydraulic limitations and/or where the wastewater application rate per unit area increases due to clusters of small WSAS or with larger commercial or smaller community systems. Modeling tools are available to aid in the analysis of mounding under WSAS (see Table 4).

Fig. 10. Ground water mounding under WSAS as a function of infiltration rate, aquifer depth and hydraulic conductivity, and distance to a hydraulic barrier (after Jenssen and Bromssen, 1985).



Soil temperatures can influence the hydraulic conductivity properties of a porous medium like soil, due to the effects of temperature on water viscosity. Comparing a temperature of 10C to that of 30C, the hydraulic conductivity is lower and the moisture retention is higher under otherwise comparable conditions. Soil temperature can also affect the solubility of dissolved gases such as O_2. The rate and extent of biological reactions can be described by an adaptation of the Arhennius relationship for chemical reactions which indicates that for a 10C decrease in temperature, the rate of reaction is 50% as fast. Some biological processes (e.g., nitrification) can effectively cease at very low (<10C) or high temperatures (>40C). Virus inactivation is highly temperature dependent with higher rates at higher temperatures (Yates and Ouyang, 1992).

Climate considerations are diverse and include air temperatures, relative humidity, wind speed, precipitation, and so forth. These characteristics can influence the unsaturated zone properties with respect to temperature and water content. Air temperature and relative humidity characteristics can influence the rate of evapotranspiration. This can be an important route for water movement in warm, dry climates. Precipitation in the form of snow can provide an insulating layer on the land surface that can help maintain subsurface temperatures above freezing and enable shallow effluent infiltration all year round. The precipitation characteristics of a region are important as they affect the moisture regime of the subsurface at a site. It not likely that precipitation will dramatically effect system function on sloping sites due to runoff as opposed to infiltration. However, on some sites, precipitation events have been linked to release of COC's such as virus, from a subsurface soil zone.

4.3.2. WSAS Design Features. *Effluent application rate and composition* can affect the rate and extent of clogging which in turn affect the long-term acceptance rate of the soil absorption unit (Siegrist et al., 1987b; Jenssen and Siegrist, 1990; Duncan et al., 1994; Tyler and Converse, 1994). Clogging zone genesis has been shown to be a function of the mass loading rate of wastewater constituents including biochemically oxidizable substances and suspended solids (Siegrist, 1987a,b). Siegrist (1987) completed field experiments with replicated test cells installed in silt loam soils that were operated for nearly 6 years. The observed time-dependent loss in infiltration rate (IR_t) was used to develop an empirical model (Siegrist, 1986; Siegrist and Boyle, 1987). The Siegrist model (Siegrist, 1986; 1987s) estimates the relative infiltration rate (infiltration rate at time t as a fraction of the initial infiltration rate at time t_0, or IR_t/IR_0) based on the cumulative mass density loadings of biochemically oxidizable substances and suspended particulates (Fig. 11). Consistent with these findings, field studies of soil absorption systems receiving aerobic unit or sand filter effluent have shown that soil clogging is highly retarded or absent altogether (e.g., Converse and Tyler, 1998).

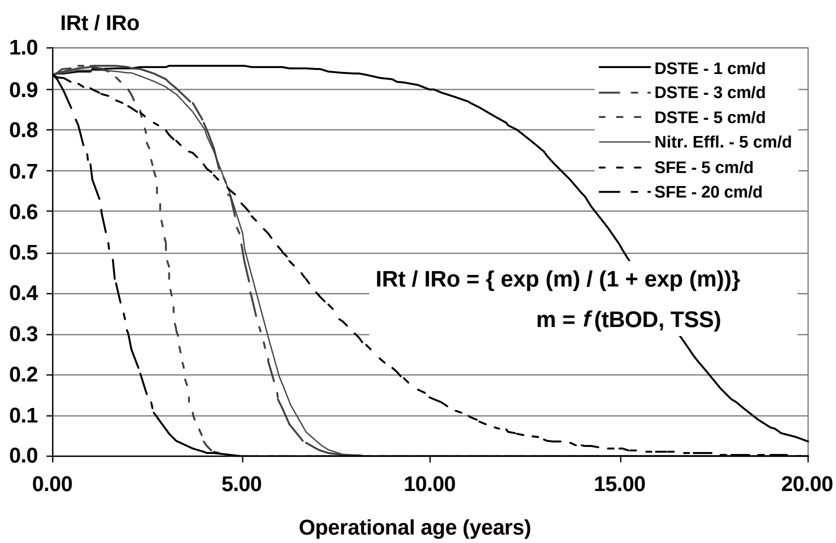


Fig. 11. Illustration of infiltration rate loss in a standard gravel-filled trench based on the cumulative mass density loading of total BOD and suspended solids (kg/m^2) at different hydraulic loading rates and effluent compositions (after Siegrist 1987; Siegrist and Boyle, 1987). (Note: This figure is for illustration purposes only. IRt/IRo values approaching zero do not imply hydraulic dysfunction; also, as IRt/IRo approaches zero and is less than the actual daily loading rate, development of ponding heads up to 30 cm can theoretically increase flux through the infiltrative zone by a factor of 10 to 100 or more depending on clogging zone thickness and resistance).

Slower or absent clogging development with higher quality effluent has led to design approaches that utilize advanced pretreatment (e.g., extended aeration or intermittent sand filtration). This is increasingly being done to enable much higher hydraulic loading rates to be used (10 to 50 cm/d rather than 1 to 5 cm/d) and to reduce the required infiltration area or unsaturated zone thickness. This may be technically sound from an infiltration rate capacity and hydraulic performance basis, but has questionable implications related to purification. While advanced treatment units can reduce BOD_5 and SS loadings and retard wastewater-induced clogging, the concentrations of pathogenic bacteria and virus may not be markedly reduced. Thus absence of a clogging zone may diminish the effective purification of pathogens before ground water recharge. There is no current research that has clearly demonstrated effective pathogen removal in relatively high-rate WSAS with retarded clogging development that are otherwise conventionally designed.

The *method of application*, including the degree of uniformity (on an IS utilization basis) as well as the frequency of application, can influence performance depending the constituent of concern and the type and rate of reactions affecting its removal (see Fig. 12). More frequent application of small doses of STE uniformly applied can yield improved purification with respect to chemical and microbiological constituents (Siegrist and Boyle, 1982; USEPA, 1978; Emerick et al., 1997). This is due to facilitating film flow over particle surfaces and enabling more intimate contact between COC's and media surfaces. As noted below, continuous operation (i.e., year-round) of a WSAS can yield potentially different performance than seasonal or intermittent use, or where long-term resting (e.g., 6 months out of each year) is planned.

The infiltration surface utilization, ISU, is a parameter that describes the fraction of the design or available infiltrative surface that is actually wetted and used for infiltration during operation. The ISU varies from a value at startup (ISUo) to a value at time, t (ISUt) and is a function of the system design including the daily loading rate, the method of application and the hydraulic properties of the natural soil. End members on the ISU continuum (near 0 up to 1.0) include gravity fed systems in high permeability soils (very low ISUo) versus pressure-dosed systems in low permeability soils (very high ISUo). In a mature system that experiences ponding or near-ponding conditions due to clogging development, the ISU approaches 1.0 independent of application method, rate, or soil properties. The ISU is important to treatment as it impacts the flow regime in the vadose zone underlying the infiltrative surface. Two additional related parameters of interest are the HRT and the volumetric utilization efficiency (VUE) of the porous media in the vadose zone beneath the IS. The HRT is important as it determines the time available for reactions to occur. Biochemical treatment reactions such as organic matter degradation, nitrification, and fecal coliform removal can often be described by 1^{st}-order kinetics which relate the concentration at a given depth to that in the applied effluent. The value of the 1^{st}-order reaction rate constant, K , can vary with time and space due to soil clogging development and the accumulation of organic matter and nutrients at the IS and an associated elevated biomass (Nilsson, 1990; Siegrist, 1987a). The rate of reaction for volatilization/sorption/degradation of most constituents is fastest at the infiltrative surface and within the 15 to 30 cm of the vadose zone below it. The purification efficiency predicted by 1^{st}-order reactions is also impacted by the HRT (or t) which is affected by the effluent delivery method and application rate, the soil grain size or pore size distribution, and the degree of soil clogging. Experimental data of Ausland (1998) clearly illustrates the interactions. Ausland completed a series of flow experiments in a large 2-D tank lysimeter (1-m wide by 90-cm deep) using medium vs. coarse sand, point loading vs. uniform distribution, and unclogged, partially clogged, and fully clogged IS conditions. As shown in Fig. 12, the predicted removal efficiencies for unsaturated coarse sand (d_{10}=0.86 mm; d_{60}/d_{10}=1.74) at a daily loading rate of 9.6 cm/d varied from 10% to 100% dependent on conditions. In general, with rate constants on the order of 0.1 to 0.4 hr^{-1} (Ausland 1998), treatment efficiencies of 90% can be achieved with HRT's of 24 hr or less. In research completed by Van Cuyk et al. (2000), four, 3-D lysimeters were studied from startup through nearly one year of operation. Hydraulic and purification behavior was evaluated by routinely monitoring as well as periodic multicomponent surrogate and tracer studies. It was observed that the treatment efficiencies observed in all four lysimeters after the initial 20 weeks of operation were on the order of 90% or higher for COD, ammonium, and fecal coliforms which was not surprising given that the median hydraulic retention times (BT_{50}) were 37 hr or greater. The comparatively lower efficiencies during the first 10 to 20 weeks of operation may be attributed to a lag phase during which initial soil clogging is evolving and active bioprocesses necessary for purification are becoming fully established (e.g., nitrification). McCray et al. (2000) completed model simulations of WSAS showing that soil clogging (base and sidewall) clearly impacts the degree of treatment of certain COC's in the unsaturated zone below the infiltrative surface.

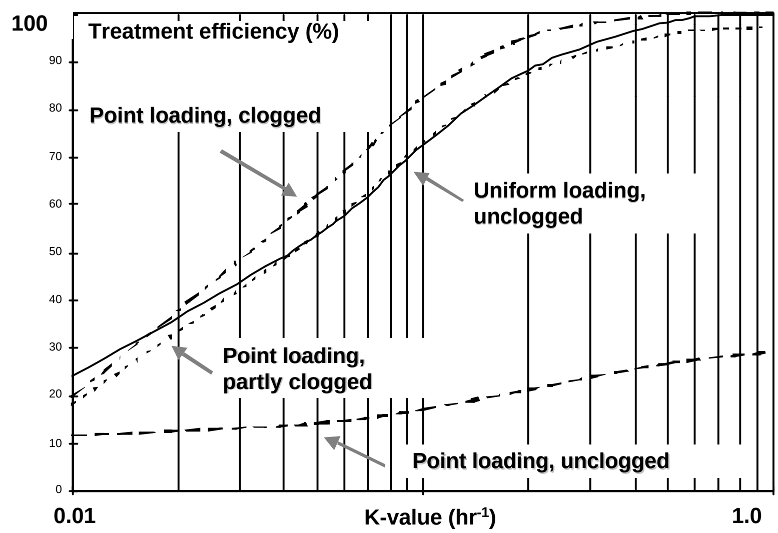


Fig. 12. Purification efficiency as affected by reaction rate and operating conditions which is turn impact hydraulic retention time (after Ausland, 1998).
(Note: data shown are from a 1-m wide by 0.9-m deep, 2-D lysimeter and flow experiments with coarse sand loaded at 9.6 cm/d and Cz/Co = exp(-*K*t), where, Cz = concentration of constituent at depth z from the infiltrative surface, Co = input concentration of constituent at the infiltrative surface, *K* = overall rate constant for combined volatilization/sorption/degradation, and t = hydraulic residence time for percolating water to reach depth z. The rate constant, *K*, is assumed constant with depth).

The volumetric utilization efficiency (VUE) can be defined as the ratio of the volume of media actually contacted by the applied wastewater as compared to the design volume (i.e., the total infiltrative surface area times the media depth) (Van Cuyk et al., 2000). The VUE parameter is directly related to and dependent on the ISU. High VUE's are desirable to enable biofilm and sorption processes, which require adequate surface area contact to achieve a desired removal rate and extent (e.g., phosphorus sorption). In 3-D lysimeter experiments completed by Van Cuyk et al. (2000), the calculated VUE's for week 0 were on the order of 50% hydraulic loading rates and depths that varied by 50% or more. The VUE's at week 8 were nearly 100% suggesting that most of the available horizontal infiltration area was being utilized after an initial two months of early clogging development. Since purification with respect to some constituents such as ammonium and fecal coliforms continued to improve until stabilizing at week 20 or later, it was speculated that there was continued wastewater-induced clogging and a further establishment of purification processes within an operative infiltration zone, rather than an increased expansion of the infiltration area being utilized.

The *depth and geometry of the infiltrative surface* have long been the subject of debate, and to this day, remain poorly understood. Intuitively, shallow placement to exploit the most biogeochemically active zone of the soil profile seems desirable. Narrow trenches that rely on both horizontal and vertically oriented IS areas also appear beneficial. However, very narrow and shallow trenches have limited storage capacity and reduced depth for STE ponding, which may adversely impact long-term hydraulic performance. Other variants include the use of at-grade or mounded systems, the latter being effectively the same as an intermittent sand filter followed by at-grade soil infiltration. An end member incorporating both facets involves the use of drip irrigation in the shallow root zone. All of these systems are often utilized to overcome site conditions that limit the thickness of the unsaturated zone for treatment, such as depth below an IS to a low permeability restrictive layer, seasonal or permanent high ground water table, or shallow bedrock.

The features of the *infiltrative surface interface* can be important to infiltration capacity. The presence of gravel or other media on the surface may reduce the permeability of the infiltration zone due to a variety of factors including (1) deposition of fines that plug pore entries initially or with time, (2) embedment of gravel with the natural soil matrix thereby reducing porosity and permeability, or (3) focusing wastewater organic matter and solids through pore entries that are not plugged or otherwise masked by the gravel. A number of alternatives exist that mitigate the need for gravel within a subsurface infiltration trench or bed. The most common option is a chamber system (Keys, 1996; May, 1996; Tyler et al., 1991) while others are based on fabric-wrapped piping designs.

System size and density of application can affect the system design and performance. As systems become larger and/or the density of application of small systems increases, the potential interaction of wastewater amendment to the landscape needs to be carefully considered. This is due to the hydrologic effects (e.g., mounding) as well as the cumulative pollutant effects and the reduction in assimilative capacity due to simple dilution and dispersion in the ground water. For example, there may be concerns where WSAS application densities are high and they necessarily are located in close proximity to private drinking water wells. Modeling tools (e.g. Table 4) can aid an assessment of the potential cumulative effects and water-shed scale concerns associated with WSAS.

<u>4.3.3. WSAS Construction and Operation and Maintenance.</u> *Construction practices* can affect the hydraulic properties of the natural, undisturbed subsurface. These effects include: (1) compaction due to vehicle traffic on the exposed IS or the dumping of gravel aggregate onto it, (2) smearing or puddling of the surface due to shear from a vehicular tire or track at the IS interface, (3) deposition of wind-blown fines while the infiltrative surface is exposed, and (4) smearing and compaction resulting from construction when the soil is too wet (i.e., soil water content exceeds field capacity) (Tyler et al., 1985).

The *age of installation and operational service life* of a WSAS can greatly influence its performance capabilities. The age of installation is important as the state-of-knowledge and standard of practice have evolved over the past 50 years. Systems installed in 1950 are not the same as those installed in 2000. Operational service life combined with age of installation can affect system performance, primarily due to the rate and extent of wastewater-induced clogging. The operation age of a system includes the actual loading rate and continuity of use (i.e., the LF and CF). In general, systems that are loaded near design specifications and used continuously will mature more rapidly in time than those that are underutilized. With operation, wastewater-induced clogging increases the effective area for infiltration and the degree of unsaturated flow conditions in the underlying soil, as well as create a biogeochemically active zone for treatment to occur. Van Cuyk et al. (2000) completed 3-D lysimeter studies to quantify the hydraulic and purification processes during the first year of system operation. Lysimeter results revealed relatively lower hydraulic retention times and vadose zone utilization during the first months of startup with breakthrough of fecal coliforms and MS-2 and PRD-1 viral surrogates in sand WSAS regardless of depth to ground water (60 vs. 90 cm) or infiltrative surface/loading rate scenario (gravel-laden at 5.0 cm/d vs. gravel-free at 8.4 cm/d) (Masson, 1999; Van Cuyk et al., 2000). After 10 months of operation, the percolates were of much higher quality in terms indigenous fecal coliforms (<10 org./100mL) and specific pathogens (non-detect) as well as viral surrogates. Thus, the operational aging process appears quite important to treatment efficiency and raises questions about the treatment performance of WSAS with discontinuous operation (e.g., at seasonal dwellings or with cyclic loading/resting operation).

Operation and maintenance are critical to the performance of a WSAS. If actual conditions such as daily wastewater flow rate or wastewater composition are different than design assumptions then systems can be overloaded and hydraulically fail. Even if the design matches actual conditions, if the WSAS is not properly maintained (e.g., repair of a broken pipe, replacement of a failed pump, periodic pumping of septic tank solids), then dysfunction and failure can result.

4.4. Modeling WSAS Hydraulic and Purification Performance

The performance *capabilities* of wastewater soil absorption systems and the factors affecting them are important, but the ability to *reliably predict* the performance to be achieved under a given set of design and operational conditions is perhaps even more critical. While relatively limited, there are conceptual and mathematical models that specifically relate the performance metrics of a WSAS (e.g., infiltration capacity and soil clogging, purification efficiency, or service life) to site conditions (e.g., soil properties, soil depth, temperature), system design (e.g., IS geometry, depth, or interface character, effluent application rate and composition), and operation (e.g., continuous or discontinuous operation) (see Table 4).

Application of mathematical modeling for describing and predicting performance of WSAS as affected by process design factors and environmental conditions continues to advance with quantitative relationships emerging, being refined, and/or validated including:

$$\text{LTAR} = f\,(\text{clogging zone genesis}), \tag{1}$$

$$\text{CZG} = f\,(\text{HLR, quality and MLR, application method, IS, } K_{sat}, {}^{o}\text{C, ...}), \tag{2}$$

$$\text{ISU} = f\,(\text{IS features, application method, and soil } K_{sat}), \tag{3}$$

$$\text{VUE} = f\,(\text{ISU}), \tag{4}$$

$$\text{Sorption efficiency} = f\,(\text{VUE, media properties, depth), and} \tag{5}$$

$$\text{Reaction efficiency} = f\,(\text{kinetics } K\text{, HRT, } {}^{o}\text{C, ...}) \tag{6}$$

where, LTAR = long-term acceptance rate, CZG = clogging zone genesis, HLR = hydraulic loading rate, MLR = mass loading rate, IS = infiltrative surface, K_{sat} = saturated hydraulic conductivity, ISU = infiltrative surface utilization, VUE = volumetric utilization efficiency, K = reaction rate constant, HRT = hydraulic retention time.

While challenging to fully develop and validate, expression of quantitative understanding through mathematical relationships and models can enable effective practice regarding WSAS design and implementation and support development and confidence in performance-based codes. Moreover, such understanding and single-site scale modeling tools are required to properly account for WSAS cumulative impacts, if any, to public health and environmental quality in applications where there are clusters or subdivisions of individual systems. Finally, such knowledge is needed to enable development and allocation of total maximum daily loads (TMDL's) for watershed-scale environmental protection (Chen et al., 1999). Selected modeling tools that have been utilized for onsite WSAS applications as well as a few others that potentially could be adapted are summarized in Table 4.

Table 4. Mathematical models and decision-support tools for design and performance of onsite wastewater soil absorption systems.

Model	Model Type / *Scale*	Developer / Reference	Model description and/or waste related application and results
----	WSAS soil clogging development *Site scale*	Siegrist (1986) Siegrist and Boyle (1987)	Estimates IR_t/IR_o based on cumulative mass loadings of total BOD and suspended solids. Specific results include time-dependent loss in IR results from input of hydraulic loading rate (cm/d) of wastewater flow and concentrations of total BOD (cBOD + nBOD) and SS,
SepTTS	WSAS chemical fate/transport *Site scale*	Lee et al. (1998)	Screening level tool for predicting fate and transport of down-the-drain household chemicals in septic systems.
VIRALT	Well-head protection model for virus *Site scale*	Bechdol et al. (1994)	Bechdol et al. (1992) conducted simulations of ground water in Rhode Island and found that the model was only capable of distinguishing risks between widely different situations. However, coarse textured soils and aquifers common to the coastal watersheds are very susceptible to virus transport. Field monitoring and validation was recommended.
VIRTUS	Virus transport and fate in unsaturated zone *Site-scale*	Yates and Ouyang (1992)	Simultaneously solves eqns. describing transport of water, heat and virus through unsaturated zone of soil. Predictive model of virus fate that allows virus inactivation rate to vary based on soil depth and temperature changes. Tested on datasets in laboratory columns with MS-2 coliphage transport.
----	Bacterial transport *Site-scale*	Harvey and Garabedian (1991)	Bacterial transport in ground water simulated using a colloid filtration model that had been modified to include advection, storage, dispersion and adsorption.
----	Microbial transport *Site-scale*	Teutsch et al. (1991)	One-dimensional model to describe microbial transport that includes decay, growth, filtration and adsorption. During lab-scale tank experiments with MS-2, the predictions closely matched measured results at high flow rates, but did not match at flow rates.
MANAGE	Decision support tool for aquifer vulnerability *Watershed scale*	Kellogg et al. (1997) Loomis et al. (1999) Joubert et al. (1997)	Used to identify ground water pollution sources, future threats, and evaluation effectiveness of various wastewater improvements.
WARMF	Decision support system for TMDL development *Watershed scale*	Chen et al. (1999)	A decision support system to calculate TMDL's for a watershed. Incorporates cumulative effects of onsite systems into pollutant loading as nonpoint sources.
BASINS	Multipurpose environ. analysis system *Watershed scale*	Lahlou et al. (1998)	Used by regional, state, and local agencies in performing watershed-based studies to facilitate examination of environmental information; to support analysis of environmental systems, and to provide a framework for examining management alternatives. Onsite systems can be incorporated as nonpoint sources.
DRASTIC	Ground water sensitivity *Site scale*	Aller et al. (1985) Stark (1997)	Ranking system that evaluates 7 hydrologic factors (depth to water table, aquifer net recharge, aquifer media, soil media, topography, impact of vadose zone, hydraulic conductivity) to yield a numerical index of an area's relative degree of potential for pollution. Has been used to evaluate aquifer sensitivity to nitrate pollution from onsite systems.
3S-NPoP	GIS-based conceptual *Watershed scale*	Stark (1997)	Relates the potential impacts at the source areas to the nitrate levels in the stream, linking septic system site characteristics to water quality.
HYDRUS	Unsaturated flow and solute transport *Site scale*	Simunek et al. (1996) Schwager and Boller (1997) McCray et al. (2000)	Schwager and Boller (1997) Investigated solute and gas transport under intermittent flushing conditions. Found that single flush size and frequency might considerably affect the performance of sand filters by impacting oxygen flux within systems. McCray et al. (2000) simulated 2-D conditions related to flow and transport in WSAS under a range of conditions.
MOFAT	Multi-phase flow and transport code *Site scale*	Kaluarachchi and Parker (1991) Schwager and Boller (1997)	Investigated solute and gas transport under intermittent flushing conditions. Found that single flush size and frequency might considerably affect the performance of sand filters by impacting oxygen flux within systems.
----[2]	(ADI) finite diff. approx. of unsaturated flow *Site Scale*	Ewing et al. (1985)	Studied the effects of unsaturated flow, inflow period length, and horizontal variations in vertical flow rates to determine the design factors having the greatest influence on flow conditions within buried sand filters. Found that application rate, retention times, and depth of the sand filter had the greatest impact on unsaturated flow conditions within the filter medium.
TOPLATS [2]	Nonpoint source *Watershed scale*	Endreny and Wood (1999)	Determines source areas of nonpoint source pollution within a watershed using a water table-driven variable source area (VSA) routine to determine runoff zones. Used for contamination from agricultural land use, but could be used to determine extent of contribution of contaminants from onsite systems.
----	GIS-based *Site scale*	Lasserre et al. (1999)	Models nitrate flux through the unsaturated zone and transport through ground water. Used for contamination from agricultural land use, but could be used for nitrate transport from onsite systems.
FLUNIT [2]	GIS-based *Watershed scale*	Van den Brink et al. (1995)	To evaluate ground water protection strategies based on risk analysis and effectiveness of possible measures. Nitrate concentration changes in the subsurface from agricultural land use sources.
SPARROW [2]	Surface water quality *Watershed scale*	Preston & Brakebill (1999)	Relates in-stream water quality measurement to spatially referenced characteristics of watersheds, including contaminant sources and factors influencing terrestrial and stream transport. Has been applied to nitrogen modeling from point sources, urban areas, fertilizer application, manure generation and atmospheric deposition.
MORELN [2]	Nitrogen cycle *Site scale*	Geng et al. (1996)	Models the nitrogen cycle and nitrate leaching in soil and simulates nitrate migration in an aquifer system. Has been used on agricultural lands.

[1] These models were not developed for wastewater soil absorption systems, but potentially could be adapted to that application.

5. Risk Assessment/Management Applied to Wastewater Soil Absorption Systems

5.1. Risk Assessment and Risk Management Framework

5.1.1. Risk Assessment. Wastewater possesses inherent hazards due to pollutants in it, which can cause adverse effects such as human disease (e.g., due to pathogenic bacteria), or ecosystem upsets (e.g., eutrophication due to nutrient input and fish kills from ammonia input and oxygen depletion). The inherent hazards become a risk when wastewater effluent containing pollutants is hydrologically linked to receptors and the transport/fate processes are not effective in reducing the concentrations at the point of exposure below a threshold which does not cause an adverse response. Formal risk assessment (RA) procedures have evolved from various applications such as organic chemicals and heavy metals in soil and ground water (e.g., Labieniec et al., 1997; Omenn et al., 1997a,b). The general RA process involves characterization of the inherent hazards, the pathways and transport/fate processes, and the routes of exposure and exposure-response properties of the receptor in question (Omenn et al., 1997a,b; Labieniec et al., 1997).

A *site-specific risk assessment* involves a single system comprised of a particular design at a particular location (e.g., classic WSAS for permeable soil conditions at a specific residence). A *generic risk assessment* involves either (1) a population of systems of the same design and location types (e.g., mound wastewater absorption systems for sites with shallow ground water) or (2) a population of different systems distributed at different types of sites within a prescribed space-time domain of interest (e.g., spatial boundaries based on a political jurisdiction such as a county or a hydrologic boundary such as a watershed, and time boundaries for old vs. recent vs. new systems). There is inherent variability and uncertainty in the information regarding site conditions, system design implementation, and system performance as well as the transport/fate to a receptor and the exposure/response characteristics. *Deterministic risk assessments* involve point estimates for the different input parameters, which are then combined to yield an estimate of the risk. Deterministic assessments often employ highly conservative input values for all inputs and as a result, the estimated risk is often highly conservative, and protective of failure in a high percentage (e.g., 99%) of the situations. *Probabilistic risk assessments* utilize distributions for the input parameters, which propagate variability and uncertainty into a distribution of estimated risk. The estimated risk distribution can then be used in risk management decision-making. Formal quantitative risk assessment has rarely been applied to onsite wastewater soil absorption systems (Jones, 2000).

5.1.2. Risk Management. Risk management (RM) involves identifying and choosing between options that might be necessary and appropriate to the mitigate the identified risk to an acceptable level. Risk management options can be addressed at one or more parts of the risk framework from the wastewater source, treatment system design and implementation (which includes engineering design, siting, construction, operation/maintenance), to the pathways and transport/fate processes to exposure points, to the receptors themselves. As noted earlier, performance goals have not been explicitly stated for WSAS. Rather a goal statement might read something like: "A proper WSAS is expected to process all wastewater generated with adequate purification provided by the prescribed system design for a long service life with little O&M." This type of goal statement implies that a properly performing system will successfully process all generated wastewater without seepage of partially treated effluent to the ground surface or back-up of wastewater into a dwelling unit. Moreover, it includes slow percolation through a vadose zone before recharge to a local ground water without causing excessive ground water mounding which could reduce the unsaturated zone depth and adversely affect purification processes.

Performance of WSAS involves both hydraulic and purification function and their interactions (Schwager and Boller, 1997; Van Cuyk et al., 2000). Adequate hydraulic performance can be defined as processing all of the wastewater flow without backing up into the dwelling or seepage to the ground surface. Purification performance can be defined as achieving a given concentration of a COC at a point of assessment. The receiving environment for treated wastewater normally includes the local ground water system, which may be connected to regional ground water systems, and/or surface waters such as rivers, lakes, and estuaries. The sensitivity of the receiving environment to perturbation must be considered.

For risk management, design of a WSAS, whether it be a classic system or one with modifications or an alternative treatment train, should be based on a set of performance goals to mitigate an explicit risk to an acceptable level. For example, a performance goal for the WSAS could be established such that the concentration of NO_3-N in soil solution entering the ground water under a WSAS is below a level where dilution and dispersion in the ground water will reduce the level of NO_3-N at a conceivable point of drinking water extraction does not exceed 10-mg-N/L (the drinking water MCL). In order to design and implement a system or systems to meet that goal requires an understanding of process function and performance with respect to wastewater nitrogen. The design to achieve the goals must account for the natural constraints of a site, the robustness of the WSAS to deviations from design assumptions, as well as the need for and ability to deliver requisite O&M in order to reliably achieve performance capabilities.

5.2. RA/RM for Onsite Wastewater Soil Absorption Systems

Formal risk assessment and risk management have rarely applied to decision-making for onsite systems. However, risk assessment and risk management concepts have been implicit in WSAS practices based on the regulations and code structures controlling their use. Recently explicit risk-based decision-making been advocated (Otis and Anderson, 1994; Hoover et al., 1998a,b; Loomis et al., 1999; Jones, 2000). Most applications of WSAS are based on prescriptive codes (e.g., promulgated at the state level and enforced at the county level) which in turn have been developed based on historical and empirical information which have evolved into a local practice that the contractors in an area are able to accomplish. Modifications have been developed as well as alternative system designs, to enable onsite system use in site conditions that are not suitable for a classic system design. When first introduced, innovations can be permitted for limited use under provisional or experimental programs. Normally an absence of reported problems can lead to a general use approval. In some cases, rigorous monitoring and formal documentation of operation and performance of alternatives have been required or have occurred due to a research interest.

Risk assessment and management concepts implicitly are involved in decision-making for WSAS. For example, most prescriptive codes are designed to constrain WSAS applications to situations where an adverse effect will not occur or be manifested by actions such as:

- Prescribing site conditions that are believed to be suitable for WSAS of a certain design. Site conditions usually involve metrics on landscape position, slope, subsurface permeability and unsaturated soil depth and possibly methods of assessment (e.g., percolation test by a licensed soil analyst),
- Establishing setback distances to points of exposure and receptors, most notably drinking water wells and surface waters,
- Prescribing design features, such as size of a soil absorption unit and related unit operations, and
- Specifying monitoring and assessment methods (usually only for larger systems).

Within the code-based implementation structure, decision-makers still must chose between different system design types and attributes. As shown in Table 5, there are a number of scenarios with WSAS where the risk varies from negligible to high. High risk situations tend to occur where the performance capabilities of the WSAS are deficient or uncertain, and where the environmental situation is sensitive with respect to receptors and exposures. In these and other situations, there are a number of management strategies and options that can be implemented for a given situation and the implicit or explicit risk assessment thereof (Table 6).

Table 5. Example scenarios for wastewater soil absorption systems that pose apparent risks to human health and environmental quality and example risk management options.

Example scenario	Risk characterization	Level of concern and basis	Traditional risk management options	Alternative risk management options
(1) Low density applications of conventional WSAS designs in environmental settings with thin vadose zones of coarse soils or shallow fractured rock and proximal private or public drinking water wells.	Health risk to residents due to ingestion of drinking water containing nitrates and pathogens, and to visitors from pathogens.	Moderate due to inherent treatment limitations of conventional WSAS under the environmental conditions.	o Holding tanks o Prohibit development	o Alternative WSAS designs (e.g.,. mounds) o Advanced treatment before conventional WSAS
(2) High density applications of conventional WSAS designs in environmental settings with thin vadose zones of coarse soils or shallow fractured rock and proximal private or public drinking water wells.	Public health risk due to ingestion of drinking water containing nitrates and pathogens	High due to inherent treatment limitations of conventional WSAS under the environmental conditions and the multiple sources contributing.	o Provide public water from a safe source o Provide sewers and a central treatment plant	o Alternative WSAS designs (e.g.,. mounds) o Advanced treatment before conventional WSAS
(3) Low or high density applications of WSAS on landscapes and soils with low permeability and seasonal saturation or flooding.	Health risk to residents, neighbors and visitors due to seepage to the ground surface and direct or indirect contact and ingestion or inhalation of pathogens.	High due to uncertainty in design and performance relationships for conventional WSAS under the environmental conditions.	o Accept intermittent seepage o Increase septic pump out during wet periods o Increase WSAS size	o Alternative WSAS designs (e.g., mounds) o Advanced treatment before conventional WSAS o Advanced treatment and disinfection for surface discharge
(4) Cesspools and old seepage pits located on the shores of sensitive receiving waters such as lakes and estuaries.	Environmental risk to ecosystem including degradation of water quality and biota habitat.	High due to inherent limitations of treatment under conditions.	o Provide sewers and central treatment plant	o Upgrade existing onsite systems with advanced treatment or alternative WSAS designs.
(5) WSAS that are seasonally used and do not develop adequate soil clogging and biofilm growth to achieve treatment for pathogens.	Health risk to residents, neighbors and visitors due to drinking water ingestion of drinking water from wells contaminated by pathogens.	High due to uncertainty about treatment in seasonally used systems.	o None since not recognized	o To be determined
(6) WSAS that are designed for higher LTAR's based on advanced treatment units that reduce BOD and SS and thereby retard soil clogging.	Health risk to residents, neighbors and visitors due to drinking water ingestion of drinking water from wells contaminated by pathogens.	High due to uncertainty about treatment of pathogens in WSAS receiving advanced treatment unit effluent.	o None since not recognized	o To be determined

Table 6. Example risk management strategies and options for wastewater soil absorption systems at the site-scale and watershed scales.

Risk management strategy	Example option	Desired impact(s) or effect(s) of option	Availability and implementability of option	Implementation & reliability of option	Impact if option fails or has a dysfunction
1. Conventional WSAS design and implementation	A. Conventional narrow trench design (e.g., narrow trenches with shallow placement)	Control flux of pollutants into ground water while treating all wastewater generated without seepage to ground surface or backup into dwelling	Design and product based options that are widely available May be constrained by site and/or environmental conditions	Design, installation and operation by trained practitioners	Excessive risk to receptors dependent on site and possibly watershed conditions
2. Source control	A. Remove toilet waste and treat graywater by soil absorption (e.g., compost toilet)	Reduce pollutant loading to WSAS, particularly N and pathogens	Product and/or engineered option Commercially available	Good unless toilet waste system (e.g., compost toilet) is circumvented by user	Loss of effect on specific pollutant loadings plus system hydraulic overloading
3. Modified or alternative WSAS design and implementation	A. Increased conservatism in design and operation (e.g., reduce loading rate from 4 to 2 cm/d)	Reduce soil hydraulic application rate; or increase vadose depth;	Engineering option May be constrained by lot size, environmental conditions, and/or cost	Design and installation by individuals/firms normally involved in onsite systems	Loss of safety factor with default back to performance of conventionally designed system (1A).
	B. Alternative soil absorption system designs (e.g., at-grade or mound systems or drip irrigation)	Increase vadose zone depth; increase chance of nutrient removal	Engineering option May be constrained by lot size, environmental conditions, local competency, and/or cost	Requires design and installation by trained professional or reliability is poor	Excessive risk to receptors dependent on site and possibly watershed conditions
	C. Advanced treatment before soil absorption (e.g., sand filtration but not disinfection)	Reduce pollutant loading to WSAS, particularly tBOD, TSS, N	Product and/or engineering option May be constrained by local competency, and/or cost	Requires O&M by trained professional or reliability is poor	Increased pollutant loadings to WSAS with potential for hydraulic and purification dysfunction and increased risk to receptors
	D. Advanced treatment before soil absorption (e.g., sand filtration with UV disinfection)	Reduce pollutant loading to WSAS, particularly tBOD, TSS, N and pathogens	Product and/or engineering option May be constrained by local competency, and/or cost	Requires O&M by trained professional or reliability is poor	Increased pollutant loadings to WSAS with potential for hydraulic and purification dysfunction and increased risk to receptors
4. Land use and exposure/receptor controls	A. Separation and setback distances (e.g., 200 ft. to drinking water well rather than 100 ft.)	Increase separation to receptors to enable dilution/ dispersion/ reactions to reduce exposure concentrations	Requires ability to model and predict environmental transport/fate and exposure benefits of increased separation	Requires site characterization and land use data and modeling expertise	If predictions are inaccurate desired benefits on exposure concentrations will not be realized and risks will be higher than predicted
	B. Increased lot size or reduced density of application (e.g., reduce density from 4 DU/acre to 0.5 DU/acre)	Reduce cumulative effects to receiving environment and enable dilution/ dispersion/ reactions to reduce exposure concentrations	Requires ability to model and predict environmental transport/fate and exposure benefits of reduced density	Requires site characterization and land use data and modeling expertise	If predictions are inaccurate desired benefits on exposure concentrations will not be realized and risks will be higher than predicted
5. Monitoring	A. Monitoring (e.g., pump and level sensors, vadose lysimeters, ground water wells)	Detect dysfunction and implement corrective action before an adverse effect occurs	Hydromechanical functions can be readily monitored but purification and subsurface conditions are difficult and costly	Design, installation and also telemetric and/or onsite sampling/analysis by trained professional	Missed dysfunction and effects could occur and be undetected

6. Critical Questions and Research Needs

6.1. Questions and Areas of Research Need

Despite a history of use and a considerable body of research and observations from practical experiences, further research is needed to support the long-term, effective use of onsite and decentralized WSAS. A nationally coordinated program of research is needed to produce an enhanced understanding of WSAS hydraulic and purification processes and their complex interactions including the effects of design, operational and environmental factors. Key questions remain at the site-scale up to the multiple-site to watershed scale. Table 7 provides a listing of research questions, while some additional remarks are given below.

- Almost all of the research needs related to WSAS have an underlying common theme: there is a need for quantitative understanding that enables rational process design and performance relationships to be modeled for predictive purposes. This is critical to our ability to move beyond empirical studies that provide information that is highly constrained to the soil and site conditions, design and operational factors, and environmental conditions of the particular study in which the data were generated. Experimental work needs to appropriately span reasonable space and time scales and incorporate modeling facets to help provide the needed insight into WSAS processes and performance and rationale methods of design and implementation. The WSAS field needs quantitative relationships and validated models and decision-support tools.

- Basic research is needed to understand clogging zone genesis and performance effects. Clogging zones are dynamic biogeochemical zones, the characteristics of which can be affected by design factors (e.g., wastewater pretreatment and loading rate, application method, IS geometry and features) and environmental conditions (e.g., soil pore size distribution, soil organic content and pH, soil wetness and temperature, soil microbial biomass and diversity). The clogging zone is known to be extremely important to hydraulic and purification processes in WSAS and there is some understanding of its time-dependent development based on composition and loading rate as well as some of its physical/chemical and microbiological properties. However, further understanding is required. In particular, it is critical to understand the role of clogging zones in reliably achieving high degrees of removal and die-off/inactivation of pathogens (bacteria and virus) both within the clogging zone itself or the underlying unsaturated soil. Application of modern environmental chemistry methods and molecular biology tools and approaches may greatly aid elucidation of the underlying processes and their effects.

- The understanding of clogging zone development must be translated into a workable design practice for WSAS including support for prescriptive- and performance-based codes.

- The complex relationships of IS features and geometry, wastewater composition and loading rate, and the method of wastewater delivery and application, and their effects on the rate and extent of soil clogging and the hydraulic and purification performance of WSAS needs to be elucidated. Fundamental information would enable rational design choices in areas such as sidewall vs. bottom IS area, open vs. aggregate-laden surfaces, shallow vs. deep placement, gravity vs. dosed application, dosed vs. pressure-dosed delivery, and uniform vs. serially loading.

- Research is needed to fully understand the effects of unsaturated zone depth including the effects of transient operational or environmental conditions, on WSAS performance. For example, seasonal saturation that reduces the vadose zone depth may temporarily reduce treatment efficiency, but over the life of the system it may not be consequential from a risk perspective. Also, seasonal fluctuations in soil temperature, moisture potential, and aeration are of concern with respect to their effects on microbial transport.

- o Site evaluation practices and their translation into WSAS design need refinement. The use of the percolation test continues despite compelling evidence that it is subject to huge errors and the values measured (MPI) have no fundamental relationship with flow processes in new or mature WSAS.

- o Operational discontinuity including planned periodic resting or random intermittent or seasonal use may have benefits with respect to retarding clogging development, but it may conversely have adverse effects on purification. Research is needed to understand operational discontinuity effects on clogging zone development or degradation once present, and the effects on purification of pathogens.

- o Reviews and studies to date have not been extensive, but they do suggest there is very little concern over heavy metals and organic chemicals (e.g., petrochemicals and chlorocarbons) in domestic septic tank effluent. However, there may be other chemicals of concern such as endocrine disruptors from contraceptives or other products used in dwellings and these warrant some investigation.

- o There are a range of modifications and alternatives to classic system designs and in many cases an absence of fundamental understanding. Some of these don't entail great cost nor cause harm if they fail to perform at a claimed level and thus decisions regarding their use are not complex. However, some approaches or technologies can pose a cost-benefit decision and even a risk consequence if dysfunction or failure occurs. Thus, performance and benefit/cost data are needed for AOU's.

- o Knowing more about the underlying processes in WSAS and their effects on treatment (as described above) would enable definition of critical parameters that could be used for assessment of operational state and performance. With such parameters defined and if there were reliable methods to monitor for critical assessment parameters and even communicate this information using telemetry, system performance could be tracked, dysfunctions detected early, and failures or serious adverse effects prevented. While hydromechanical functions can be monitored successfully at this time, there remain problems with the necessary strategies for and methods to be used regarding bacteria and virus.

- o The correct measures of system performance that provide a desired risk reduction must be defined and the means by which the requisite measures can be made, both cost-effectively and reliably, needs to be determined. There is also a need to develop effective and efficient monitoring and measurement methods that can be used to identify and diagnose WSAS operation and performance status. Such an understanding would enable diagnostic strategies for identifying individual systems within a cluster or community that are in need of upgrading or replacement.

- o At the multiple site to water shed scale of applications, there is a great need for information on the impact of onsite WSAS on surface and ground water quality. Decision-makers are confronted by continuing debates over questions such as how to establish minimum lot sizes, how to determine and defend setback distances, discriminating out the WSAS contributions of nutrients and pathogens to receiving waters (drinking water wells, bathing lakes, etc.), and so forth. Moreover, with continuing concerns and resulting drinking water regulations (e.g., source water protection and ground water disinfection) as well as the establishment of TMDL's for watersheds, there will be an increasing need for understanding as well as models and decision-support tools that are applicable, process-based, and readily useable for the multiple site up to watershed scale applications.

- o National, integrated studies using systematic state-of-the-art methods needs to be completed to define the hydraulic and purification performance of classic, modified and alternative systems under a range of conditions. This study could be completed by a multi-institutional team that has access to and use of demonstration projects and test facilities located in many parts of the U.S. Such study could include design and implementation process review, installation and operation records review, onsite inspections, sampling and analysis, and microbial surrogate/chemical tracer testing.

6.2. Prioritization of Research Needs

Prioritization of the various research needs was completed by the authors of this white paper using a multi-criteria, weighted evaluation scheme (e.g., Kepner and Tregoe, 1973). Five attributes were identified as important to assessing the priority of a research need and a weight was assigned to each. These included: (A) level of current science and technology understanding (wt. = 0.3), (B) level of importance to design and performance of new WSAS (wt. = 0.2), (C) impact if the research is completed and the need is satisfied (wt. = 0.3), (D) feasibility and cost of completing the work (wt. = 0.1), and (E) known or suspected work already in progress (wt. = 0.1). Before scoring the various needs listed in Table 7, some were first combined with other closely related needs. Then for each of the research needs, each attribute (A to E) was considered and given a numerical score of 1, 3 or 5 points based on explicit metrics (see Table 8). The weighted summary score was finally computed for each research need yielding the results shown in Table 8. The needs were then ranked in four priority groups (i.e., very high, high, moderate, and low). This group ranking was based on similarity of scores between research needs and the breakpoints in the numerical ranking.

7. Conclusions

Onsite and decentralized wastewater treatment in the U.S. relies on subsurface infiltration and percolation through the unsaturated zone prior to ground water recharge. These systems have evolved during the 20th century from early designs that were focused on simple disposal to contemporary designs that are intended to achieve advanced treatment. While the experience base does not suggest serious or broad-based problems with recent or current WSAS practices, it also does not demonstrate consistently adequate performance based on an established design and implementation understanding. Moreover, it is possible that WSAS technology is not being exploited fully and/or as effectively as it might, or alternatively, inappropriate and deficient applications may be evolving. There is a considerable knowledge base regarding WSAS design, implementation, and performance, that enables most systems to be protective of public health and environmental quality. However, understanding is not fundamental enough to discriminate between different approaches to WSAS system design and implementation, such that rational decision-making will lead to the most cost-effective approach for reducing risk to an acceptable level. Of the many research needs identified, several are given high priority because of their judged importance. As shown in Table 8, high and very high priority research needs include those that support: (1) fundamental understanding of clogging zone genesis and unsaturated zone dynamics and their effects on treatment efficiency, particularly pathogenic bacteria and virus, (2) development of modeling tools for predicting WSAS function and performance as affected by design and environmental conditions, (3) identification of indicators of performance and methods of cost-effective monitoring, and (4) development of valid accelerated testing methods for evaluating long-term WSAS performance.

Table 7. Research questions for wastewater soil absorption systems. [1]

Strategic focus	Tasks	Products	Uses
I. Function and Design Needs			
1. What is the effect of pretreatment on soil clogging zone genesis and WSAS hydraulic and purification performance?	A. Conduct experiments with instrumented WSAS systems of the same total size but with optional levels of pretreatment such as (1) domestic STE, (2) graywater STE, (3) sand filter effluent, and (4) aerobic unit effluent, and define IRt / IRo and pollutant fluxes through a given depth of unsaturated soil over time. Utilize surrogate/tracer testing to also assess treatment potential for virus and other non-routine constituents of concern.	1. Quantitative data and empirical and/or mechanistic models for LTAR's and treatment efficiency. 2. Relationship of unsaturated soil depth to treatment efficiency for different soil systems with different degrees of soil clogging.	(a) Support design of specific systems and development of performance and prescriptive code requirements. (b) Provide input for cumulative effects and watershed scale assessment models. (c) Support decisions regarding use of advanced pretreatment with higher application rates under otherwise comparable design and site conditions.
	B. Conduct experiments to assess pathogen purification as affected by clogging zone biogeochemical activity levels as a function of pretreatment level, temperature, and operational age. Apply advances in environmental chemistry, microbiology and biotech molecular methods to understand fundamental nature of WSAS and cause/effect relationships.	1. Information on the role and importance of the clogging zone to pathogen purification (especially virus) and public health protection.	
2. What is the relationship of infiltrative surface character on short- and long-term hydraulic properties of the infiltrative surface zone?	A. Conduct factorial designed experiments to explore the main and interaction effects of (1) vertical vs. horizontal surfaces and (2) gravel vs. gravel free surfaces, on infiltration capacity over time and pollutant fluxes in the underlying vadose zone.	1. Information on the relative behavior of different infiltrative surface features on infiltrability and LTAR's and pollutant fluxes to ground water.	(a) Support design to achieve optimum LTAR's without compromising treatment efficiency.
3. What is the relationship between clogging zone genesis and the resultant loss in infiltration rate over time with common STE WSAS design, operation, and environmental factors?	A. Conduct factorial designed experiments to evaluate the effects and interactions of (1) hydraulic loading rate, (2) application method, (3) infiltrative surface character, (4) soil texture/structure, (5) soil temperature, and (6) soil moisture conditions, on development of an LTAR.	1. Semi-quantitative and quantitative data and process models that describe infiltrative surface utilization and volumetric utilization efficiency with time and the HRT within the soil.	(a) Predict ISU, VUE, and HRT, which can be used to predict treatment efficiency.
4. What is the treatment efficiency achieved in a WSAS designed with different methods of application of domestic STE?	A. Conduct experiments with instrumented WSAS systems of the same size but with optional methods of delivery and distribution including (1) gravity/trickle, (2) serial loading, (3) dosed, and (4) pressurized uniform distribution. Identify pollutant fluxes through a given depth of soil over time.	1. Quantitative data on pollutant fluxes through a given depth of unsaturated soil over time.	(a) Provide information for use in prescriptive codes. (b) Support performance-based design.
5. For a population of similar WSAS in a similar environmental setting, what is the time-dependent relationship between performance and age of operation (a.k.a., service life)?	A. Survey large populations of systems with stratified random sampling methods to assess hydraulic performance and develop % functioning vs. age of operation relationships.	1. Fraction functioning vs. age of operation curves for different WSAS designs in different environmental settings.	(a) Support benefit/cost analyses of decentralized vs. centralized systems. (b) Input for performance guarantees, inspection, and certification programs.
6. How is treatment efficiency affected by transient and extreme environmental conditions?	A. Conduct monitoring and experiments of treatment efficiency as affected by ephemeral or seasonal saturation in the vadose zone beneath a WSAS.	1. Information on the adverse effects if any of ephemeral or seasonal saturation in the vadose zone beneath a WSAS.	(a) Provide basis for design for common but challenging conditions.
	B. Conduct monitoring and experiments of treatment efficiency as affected by very low and very high soil temperatures.	1. Information on the temperature dependency of WSAS function.	
7. What models are appropriate for predicting treatment efficiency as a function of siting, design, and operation?	A. Utilize databases produced by experimental work (I.1 to I.6) to support model development and validation for application at the single site to multiple site and watershed scales.	1. Modeling tools for screening level assessments to quantitative prediction of performance.	(a) Provide information for use in prescriptive codes. (b) Support performance designs. (c) Enable cumulative effects assessment and TMDL allocations.
	B. Develop a methodology for evaluating the degree of model complexity required for a given decision-making situations.	1. Decision-logic for selecting one modeling approach over another in a given situation.	

Table 7. cont. Research questions for wastewater soil absorption systems. [1]

Strategic focus	Tasks	Products	Uses
II. Site Evaluation Needs			
1. What is the relationship between natural soil profile properties and the hydraulic capacity of a single site? Alternatively stated, what are the essential field data needed to support understanding and/or modeling of unsaturated flow and hydraulic capacity?	A. Develop methods to directly measure the soil pore size distribution at the local-scale and based on scaling theory, estimate the relevant site-scale value.	1. Methods and apparatus to directly estimate pore size distribution.	(a) Input to models on water and gas flow and pollutant transport in the vadose zone beneath a WSAS. (b) Support siting and design including landscape loading effects on degree of saturation and ground water mounding.
	B. Develop relationships between morphology and indirect measures (e.g., penetration resistance) with pore size distribution in different soil environments.	1. Methods and apparatus to indirectly estimate pore size distribution.	
	C. Develop methods and apparatus to enable a more accurate calculation of hydraulic conductivity and capacity of low permeability soils and soils that have shallow ground water tables.	1. Apparatus and methods for determining hydraulic conductivity properties in low permeability media.	
2. What methods can be used to assess the hydraulic capacity of a site for larger and clustered WSAS applications?	A. Evaluate existing hydrologic assessment approaches (e.g., for spatial variability and heterogeneity), field techniques (e.g., geophysical methods), and modeling tools for their applicability and reliability for WSAS applications.	1. Approach to assessment and list of applicable assessment and modeling tools.	(a) Evaluate sites and develop landscape configurations for WSAS to avoid exceeding site capacity.
3. How can the natural soil properties that impact wastewater-induced soil clogging development be assessed in the field during site evaluation?	A. Utilize experimental data generated in I.3.A. to assess impact and relationship of soil properties to design application rates and LTAR.	1. Matrix of soil properties and appropriate design application rates.	(a) Support rational sizing of infiltrative surfaces.
4. What methods can be applied to assess the treatment capacity of a site for nutrients, bacteria and virus?	A. Develop and validate testing techniques that can reliably measure the effectiveness of a given profile for treatment of key constituents of concern.	1. Test methods for assessing treatment effectiveness.	(a) Support system design for a given treatment objective. (b) Provide more accurate methods to assess site capacity and treatment capabilities. (c) Input to flow and transport models for treatment efficiency.
5. What methods can be used to estimate the contribution of existing or new WSAS to pollutant loads in a watershed or sub-watershed?	A. Utilize modeling and decision-support tools produced from I.7.	1. Modeling tools for screening level assessments to quantitative prediction of performance.	(a) Enable assessment of cumulative effects (b) Enable explicit incorporation of WSAS in TMDL assessment and allocations.

Table 7. cont. Research questions for wastewater soil absorption systems. [1]

Strategic focus	Tasks	Products	Uses
III. Performance Monitoring Needs			
1. What is an appropriate methodology for defining the space-time domain for evaluating performance of one or many WSAS, that is protective of public health and environmental quality in a given setting?	A. Develop a paradigm for the WSAS as a treatment unit and what the appropriate space and time dimensions are within the context of public health and environmental receptors and risks.	1. Improved performance monitoring approaches that enable risk assessment and management.	(a) Support development and application of permits and monitoring approaches to verify compliance.
2. What are easily measured “indicators” of WSAS function that can be used to predict treatment performance?	A. Based on system function, identify key indicators, and test those indicators under varied conditions.	1. Information on readily measured system functional attributes that are indicative of treatment performance.	(a) Support application of relatively cheap and reliable monitoring and performance assessment methods.
	B. Conduct studies to develop correlations between percolate fluxes estimated from analysis of soil solids versus soil solution for nutrients, bacteria and virus.	1. Statistical relationships for alternative sampling and analysis approaches (e.g., soil coring vs. soil solution sampling for N, P, fecal bacteria, virus).	
	C. Conduct studies to determine the utility and reliability of online monitoring of water quality (air composition) indicators (e.g., pH, Ec, D.O.) and treatment performance.	1. Statistical relationships for indicators vs. treatment parameters in WSAS.	
3. What methods can be reliably used to provide data on purification performance and the flux of pollutants, particularly bacteria and virus, from a WSAS into an underlying ground water?	A. Conduct studies of methods and statistical data analysis approaches for monitoring in the vadose zone and ground water beneath single systems and small clusters of systems.	1. Methods and apparatus for monitoring and performance assessment for single sites and small clusters.	(a) Monitoring of existing and new systems suspected to have poor or inadequate treatment to target upgrade or replacement needs. (b) Routine monitoring of WSAS to verify compliance with permits. (c) Monitoring of WSAS to verify pollutant allocations made as part of TMDL’s in a watershed.
	B. Conduct studies of methods and statistical data analysis approaches for monitoring in the vadose zone and ground water for larger developments and watersheds. Carryout out field testing to validate viability and utility of methods and approaches.	1. Methods and apparatus for monitoring and performance assessment for cumulative effects in larger developments and watersheds.	
	C. Conduct controlled studies to determine the nature and reliability of the correlation between fecal coliforms and pathogens (infectious bacteria and enterovirus), and assess the validity of fecal coliforms as an indicator.	1. Information on the validity of fecal coliforms as an indicator of pathogen treatment in WSAS.	
4. What methods are available to assess the treatment performance directly or by estimation, of old WSAS of unknown design and installation, and operational history?	A. Test methods and approaches developed in III.1 to III.3 at old systems in different environmental settings.	1. Methods and apparatus for monitoring and performance assessment for old unknown WSAS.	(a) Monitoring of existing and new systems suspected to have poor or inadequate treatment to target upgrade or replacement needs.
5. What is the role and impact of remote sensing and monitoring on performance assurance for decentralized systems?	A. Conduct studies of cost/benefit of remote sensing and SCADA technology for controlling and monitoring WSAS function and performance.	1. Information on available methods, costs, and viability.	(a) Provide information to support development of contract services in the private and public sectors. (b) Enables performance-based design and implementation.

Table 7. cont. Research questions for wastewater soil absorption systems. [1]

Strategic focus	Tasks	Products	Uses
IV. Other Supporting Needs			
1. What is the composition of the effluent produced by different types of emerging tank-based treatment units?	A. Complete monitoring of test systems or full-scale installations to define effluent concentration distributions for key constituents in the effluent from different treatment units such as (1) add-on units to septic tank, (2) spin-disk filters, (3) foam and textile filters, (4) peat filters, (5) LECA filters, and (6) aerobic package plants. Key constituents include those that affect soil clogging development (e.g., tBOD and SS) and also constituents of public health and environmental quality concern.	1. Information on the effluent composition characteristics (central tendency and spread of concentrations) and reliability of alternative pretreatment units as compared to septic tank treatment prior to WSAS.	(a) Provide input to analysis of effects of pretreatment on soil clogging and performance effects. (b) Provide input to risk reductions afforded by increased pretreatment prior to WSAS.
2. What is the composition of the effluent produced by different types of emerging tank-based treatment units?	A. Conduct international literature review and if needed, complete monitoring of source separation systems.	1. Information on the effluent composition characteristics (central tendency and spread of concentrations) and reliability of source separation to produce a stated quality of effluent (e.g., graywater STE) prior to WSAS.	(a) Provide information on the benefits and reliability of source separation to achieve pollutant mass reductions and risk reductions therefrom.
3. What methods can be used to restore the infiltrative capacity of a WSAS with excessive wastewater-induced soil clogging?	A. Conduct studies of alternative approaches based on knowledge of soil clogging genesis such as (1) long-term resting, (2) improved pretreatment, (3) forced aeration, (4) physical disruption, and (5) chemical amendments.	1. Methods and equipment to restore excessively clogged WSAS such as occurs with very old, overused, or commercial systems.	(a) Restore dysfunctioning WSAS and mitigate adverse health effects due to exposure to partially or untreated wastewater.
	B. Develop a methodology to diagnose WSAS function and performance dysfunctions and select the most effective restoration technique.	2. Information and methods for assessing hydraulic dysfunction and choosing a restoration technique.	(a) Support continued use of WSAS and prevent unnecessary replacement of systems at higher cost and disruption compared to that of restoration.
4. What short-term tests can be used to predict long-term performance?	A. Conduct comparative experiments with accelerated loading schemes and based on findings, evaluate implications of time scales on testing of new technologies.	1. Practical test procedures that allow short-term testing of technologies with long functional service lives	(a) Test protocols for evaluating WSAS system performance to satisfy regulatory demands and/or certification programs
5. What improvements in performance of WSAS can be attributed to training and certification programs?	A. Conduct studies to evaluate the improvements in performance that have occurred and can be attributed to training and certification programs.	1. Information on value and need for training and certification programs.	(a) Support state and local decisions regarding training and certification programs.

[1] Research needs for WSAS as outlined in Table 7 were intentionally developed to exclude the facets of implementation such as education and training, materials and construction quality, regulatory/certification programs, system management, septage management, and so forth. While these are recognized as important and relevant to WSAS design and performance, but beyond the scope of this research needs white paper.

Table 8. Prioritization of research questions into ranked research needs.

Research question and need		Attribute score[1]					Weight. score[1]					Total	Rank by	Priority	Comments
No.	Description	A	B	C	D	E	A 0.3	B 0.2	C 0.3	D 0.1	E 0.1	score	score[2]	group[3]	
I.1.A.	Effect of pretreatment IRt/IRo and fluxes	3	3	5	3	5	0.9	0.6	1.5	0.3	0.5	3.80	6	H	
I.1.B.	Pathogen fate vs. soil clogging genesis	5	5	5	5	3	1.5	1.0	1.5	0.5	0.3	4.80	1	VH	
I.2.A.	I.S. character and hydraulics	3	5	3	3	3	0.9	1.0	0.9	0.3	0.3	3.40	13	M	Needs I.2.A., I.3.A., and I.4.A. are related and were grouped for ranking
I.3.A.	Effects/interactions of STE on LTAR														
I.4.A.	Effects of application methods on trtment														
I.5.A.	Survey large populations for service life	5	3	1	3	5	1.5	0.6	0.3	0.3	0.5	3.20	16	M	
I.6.A.	Treatment effects of seasonal saturation	3	3	3	3	5	0.9	0.6	0.9	0.3	0.5	3.20	16	M	Needs I.6.A. and I.6.B. are related and were grouped for ranking
I.6.B.	Treatment effects of temp. extremes														
I.7.A.	Model development from databases	5	5	3	5	3	1.5	1.0	0.9	0.5	0.3	4.20	4	VH	Needs I.7.A. and I.7.B. are related and were grouped for ranking
I.7.B.	Develop methodology for model selection														
II.1.A.	Methods to directly measure pore size	3	3	1	5	3	0.9	0.6	0.3	0.5	0.3	2.60	19	L	Needs II.1.A., II.1.B., and II.1.C. are related and were grouped for ranking
II.1.B.	Methods to indirectly estimate pore sizes														
II.1.C.	Hydraulic cap. of LPM and shallow soils														
II.2.A.	Evaluate hydro assessment methods/tech.	3	5	3	5	3	0.9	1.0	0.9	0.5	0.3	3.60	8	H	
II.3.A.	Field assessment of soil effects on LTAR	3	5	1	3	3	0.9	1.0	0.3	0.3	0.3	2.80	18	L	
II.4.A.	Testing techniques for treatment potential	3	5	3	5	3	0.9	1.0	0.9	0.5	0.3	3.60	8	H	
II.5.A.	Estimate pollutants loads to watersheds	3	5	3	3	3	0.9	1.0	0.9	0.3	0.3	3.40	13	M	
III.1.A.	Paradigm for WSAS treatment unit	5	5	1	5	5	1.5	1.0	0.3	0.5	0.5	3.80	6	H	
III.2.A.	Key indicators of system function	3	5	3	5	3	0.9	1.0	0.9	0.5	0.3	3.60	8	H	Needs III.2.A., III.2.B., and III.2.C. are related and were grouped for ranking
III.2.B.	Correlation's for solids vs. percolate														
III.2.C.	Utility of online water quality sensors														
III.3.A.	Monitoring of VZ/GW for small appl.	3	5	3	5	1	0.9	1.0	0.9	0.5	0.1	3.40	12	M	Needs III.3.A. and III.3.B. are related and were grouped for ranking
III.3.B.	Monitoring of VZ/GW for larger appl.														
III.3.C.	Correlation's of FC and pathogens for WSAS	5	5	3	5	3	1.5	1.0	0.9	0.5	0.3	4.20	4	VH	
III.4.A.	Test methods for old systems	5	1	3	3	5	1.5	0.2	0.9	0.3	0.5	3.40	13	M	
III.5.A.	Cost/benefit of remote sensing/SCADA	5	5	3	5	5	1.5	1.0	0.9	0.5	0.5	4.40	3	H	
IV.1.A.	ATU effluent profiles	3	3	1	3	3	0.9	0.6	0.3	0.3	0.3	2.40	21	L	
IV.2.A.	Source separation systems lit review / mon.	3	3	1	5	3	0.9	0.6	0.3	0.5	0.3	2.60	19	L	
IV.3.A.	Restoration of clogged systems	3	1	1	3	5	0.9	0.2	0.3	0.3	0.5	2.20	22	L	Needs IV.3.A. and IV.3.B. are related and were grouped for ranking
IV.3.B.	WSAS performance dysfunction diagnoses														
IV.4.A.	Accelerated loading testing	5	5	5	3	3	1.5	1.0	1.5	0.3	0.3	4.60	2	VH	
IV.5.A.	Performance benefits of training/certif.	3	5	3	3	5	0.9	1.0	0.9	0.3	0.5	3.60	8	H	

[1] Prioritization categories and attributes with points: Total score = sum{(wt.A)(A) + (wt.B)(B) + (wt.C)(C) + (wt.D)(D) + (wt.E)(E)}

A = Level of current science and technology base (wt. = 0.3)

1 = Mechanistic understanding exists; 3 = Empirical data exists for WSAS under common conditions or relevant understanding in allied fields; 5 = Limited/qualitative understanding exists for a few conditions.

B = Level of importance to design and performance of "new" WSAS (wt. = 0.2)

1 = Not very critical, confirms expected/likely performance with more data; 3 = Enables semiquantitative design/analysis; 5 = Enables rational design and/or performance assessment.

C = Impact if research is completed and need is satisfied (wt. 0.3)

1 = No extension or use or risk reduction for current applications likely; 3 = Reduces risk uncertainty regarding WSAS uses; 5 = Enables WSAS applications under new situations with known risk.

D = Feasibility and cost of completing work (wt. = 0.1)

1 = Uncertain and/or very high cost; 3 = Can surely be done but at mod. to high cost; 5 = Has been / can be done at low cost and/or has been done in other fields and can be applied to WSAS with leverage.

E = Known or suspected work in progress (wt. = 0.1)

1 = Programs or projects ongoing that should produce needed results; 3 = Programs or projects ongoing that should produce part of needed results; 5 = No known programs ongoing or pending.

[2] Numerical ranking is based on weighted attribute scoring with the highest weighted score yielding the highest rank in terms of priority of research need.

[3] Ranking into four groups is based on similarity of scores \ and breakpoints in the numerical ranking: Very high (VH) = $\geq$4.0; High (H) = 3.5 to 3.9; Moderate (M) = 2.9 to 3.4; and Low (L) = 2.8 and below.

8. Acknowledgments

This paper was prepared with contributions made by several individuals and organizations. Students, staff and faculty at the Colorado School of Mines are acknowledged for their valuable contributions in contributing to the tables and figures in the paper, as well as providing reviews of the draft manuscripts. The USEPA and the National Decentralized Water Resources Capacity Development Project are acknowledged for their sponsorship of the preparation of this and related white papers and the convening of a national workshop on research needs. The constructive comments received from the white paper reviewers, Dr. Aziz Amoozegar and Dr. James Converse, are acknowledged, as are those received from attendees at the national workshop convened in May 2000 at Washington University in St. Louis, Missouri.

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10.0 Appendix - Acronyms, Abbreviations and Symbols

AOU	- add-on unit
ATU	- advanced treatment unit
AUE	- aerobic unit effluent
BOD_5	- 5-day biochemical oxygen demand
BT_{10}	- breakthrough at Ce/Co = 0.10
cBODult	- ultimate carbonaceous BOD
CF	- continuity factor
cfu	- colony forming unit
COC	- constituent of concern
Ce	- concentration in percolate
Co	- concentration applied and/or at t=0
Cz	- concentration at depth, z
CZG	- clogging zone genesis
D.O.	- dissolved oxygen
DSTE	- domestic septic tank effluent
EPRI	- Electric Power Research Institute
gpd	- gallons per day
gpcd	- gallons per capita per day
GIS	- geographic information system
GW	- ground water
H	- high
HLR	- hydraulic loading rate
HRT	- hydraulic retention time
IRo	- infiltration rate at time, 0
IRt	- infiltration rate at time, t
IS	- infiltrative surface
ISU	- infiltrative surface utilization
ISUo	- infiltrative surface utilization at t=0
ISZ	- infiltrative surface zone
K	- reaction rate constant
Ksat	- saturated hydraulic conductivity
L	- low
LF	- loading factor
Lpcd	- liters per capita per day
Lpd	- liters per day
LPP	- low pressure pipe
LTAR	- long-term acceptance rate
M	- moderate
MLR	- mass loading rate
MPI	- minutes per inch
nBOD	- nitrogenous BOD
N	- nitrogen
NSF	- National Sanitation Foundation
O&M	- operation and maintenance
P	- phosphorus
pfu	- plaque forming unit
PMB	- porous media biofilter
RA	- risk assessment
RM	- risk management
SFE	- sand filter effluent
SS, TSS	- suspended solids
STE	- septic tank effluent
t	- time
tBOD	- total BOD (cBODult + nBOD)
TMDL	- total maximum daily loading
USEPA	- U.S. Environ. Protection Agency
UV	- ultraviolet light
VH	- very high
VOCs	- volatile organic compounds
VUE	- volumetric utilization efficiency
WSAS	- wastewater soil absorption system
z	- depth below the infiltrative surface

Peer Reviews

The preceding White Paper, *Design and Performance of Onsite Wastewater and Soil Absorption Systems,* by R.L. Siegrist, E.J. Tyler, and P.D. Jenssen was solicited for peer review. Reviewers comments are provided in this section.

Aziz Amoozegar, Professor
Soil Science Department
North Carolina State University

The authors have done an excellent job in reviewing various aspects of decentralized (onsite) wastewater management, and compiling a list of research needs. A copy of the manuscript with editorial comments and questions will be submitted to the authors. Comments on the technical aspect of the manuscript and research needs are listed.

Technical Comments

- The low pressure pipe (LPP) system is superior to conventional gravity fed and pressure manifold systems for uniformly distributing wastewater to the trenches. Two of the problems with LPP systems are the masking of the holes by gravel aggregate and root intrusion. The effect of the partial blockage of the holes on LPP lines by gravel is shown in Fig. 4 of the article "Performance evaluation of pressurized subsurface wastewater disposal systems, Amoozegar et al., 1994, in Proc. of the 7th Intern. Symp. of Indiv. and Small Comm. Sewage Systems, ASAE." To overcome gravel shadowing and root intrusion problems, an LPP pipe can be placed in a section of 4-inch corrugated pipe installed in a gravel envelop in the trench (something that is currently practiced). This method of installation will also allow replacement of a line without digging up the trench if the pipe becomes dysfunctional (e.g., holes become clogged).
- I believe Hargett's study was conducted using 4-inch wide trenches. We have shown that 4-inch wide trenches have problems, and in North Carolina wider LPP trenches are used to overcome some of the problems. The main problem with 4-inch wide trenches is the lack of storage for occasional over use (shock use) or during high rainfall events. Eight- to 12-inch wide trenches should be used for LPP.
- Smearing of the trenches during construction due to high soil water content needs to be addressed.
- The mathematical modeling section is too technical, and reads like a research journal article explaining the models. For this White Paper, the modeling approaches need to be explained without getting into the technical aspects of the models.

Research Needs

Function and Design Needs

- Design of septic systems should be done in the context of assessing the entire system of soil and not the infiltrative surface. There are three basic areas of hydrological evaluation that must be addressed. (1) Wastewater applied to trenches must infiltrate the soil. (2) Wastewater infiltrating the soil must move vertically through the unsaturated zone before encountering a water table or a restrictive layer. (3) Wastewater entering the water table or reaching a restrictive layer must move laterally away from the drainfield area. Failure in any of these three areas will result in hydraulic failure of the system. Therefore, granting reduction in drainfield size for pretreatment or use of gravel-less trenches must be assessed carefully.

- Item #3. Hydraulic conductivity of the zone below the infiltrative surface must also be considered.
- Item #6. Assessment of the impact of very low and very high temperature can only be accomplished under seasonal conditions at a given site. Since soil temperature (as part of climate) is one of the soil forming factors, it will be extremely difficult, if not impossible, to separate the impact of temperature from that of soil on the functioning of WSAS. Therefore, impact of temperature should be assessed in conjunction with other seasonal variations (e.g., soil water content).

Site Evaluation Needs

- Item #1. The idea of developing methodologies and designing devices for assessing pore size distribution, soil hydraulic conductivity, and other pertinent soil hydraulic properties with relative ease and speed is well justified. Currently, there are devices and approaches that can be modified and/or refined for field assessment of various soil hydraulic properties. These devices and approaches, however, need to be tested and verified for application to site evaluation.
- Item #2. The hydraulic capacity of a site must be related to the long-term acceptance rate for designing WSAS. Furthermore, the assessment protocols should be addressed at national level. There should be some sort of uniformity in the protocols if we expect the decentralize systems to become acceptable as an alternative to centralized sewage treatment plants. In this context, soils should be considered as a system and not be assessed only on the basis of infiltrative surface. The use of a standardized technique, such as Ksat measurement should be studied, and the relationship between Ksat and LTAR must be determined.
- Item #4. Statistical (regression type) models can be developed for predicting the transport of various constituents of wastewater.
- Item #5. We need to assess the impact of WSAS on surface and ground water quality at watershed scales. The data are needed for developing model(s) for screening purposes as well as for delineating the contribution of septic systems from other nonpoint sources.

Performance Monitoring Needs

- We need to come up with national standards as what is considered treatment, and also what parameters we need to identify for achieving our goals.
- Item #5. We need to develop non-intrusive techniques, including remote sensing, for assessing WSAS. Examples of non-intrusive techniques are using ground penetrating radar to locate drainlines and soil wetness, and gas monitoring to assess denitrification.

Other Supporting Needs

- We need to develop standards that go beyond the NSF standards for assessing tank-based (pre) treatment units.
- Item #3. In addition to developing methods for restoring infiltrative capacity of the WSASs resulting from wastewater-induced clogging, we need to address the loss of infiltrative capacity and hydraulic conductivity due to soil dispersion resulting from chemicals in wastewater.
- Separation of blackwater from graywater and the level of treatment that can be achieved also needs to be studied.
- Onsite reuse of wastewater for indoor (e.g., toilet flushing) and outdoor (e.g., irrigation) must also be studied. Currently, there are systems that address total reuse. In light of water shortage in many areas, reuse of wastewater may be unavoidable for many areas of the country.
- Gravel shadowing and the impact of treated wastewater on hydraulic properties of the soils (e.g., infiltration rate) must be studied more rigorously. Aggregate-free systems, and systems with pretreatment receive a substantial reduction in the size of the drainfield area. Many soils have a limiting layer at 30 to 60 cm below the depth where the infiltrative surface for an onsite system will be located. The limiting layer will control the vertical flow from the site and is independent of the infiltrative capacity of the trench.
- Impact of duration of saturation and degree of wetness on the performance of onsite systems must be studied. Many areas (e.g., eastern United States) have soils that become saturated for a short period of time during the year.
- Impact of onsite systems on ground and surface water quality and separating the contributions of onsite systems from fertilizer applications to home lawns and ornamental plants at watershed scale need to be evaluated.
- Large onsite systems are used for subdivisions, hotels, apartments, schools, and other facilities with design loading rates in excess of few thousand gallons per day. The functioning of these systems, their efficiency in removing pollutants, and their impact on soil, ground water and surface water need to be studied.

James C. Converse
Biological Systems Engineering
University of Wisconsin-Madison

This paper is an excellent compilation of the literature relating to onsite wastewater soil absorption units. It appears to be very complete relative soil absorption systems.

The research needs for wastewater soil absorption systems (Table 6) is very comprehensive including strategic focus, tasks, products, and uses. The research needs have been subdivided into four categories. They are quite broad based, but at the same time somewhat specific which gives the researcher some latitude in designing a research proposal and gives guidance to the funding committee as to what the needs are. It is important to prioritize the research needs as funding is limited, but at the same time the funding committee needs to be cognizant of the fact that the top priority in one part of the country may not be the same elsewhere.

There needs to be a balance between laboratory research, controlled research, and somewhat random field research. What may perform well in a controlled research activity may not perform that way in the real world. Laboratory and controlled research are needed to provide for the model development and field evaluation provides a more accurate data base and verification of the models.

We need to have a good understanding of what is really happening in the real world. Are there risks associated with well designed and properly operating systems? We need to do a comprehensive evaluation as to what the risks are. Table 6 makes reference to developing a comprehensive study and Table 5 gives risks associated with onsite systems.

I could list specific research needs, but most or all would be associated with one of the items listed in strategic focus as the strategic focus is very broad based. I will list some of the items that came to mind as I studied the proposal.

- Determine the relationship between MPN/g dry soil vs. colonies/100 ml. If we are going to work in the vadose zone, we need to evaluate the soil and not try to extract water from the unit. Is there a relationship or do we need to develop a new paradigm and risk assessment tool? We can all relate to 200 col/100 ml in water. When that level is reached in surface waters, it triggers a decision by decision maker. However, when it reaches 200 MPN/g dry soil, what does it provide the decision maker?
- The steering committee needs to put funding toward educational proposals relating to the education of the practicing professionals (regulators, contractors, manufacturers, and also the public decision makers). Our biggest problem today is not so much the technology, but the attitude that the cheapest system is the best and the lack of education about the technology. We need to invest in educational proposals that will assist in changing attitudes and ways of doing business.
- If this industry is to advance and be acceptable, society needs to implement management protocols. Management, at this time, is the weak link to implementing the new technologies. Funding needs to be made available to develop new management paradigm.

- Is it better to distribute the effluent by gravity equally to each trench as some promote, or is it better to quickly clog a trench with overflow going to the next trench, or should all systems have pressure distribution with how many ft^2/orifice?
- What should be the indicator organism? Fecal, *E. coli*, coliphage, etc.
- What are the effects of medications on system performance? There appears to be some evidence that medication will affect system performance, though unscientific at this time.
- What is the risk to public health on seasonal saturation sites where the separation distance may be limited for a short period of the year?
- How much downsizing and how much soil credit can be allowed for aerobically treated effluent, and as importantly, what type of effluent distribution is needed; gravity distribution or pressure, without increasing the risk to public health and the environment?
- Over the past several years we have heard a lot about risk assessment and risk management. Table 5 lists risks associated with onsite systems. How is risk assessment incorporated into research proposals?
- In the text (Conference version) the authors make the following statements relative to the inherent risks posed by wastewater due to microbial and chemical constituents. Table 2 provides a listing of possible constituents of concern. I quote the following:

  "For wastewater treatment, one can state the ultimate goal as being design and operations so that (1) there is no infectious disease attributable to an onsite wastewater system and 2) there is no measurable change in an ecosystem attributable to wastewater systems inputs. Clearly, in a given setting a system that provides no treatment at all may present the highest risk while increasing levels of reliable treatment effectiveness yield reduced levels of risk. However, since risk management requires consideration of non-technical issues such as socioeconomic factors, the most advanced treatment system will generally not be the best overall risk management solution."

  I certainly understand this concept but I am not sure that it has been translated into research needs in Table 6. My major concern is at what level of risk to public health and the environment are we going to accept. That needs to be defined and research dollars need to be made available for this endeavor. My fear is that we as a society have and will become more lax due to the *fact* that we are hard pressed to find many health impacts attributable to onsite systems and the mentality of the cheaper the onsite system, the better.

Those are some thoughts about the White Paper. Again, I feel that it was well documented and is as complete as can be expected within the time frame allowed.

revised

Research Needs in Decentralized Wastewater Treatment and Management: Fate and Transport of Pathogens

Dean O. Cliver, Ph.D.

Department of Population Health and Reproduction

School of Veterinary Medicine

University of California, Davis

One Shields Avenue

Davis, CA 95616-8743

phone: 530-754-9120

fax: 530-752-5845

email: docliver@ucdavis.edu

Contents

Introduction

In the late summer of 1999, at least 781 persons who had attended the Washington County (NY) Fair became ill; 127 cases of *Escherichia coli* O157:H7 infection and 45 cases of *Campylobacter jejuni* infection were confirmed by culture (Department of Health, State of New York, 2000). Among 71 people who were hospitalized during the outbreak, 14 developed hemolytic uremic syndrome (HUS); two people (ages 3 and 79) died. Illnesses appeared to be associated with drinking beverages, on August 28 or 29, prepared with water from Well 6 on the fairgrounds; the contamination may have originated from a septic system serving a 4-H dormitory on the fairgrounds close to Well 6. A dye test of the 4-H dormitory septic system showed a hydraulic connection between the dormitory septic system and Well 6 at the time of the investigation; hydrologic conditions were not exactly as they had been at the time of the Fair. DNA fingerprints of *E. coli* O157:H7 from the septic tank, Well 6 and distribution pipes leading from Well 6 were indistinguishable from the DNA fingerprints of *E. coli* O157:H7 in many culture-confirmed patients. *Campylobacter* was not detected in the septic tank. Results of a telephone survey suggested that between 2,800 and 5,000 Washington County Fair attendees may have developed gastrointestinal illness.

Even though the majority of alternate wastewater treatment systems serve single families and have no opportunity to claim this many victims, it is clear that there is a threat to public health when systems are not functioning properly. This relates to the ability of alternate, on-site systems to contain (prevent dissemination of) pathogens in human waste. Unfortunately, the few compilations of outbreaks of groundwater-associated illness in the U.S., even when they implicate septic systems as the source of pathogens, rarely indicate whether the system was leaking, malfunctioning, or simply in soil that could not perform its part of the purification task (Craun, 1979; Yates, 1985). For example, Craun (1981), reviewing outbreaks of waterborne disease in the U.S. from 1971 to 1978, says, "Overflow or seepage of sewage, primarily from septic tanks or cesspools, was responsible for 41 percent of the outbreaks and 66 percent of the illnesses caused by contaminated, untreated groundwater. These percentages include outbreaks where contaminants traveled through limestone or fissured rock."

This White Paper will consider the risks associated with various human pathogens and recommend needed research to ensure that alternate wastewater treatment systems prevent the transmission of these pathogens by the water route. Risk assessment involving infectious agents presents special challenges (ILSI Risk Science Institute Pathogen Risk Assessment Working Group, 1996). The area of dose-response from exposure to pathogens is very different from models with toxicants. For example, the toxin of *Clostridium botulinum* (not waterborne) is said to be the most toxic substance known, yet it takes about twenty million molecules of the toxin to kill a mouse. Most waterborne pathogens can cause human infections, and possibly severe illness, with 10 to 1000 infectious units.

Micro scale considerations are:

- Pathogen containment (retention, destruction) by conventional septic-tank, soil-field on-

site wastewater treatment systems.
- Effects of using alternative on-site wastewater treatments on anti-pathogen effectiveness.
- How can the sources (human, domestic animal, feral animal) of pathogens in water be determined?
- How can valid samples be obtained from the vadose zone?
- How does biomat formation affect pathogen removal?

Macro scale considerations are:
- Public health impacts of using on-site wastewater treatment systems as alternatives to urban, centralized systems.
- Public health impacts when either type system is disrupted by catastrophes (e.g., flooding).
- Public health impacts of inadequate maintenance ─ factoring maintenance requirements into design and implementation of alternative systems.
- Public health of impact of solids from on-site treatment, compared to biosolids from urban systems.
- How can the fate and transport of pathogens be incorporated into overall watershed modeling, with emphasis on subsurface flows?
- What doses of various pathogens are required to cause disease in humans?

1. Pathogens of concern

Pathogens that may be shed in human waste (principally feces) belong to several groups of microorganisms (Feachem et al., 1983). We will not consider helminths (multi-celled parasites, such as *Ascaris lumbricoides*) because they are relatively rare in the U.S.; however, it should be recognized that these agents are prevalent in many parts of the world, and the microscopic eggs by which they are transmitted may occur in the feces of persons visiting here from other countries. On the other hand, helminth eggs are consistently larger than *Cryptosporidium* oocysts, so many measures that retain or remove oocysts should also be effective against these eggs.

1.1 *Bacteria*

Members of various genera in the family Enterobacteriaceae are of concern (Benenson, 1995). Most prominent are *Shigella* spp. (which cause bacillary dysentery), *Salmonella* spp. (which cause severe diarrhea, sometimes with long-term aftereffects), and various types of *E. coli*, including the O157:H7 type mentioned above. Other *E. coli* groups can cause diseases resembling bacillary dysentery and cholera, whereas *E. coli* O157:H7 and a few other types cause enterohemorrhagic disease (bright, red blood in the stool), HUS (a severe form of kidney failure that usually affects children under 5 years old), and thrombotic thrombocytopenic purpura (TTP, which most often affects the elderly). Typhoid fever, caused by *Salmonella typhi*, is now relatively rare in the U.S.; and cholera, caused by *Vibrio cholerae*, is foodborne rather than waterborne in the U.S. *Campylobacter jejuni*, mentioned in the preceding outbreak description, is less often transmitted from human to human than from animals to humans. These bacteria may persist in water and wastewater for extended periods, depending on temperature, but they are relatively unlikely to multiply in the environment. They do not form spores and are relatively sensitive to heat and disinfectants.

1.2 *Viruses*

Viruses transmitted by the fecal-oral route, and therefore occurring in wastewater, include the hepatitis A virus, the Norwalk-like viruses (also called small round structured viruses — SRSV), and probably the astroviruses and rotaviruses (Benenson, 1995; Feachem et al., 1983). The virus of hepatitis E, which is often transmitted via water in some of the poorer countries of the world, seems not to occur in the U.S. Hepatitis A is a debilitating disease with an incubation period averaging 28 days (range 15-50); severely affected persons may be incapacitated for weeks. The Norwalk-like viruses, astroviruses, and rotaviruses cause transient diarrhea with a brief incubation period (1 to a few days); rotaviruses are major causes of infant mortality in developing countries. All of the virus particles are much smaller than bacterial cells and incapable of multiplying in the environment. Outbreaks of human disease associated with on-site systems have included 418 cases of gastroenteritis from a 27-nm diameter virus (detected by immune electron microscopy) from a broken sewer line to a well at a resort camp in Colorado (Morens et al., 1979) and perhaps 900 cases of gastroenteritis due to Norwalk virus (diagnosed by antibody response) at an Arizona resort that had earth fractures connecting the wastewater leach field with the aquifer from which well water was drawn (Lawson et al., 1991).

1.3 *Parasites (protozoa)*

Parasites transmitted via water and wastewater in the U.S. are principally the single-celled protozoa, *Cryptosporidium parvum*, and *Giardia lamblia* (syn. *duodenalis* or *intestinalis*; Benenson, 1995). Elsewhere *Entamoeba histolytica* is an important cause of disease (Feachem et al., 1983). *Toxoplasma gondii* causes human illness in the U.S., but would occur in wastewater only if cat feces were flushed down a toilet. *Cyclospora cayetanensis* may be transmissible via water under special circumstances, but remains rare in the U.S., even after several food-associated outbreaks from imported produce. *Cryptosporidium* and *Giardia* cause severe but self-limiting diarrhea in people with normal immune systems. Because there is no treatment for *Cryptosporidium*, it causes lifelong and life-threatening infections in AIDS patients and perhaps others whose immunity is impaired. Because the transmission forms (cysts, oocysts) of these agents are larger than bacteria, they are more likely than other waterborne pathogens to be removed by filtration through soil; however, systems installed on coarse gravel or over fissured bedrock are as likely to transmit protozoa as the other pathogens.

1.4 *Indicators and risk assessment*

Because of the relative difficulty of monitoring for these agents (especially the viruses and protozoa) in water and wastewater, there is a continuing search for valid indicators — microbes (or anything else) whose presence in water denotes a high likelihood that pathogens are there, too (Clesceri et al., 1999). Indicators would be expected to help in making valid risk assessments in situations where there might be a significant probability of transmitting pathogens via wastewater receiving alternate treatments. Another approach that is sometimes used in risk assessment is to use nonpathogenic microbes as surrogates in evaluating the efficiency of processes. Harmless bacteria, such as nonpathogenic *E. coli*, may serve as general indicators of fecal contamination and as surrogates in evaluating antibacterial processes (Garcia and Bécares, 1997). The absence of *E. coli* from a water sample does not, however, strongly indicate that viruses and protozoa are also absent. Bacteriophages, often certain types that infect *E. coli*, have been studied as indicators of virus contamination of water for a long time (Calci et al., 1998; Havelaar, 1993; IAWPRC Study Group on Health Related Water Microbiology, 1991; Nestor, 1984; Rose et al., 1999) and may be of some value as surrogates for evaluation of antiviral processes, including purification of wastewater by transport in soil (Bales et al., 1989; Higgins et al., 1999; Jin et al., 1997; Jolis et al., 1999; Moore et al., 1975; Powelson and Gerba, 1994; Powelson et al., 1990; Thompson and Yates, 1999; Thompson et al., 1998; Yates et al., 1985). Phage surrogates may also yield artefacts ─ coliphages MS2 and φX-174 gave quite disparate results in trials of their removal in saturated sand columns, apparently due to differences in their isoelectric points (Jin et al., 1997). Nonhuman enteroviruses can also be used as surrogates (Scandura and Sobsey, 1997). To date, no indicators related to protozoal contamination of water have been identified; some alternate organisms, e.g. cyanobacteria, might be useful surrogates.

2. Potential fates of pathogens

Pathogens shed in feces must meet certain conditions if they are to be transmitted via wastewater and water. Mainly, they must remain infectious on the way from their source host to another, susceptible person.

2.1 *Multiplication*

Among the agents discussed, only bacteria have any potential to multiply between hosts. If appropriate nutrients are present and temperatures are in the range of, say, 10 to 45°C (roughly 50 to 110°F), numbers of pathogens may increase 10- to 100-fold; but such multiplication is usually limited by competition from other, better adapted organisms. Otherwise, bacteria, as well as viruses and protozoan parasites, can only persist for periods of time or lose their infectivity before another host is encountered (Feachem et al., 1983).

2.2 *Inactivation ─ killing by physical, chemical, or biological mechanisms*

In the wastewater and water environment, persistence of pathogens is generally favored by temperatures near, but just above, freezing. Protozoan cysts or oocysts are generally killed by freezing, whereas viruses are preserved. The only other physical means of killing pathogens is ultraviolet light, which may have application in some on-site treatment systems. Other than extremes of pH, which might cause problems in on-site systems, chemical inactivation is usually accomplished by strong oxidizing agents, such as hypochlorite and, less often, ozone or chlorine dioxide. These are established technologies and may be necessary if effluent is to be discharged to surfaces; disinfection will be discussed in greater detail in section 5.2 below. Biological inactivation mechanisms are generally slow and poorly characterized; aeration may promote attacks on pathogens by nonpathogenic bacteria; but in anaerobic environments, bacteria from human waste that might accomplish biodegradation of pathogens are apparently inefficient (Deng and Cliver, 1995; Stramer, 1984).

2.3 *Mechanical removal, retention*

Septic tanks and various filters are used to remove or retain pathogens in wastewater (Crites and Tchobanoglous, 1998). One study suggested a 74% removal of viruses from wastewater passing through a septic tank (Higgins et al., 1999). Pathogens associated with fecal or other solids may collect in the sludge layer of a properly functioning septic tank but may remain infectious for periods of time and subject to resuspension and passage out with the effluent. Virus clearance from septic tanks dosed experimentally with virus in suspension (Scandura and Sobsey, 1997) is apparently more predictable than when the virus is in infant feces flushed down a toilet into the septic tank (Stramer, 1984).

Retention of pathogens in the soil vadose zone is apparently an important mode of wastewater purification; whether these pathogens may also be mobilized later and present a threat is an important question. In a benchmark review, Yates and Yates (1988) assembled published data and proposed modeling approaches to predicting inactivation and transport of both bacteria and viruses in soil. Removal of MS2 coliphage from septic tank effluent percolating through unsaturated, medium sand was found to diminish with depth (Higgins et al., 1999). However, MS2 phage that was retained during unsaturated flow of water in loamy sand columns was found largely to have been inactivated by the time recovery from the soil was attempted, even at 4°C; the MS2 phage moved freely with water under saturated conditions in the same soil (Powelson et al., 1990). In an experimental field installation in Florida, ~99% of PRD1 phage was removed from septic tank effluent by percolation through 60 cm of fine sand; removal performance was degraded by heavy rainfall (Rose et al., 1999).

2.4 *Persistence, transmission*

Barring specific intervention, such as chemical or UV disinfection, it has been usual to assume that pathogens retained in the septic tank or the soil will persist and continue to represent a threat of infection. This assumption wants testing under a variety of relevant conditions. If the pathogens are to be transmitted via an on-site system, they must retain their infectivity during any detention period and then be dislodged under circumstances that enable them to reach a susceptible person and be ingested. Clearly, each transmission scenario presents several opportunities for transmission of the pathogen to be prevented. Poliovirus in the stools of vaccinated infants persisted well in the sludge layers of septic tanks (Stramer, 1984), whereas MS2 coliphage retained in unsaturated soil was perhaps 61% inactivated at 4°C (Powelson et al., 1990). Human enteroviruses are apparently inactivated by drying in soil (Yeager and O'Brien, 1979). Risk assessment entails estimation of the probabilities that these deterrents to transmission will fail under real-world conditions.

3. Classes of wastewater to be treated

Where no central, urban wastewater collection and treatment system exists, any water user must provide for treatment and disposal of the resulting wastewater; ideally, reuse should be attempted, rather than just disposal (Tchobanoglous et al., 1999a). Human excrement remains the source of almost all pathogens of concern, but the outcomes of wastewater treatment from the standpoint of pathogen containment will inevitably be influenced by variations in the population served and in waste strength and quality.

3.1 *Household systems*

Systems that treat household wastewater usually are expected to receive all of the used water from the household, but systems may be designed to treat only graywater or blackwater, with other provision being made for the rest of the waste. For example, if a composting toilet is used, the on-site system may have only graywater to treat. This is likely to diminish the potential pathogen load, but the load should not be assumed to be zero. Reduced-flow plumbing fixtures may significantly increase the strength of the wastewater; this is supposed to be helpful to on-site treatment, but it may not be. In general, however, any measure that reduces per capita daily water use is likely to prove beneficial.

Pathogens may well be shared among members of a family regardless of the wastewater treatment system in use. Unlike centralized, urban systems, an on-site household system will be receiving no pathogens most of the time, but relatively high levels of pathogens on occasion. It is important that the system be able to deal with these surge loads of pathogens, as well as hydraulic surge loads. Clustered residential systems may mitigate this surge effect, unless the entire community served is infected at essentially the same time.

3.2 *Restaurants, filling station restrooms, and other commercial systems*

Because such establishments serve a varied and transient clientele, they are likely to receive a greater variety of pathogens than a household system. Some of the agents may be exotic to the area, depending on how far the patrons have traveled. Additionally, food wastes or other materials present in wastewater may affect the ability of the on-site system to deal with pathogens.

4. Septic tanks vs other treatment facilities

For many years, the septic tank has been the norm against which other on-site treatment facilities have been compared. Every alternative has “best-case” and a “worst-case” performance expectation.

4.1 *Septic tanks*

The design of septic tanks continues to be refined. Innovative designs involving multiple chambers, changes in baffle placements, etc., are continually being tried. Even an upflow, anaerobic sludge bed reactor septic tank has been described (Kalogo and Verstraete, 1999). Unlike most alternatives, the basic septic tank does its job without resort to moving parts, which largely avoids sudden malfunctions; 74% removal of coliphage MS2 in transit through standard experimental septic tanks has been reported (Higgins et al., 1999). It is still the basic function of the septic tank to remove solids from the wastewater as efficiently as possible. To the degree that this is accomplished, it is essential that the accumulated solids be removed before they accumulate to an extent that interferes with function, either because detention of the wastewater is too brief or because accumulated solids are being carried out with the effluent. Many additives that are supposed to delay or obviate pumping have been developed and aggressively marketed, but timely pumping seems still to be essential to proper function. Where soil conditions are not appropriate to accept septic tank effluent and provide efficient final purification, the effluent must receive more technically sophisticated treatment before discharge.

4.2 *Aerobic systems*

Most often, systems used as alternatives to septic tanks or in series with them include an aeration step to encourage growth of bacteria that help control solids and perhaps even attack pathogens. Model studies with coliphages representing human pathogenic viruses have shown inactivation to occur at the air-water-solid interface (Thompson and Yates, 1999). Filtration or other means of removing solids, including the biomass generated as a result of aeration, may also be applied. These measures impose an energy cost and a need for maintenance. Abrupt malfunctions are possible, and performance may be bad if the apparatus is not properly maintained. Inasmuch as such systems may be discharging their effluents to surface waters, treatment failures may entail significant risks of pathogen discharge.

4.3 *Other innovative approaches*

Because most pathogens enter the wastewater in human feces, incinerating or composting toilets may limit the pathogen load to the system, so that only graywater needs to be treated. This does not guarantee total freedom from fecal material, in that smaller quantities of feces on clothing or bodies will still be washed into the waste stream and need to be dealt with.

Clustering systems is another innovative approach (Rubin and Otis, 1999). Household wastewater may be treated in individual septic tanks and then transported by gravity or via force mains to a central final treatment and disposal site. Many alternatives are possible — it would not be possible to mention them all here.

5. "Secondary treatment and disinfection" by alternative systems

5.1 *Filtration media vs sand in a mound or column*

In addition to aeration, filtration may be included as a means of enhancing treatment. Filtration is especially important in the control of *Cryptosporidium* oocysts and *Giardia* cysts (Bellamy et al., 1985); these have human, as well as animal, sources and are unlikely to be controlled by final disinfection (especially by chemicals). Sand is the default filter medium; because it is a natural substance, there are endless variations in texture, uniformity, and even chemical composition. Assuming that the properties of the perfect sand (for this purpose) could be agreed upon, it would probably prove to be in very limited supply. Various other filter media have been proposed and used, in efforts to achieve more predictable performance and more straightforward design parameters (Emerick et al., 1999a). Some require only periodic backwashing, as sand does, whereas others require that the medium be replaced after some interval of use. As bacterial communities establish themselves by adhesion to the medium (Bower et al., 1996), removal of viruses (Emerick et al., 1999a), bacteria, and protozoa (Bellamy et al., 1985) is likely to improve.

5.2 *Chemical or UV disinfection systems*

Disinfection is often required before treated wastewater can be discharged to surface water or land. Inevitably, small-scale systems are at a relative disadvantage from the standpoints of cost and maintenance requirements; much more has been done and written regarding disinfection of effluents from centralized systems, but some of this information is surely applicable to on-site systems (Gerba, 1999). The efficiency of final disinfection depends intimately on the efficiency of previous treatment of the wastewater — neither chemical disinfectants nor UV will act well against pathogens that are protected by suspended material (Gross, 1999; Sobsey, 1989). Disinfection of viruses by UV light has a fundamentally different basis than chemical disinfection, and there are significant differences among the effects of chemical agents, even though almost all the useful types are strong oxidizing agents (Thurman and Gerba, 1988). Assuming that a low-turbidity, low-demand effluent is consistently produced, the remaining task is to apply the disinfectant in a way that delivers the design dose to all of the effluent all of the time. This means that the supply of disinfectant chemical must be replenished without fail or that the UV source must be calibrated periodically to ensure that its emissions in the antimicrobial portion of the spectrum are adequate. Replenishment of disinfectant chemicals may be within the domain of the individual homeowner, but testing the UV source probably is not. Built-in meters and feedback loops offer some advantages; these, too, may require preventive maintenance and repair.

Chemical disinfection was studied in poor quality secondary urban wastewater effluent, which might have been equivalent to wastewater from some on-site systems (Tyrrell et al., 1995). Coliphages, as surrogates for viruses, were much less sensitive than bacteria to chlorine, but more sensitive than bacteria to ozone. In demand-free water, ozone was slightly more effective than chlorine dioxide against *Cryptosporidium parvum* oocysts, and both were much more effective than chlorine or monochloramine (Korich et al., 1990). The results indicated that *C. parvum* is 30 times more resistant to ozone and 14 times more resistant to chlorine dioxide than are *Giardia* cysts.

Tchobanoglous et al. (1999b), at a recent on-site wastewater treatment short course, summarized the results of several earlier studies of UV disinfection with special emphasis on effects of particulates (Emerick et al., 1999; Loge et al., 1996, 1999; Parker and Darby, 1995) but did not explicitly recommend UV for use in on-site applications. One approach to avoiding the problematic protection of viruses (represented in this study by coliphage MS2) from UV by particulate matter is to precede UV disinfection by microfiltration that eliminates the particles (Jolis et al., 1999); whether this approach is applicable to small-scale systems is not conjectured.

6. Disposal to soil (and to groundwater)

From the time that people learned to dig wells, great faith has been placed in the ability of soil to purify water that percolates through it. Deeper, drilled wells have been preferred in more recent years, among other reasons, because it has been recognized that the purification achieved in some soils is limited. At its best, this natural medium does a remarkable job of undoing human contamination of water and has long served to purify water for reuse.

6.1 *Wastewater purification in soil*

The selection of soil for wastewater treatment and disposal is critical to control of pathogens. Soil is seldom transported to construction sites for this purpose, except where mounds or similar installations are to be built, so the character of the soil in situ should largely determine where homes and other establishments that depend upon on-site wastewater treatment are built (Rubin and Otis, 1999). Much more effort is needed to characterize soils from the standpoint of pathogen removal in wastewater treatment, establishing which variables are truly pertinent to this function. Where soil is inappropriate for wastewater treatment and fails in its purification function, bacteria can be carried for long distances from septic systems in groundwater, as in the 1972 typhoid outbreak in Yakima, Washington (McGinnis and DeWalle, 1983). Alternatives to the percolation test for evaluation of soils have been developed and evaluated; in general these are not directly related to pathogen removal, although they may be highly correlated with this desired capability. All but the coarsest soils are probably capable of retaining *Cryptosporidium* and *Giardia*, but transport of bacteria and viruses depends on many factors, not all of which are necessarily included in present models (Yates and Yates, 1988). To some degree, removal of pathogens is a special case of remediation; soil is expected to remove many other contaminants from water, as well as infectious agents. Even saturated flow can be expected to accomplish some purification, but it is clear from column studies that the greatest removal of viruses occurs in the vadose zone (Bales et al., 1989; Powelson and Gerba, 1994; Powelson et al., 1990).

In some contexts, it appears that virus removal is negligible below the vadose zone. This calls for good understanding of mechanisms of virus removal, in order to specify a minimum depth of vadose zone required for adequate treatment. Even when this is possible, there is some lingering uncertainty in estimating the minimum depth of the vadose zone that occurs during periods of high ground water. Further, there seems to be some lingering uncertainty as to whether the vadose zone performs identically under aerobic and anaerobic (no-free-oxygen) conditions. Useful data regarding transport of viruses, etc., are becoming available; but it often remains undetermined whether the viruses are retained in potentially infectious condition or are being inactivated, whereby later dislodgement represents no threat to human health. Net retention of virus from operating septic systems in the field has been monitored by testing ground water (Rose et al., 1999; Scandura and Sobsey, 1997) or by sampling water in the vadose zone with ceramic suction lysimeters (Oakley et al., 1999), which have also been used in the vadose zone of a groundwater recharge project (Wilson et al., 1995). In pilot-scale studies, effluent from unsaturated flow can be collected beneath the units (Van Cuyk et al., 1999) or in pans placed at various depths in the vadose zone (Higgins et al., 1999). Epi-aquifers (seasonal perched groundwater) might also be used in this way.

Prediction of virus transport, especially, has been of great interest (Yates and Ouyang, 1992). A model applied to data from several sites showed considerable variation in goodness of fit (Yates, 1995), presumably because not all of the critical variables were included in the model. One site in particular (Lake Redstone) was a conventional septic system that was newly placed in service; increases in performance of sand lysimeters over the first few months of use has been demonstrated (Van Cuyk et al., 1999). Although some interaction of viruses with the soil medium may be based on electrostatic charges, adsorption of viruses also appears to have a hydrophobic component (Bales et al., 1991); hydrophobic surfaces also play a role in virus inactivation at the air-water-soil interface in batch sorption experiments (Thompson et al., 1998). Virus that escapes to groundwater may remain infectious for days to months (Yates et al., 1985; Yeager and O'Brien, 1979). Ability to predict where this contaminated groundwater will go needs considerable improvement.

6.2 Distribution systems

The effectiveness of soil as a treatment medium also depends on how the wastewater is applied. The existence of an adequate vadose zone is moot if the wastewater enters the soil in a concentrated plume that displaces the air along its path. This may occur early in the life of a gravity distribution system, in that the wastewater arrives at a small portion of the constructed infiltrative surface and enters, rather than being widely distributed over the surface of the bed. Pressure distribution systems are designed to attain uniform distribution from start-up, but conventional, gravity systems may eventually perform well after some period in service. The length of this maturation period seems not to have been established (Van Cuyk et al., 1999). As an alternative to pumping chambers and pressure distribution systems, “engineered” media or systems have been developed for applying wastewater to the infiltrative surface. Some of these appear promising and offer the advantage of no moving parts.

6.3 *Infiltrative surface*

Events at the infiltrative surface, where the wastewater first enters the soil as a treatment medium, are probably crucial to the treatment outcome. The spectrum from completely open pores, to partial clogging and gradual infiltration, to total closure and rejection of the wastewater may be encountered over time in a single system (Ronner and Lee, 1998). The term borrowed from German, *Schmutzdecke*, is well established (Bellamy et al., 1985) but generic; calling the clog zone "biomat" is probably more descriptive. Further, clogging of the infiltrative surface that causes wastewater backup into the household or ponding above the bed is not inevitable, assuming the absorption system is sized appropriately and the septic tank is pumped at proper intervals. The microbiology and physics of the infiltrative surface are critical to pathogen containment in on-site wastewater treatment in soil, not only to avoid catastrophic failure, but to ensure that the wastewater enters the across the broad surface of the field and at a rate that does not lead to saturation of any portion.

7. Disposal to surfaces (land or water)

To enable construction at sites whose soil is inappropriate, or development at densities that are too great to permit on-site treatment, alternatives to the basic septic-tank, soil-absorption approach are required. Means of treating wastewater for surface discharge have been reviewed in sections 4 and 5. This section will consider the potential impacts of the surface discharges that take place after the application of the alternate wastewater treatments. Impounded surface water (ponds or lagoons) may serve as treatment facilities for wastewater before discharge to "natural" surface waters (Crites and Tchobanoglous, 1998; Garcia and Bécares, 1997).

7.1 *Soil surfaces*

Some soil surfaces will rapidly accept wastewater that is applied to them, and they may maintain this high permeability indefinitely. In that the infiltrated water may rapidly arrive at an aquifer, and then in someone's well, it is probably important to ensure that treatment and disinfection are complete before the effluent is discharged to such surfaces (Schaub and Sorber, 1977). Overland flow, in which the effluent travels some distance across the soil surface before complete infiltration, probably also affords limited purification. Application of wastewater or reclaimed water to a soil surface as a means of groundwater recharge requires careful planning (Powelson and Gerba, 1994; Powelson et al., 1993). Surface infiltration for effluent disposal or water recycling is, of course, seriously limited in areas where the soil is frozen during a significant portion of the winter. Although pathogens such as protozoan cysts and oocysts are killed by freezing, other pathogens may be preserved on the frozen surface, awaiting the spring thaw and run-off into a surface waterway, perhaps assisted by spring rains. Maintenance of surface cover vegetation is probably important in stabilizing the soil surface; the effect of surface cover on the fate of pathogens is generally unknown. Again, it is probably best that treatment and disinfection of the wastewater be complete before application to most soil surfaces.

7.2 *What is an appropriate receiving waterway for on-site treated wastewater?*

Although discharge to surface waters is very common with treated urban effluents, this practice is generally discouraged with wastewater treated on-site. The perception that discharge to surface waters from on-site units is a threat is probably based on occurrences where surface water was contaminated from on-site systems that were not designed for surface discharge of their effluents (Griffin et al., 1999; Paul et al., 1995). There is surely no reason why wastewater cannot be purified sufficiently in alternate treatment systems so that pathogen-free effluent is consistently produced. The challenge is to develop this capability, including high reliability, at prices that individual homeowners can afford. The use of holding tanks as “zero-discharge” alternatives is probably not realistic in the long run.

8. Other aspects

8.1 *Water re-use*

Though seldom represented as such, on-site wastewater treatment systems are important components of our water re-cycling complex (Tchobanoglous et al., 1999a). Used water, processed by alternate methods, will inevitably be presented for use by someone else in any situation other than discharge to the sea; and even in the sea, recreational use and seafood production may be impacted by effluents (Griffin et al., 1999; Paul et al., 1995). This is not an argument against on-site or alternative wastewater treatment, but rather a realistic acceptance of the necessity for doing the treatment job right, all of the time. Outbreaks as large as the one described at the beginning of this paper are rare, but this does not mean that the potential impact of on-site wastewater treatment on public health is not large. On a person-for-person basis, the incidence of pathogens is probably about the same among people whose waste is processed on-site as those connected to urban systems. An important difference is that individual home systems are not operated around-the-clock by trained engineers, as many urban systems are (Jolis et al., 1999). This places the onus on designers, manufacturers, and installers of on-site systems to make them as fail-safe as possible. The use of water to carry human waste is firmly entrenched in our society and has led to great strides in hygiene and public health; it is our responsibility to close the gaps in the water cycle so that pathogens in the waste are not transmitted to other people, or to animals.

8.2 *Treatment and disposal of solids*

Septage, used here to mean the suspension of solids that is pumped from a septic tank, is a by-product of wastewater treatment that requires safe disposal, if not treatment. To the extent that human pathogens are associated with fecal solids or may adsorb to other sedimentable materials, the pathogens are likely to be retained (at least temporarily) in the sludge layer of the septic tank. Means of disinfecting septage while still on the pumper’s truck have been investigated (Stramer and Cliver, 1984), but nothing of the sort seems yet to be in use. Freezing and thawing of urban wastewater sludge offers some disinfection, particularly of parasites, in cold-weather areas (Sanin et al., 1994) and could probably be applied to septage as well. Other solids accumulations from wastewater treatment, such as solids collected on filters, may also harbor pathogens. Heat, which may be unduly costly, is the only rapid method of disinfecting septage and other such solids. As alternatives to surface spreading, septage may be composted or delivered to urban wastewater treatment plants for treatment, either with the influent or the plant's own biosolids. These options often represent a major cost to the owner, in the spectrum of operating costs, so it is important that public health concerns be addressed adequately, but without anxiety-induced overreaction that escalates costs to no useful purpose.

9. Risk assessment components

Assessment of risks to public health from on-site wastewater treatment begins by determining the quantities of pathogens produced and shed by the population served. At present, there is no reason to believe that the prevalence of pathogen shedding is any different among people served by on-site systems than those served by centralized, urban systems. Urban systems have the advantage of relative homogeneity of influent, in that people who are infected and shedding pathogens will almost always comprise a minority of the population served. On-site systems, on the other hand, have the advantage that they probably do not receive pathogens with their influent most of the time. However, urban systems probably do not retain pathogens for very long, except perhaps in their biosolids (sludge); whereas on-site systems may have pathogens in septic tank sludge, at the infiltrative surface of the soil field, and in the soil matrix, which could potentially be dislodged and impact people via water. The relative risks of on-site and urban wastewater treatment can be compared with each other, including all events from, say, when the toilet is flushed until the effluent rejoins the environment as water that may be used again for some purpose (Jones, this symposium). Additionally, the risks of disease transmission via wastewater that is treated in one way or another need to be compared with those of transmitting these same diseases by direct human contact within families and via other human associations, as in schools, workplaces, etc. Arbitrary standards of acceptable risk, as applied in other public domains, may also be evoked here. Wastewater treatment is a highly specialized task, but it needs to be considered in the context of public health and disease prevention, which offers a number of other alternatives. Risk assessment would be greatly facilitated if valid indicators of probable virus or protozoan presence in water could be identified. Various bacteriophages appear to have some application as indicators of virus contamination, but even these are probably more predictably present in urban than on-site wastewaters. On the other hand, bacteriophages, and perhaps cysts or oocysts of protozoa that are not infectious for humans, might be useful as surrogates for pathogens in evaluating treatment strategies on a pilot scale or in existing installations.

The train of events to be considered in a *micro-scale risk assessment* would include:

- likelihood of any person (in the population under consideration) having an infection from which agent is shed in excreta, and likelihood that other pathogens from pet wastes, etc., are flushed down the toilet;
- likelihood that the pathogens in the wastewater are retained in the septic tank or other

primary treatment apparatus, rather than passing through;

- degree to which pathogens in the primary effluent are retained or inactivated during subsequent treatment (e.g., in soil or in an aeration, filtration, or disinfection step) before discharge to the surface or to groundwater (Powelson et al., 1990; Yeager and O'Brien, 1979);
- proportion of pathogens, which were retained in treatment, that are still infectious when the solids or other treatment by-product are discharged to the environment;
- probability that these pathogens will contaminate water or food (vehicles) that someone else may ingest;
- probability that the pathogens will not be inactivated by disinfection, cooking, etc., before the vehicle is ingested;
- probability that the person ingesting the pathogen will be infected (colonized) by it (Haas, 1983);
- probability that the infected person is susceptible enough to become ill as a result of the infection (Naylor, 1983);
- probability that the infected person transmits the pathogen to others, directly or by way of another alternate wastewater treatment system.

Each of these contingencies has some range of probability values; however, this range may not be known with certainty. For example, the proportions of the U.S. population who are especially susceptible to disease by reason of special circumstances (e.g., age, pregnancy, immune suppression) is known (Council for Agricultural Science and Technology, 1994), but the proportion of these in a given study population may not be known, and the effects of these special circumstances may differ, depending on the pathogen under consideration. It is also reasonable to suppose that even a conventional septic system changes more over time than a centralized, urban system — as the septic tank collects sludge, the detention time of the wastewater is reduced; as the soil field matures, its purification performance improves. Great gaps, many of which could be filled by further research, exist in the knowledge needed to perform such risk assessments accurately.

Macro scale considerations include environmental events since the septic system was permitted and installed. For example, adjacent dwellings may contribute to an excessive load in groundwater, etc.; or other events may lead to a substantial change in the depth of groundwater below grade. With the EPA undertaking to manage Total Maximum Daily Load of nutrients for entire watersheds at once, one can only guess what the impact of future regulations will be on the operation of alternate waste treatment systems. Optimal "nutrient management" may not result in the best pathogen containment — each agency has its own perception of public health and benefits.

10. Risk management applications

The development, installation, regulation, and maintenance of alternate wastewater treatment facilities (Rubin and Otis, 1999) are all exercises in risk management. Accurate risk assessment is needed to provide the basis for a rational approach. It is also important that risks associated with the central-facility, urban treatment option be accurately assessed, so that alternatives can be compared on valid bases. Macro-scale considerations must be addressed, such as how to control selected pathogens over large segments of the population and how to limit the pathogen loads in entire watersheds or other ecosystems. Programs to limit contamination on a macro scale will require means to identify sources of viruses (Jothikumar et al., 1998), bacteria (Hagedorn, 1999), and protozoa (Deng and Cliver, 1998, 1999) detected in the watershed. It is important that these issues not be addressed only via a business-as-usual assumption; catastrophic events, such as earthquakes and floods, will require very different remedies in restoring centralized and decentralized systems to their functional states. Progress on these public policy fronts will require the participation of persons who can take a broad view of the issues, over narrow perceptions and self-interest. One hopes that such persons will be found among industry, academia, government regulators, and home owners.

11. Overview and conclusions

This White Paper has broadly reviewed what is known regarding the incidence and transmission of pathogens via alternate or on-site wastewater treatment. I has been shown that there are hazards in on-site treatment, though these are not at present able to be compared directly with hazards accompanying centralized wastewater treatment. Among other differences, standards for discharges from urban plants have been somewhat relaxed on the basis that effluents would be subjected to sophisticated treatments before becoming drinking water. On-site systems, on the other hand, are perceived as discharging to groundwater that will be abstracted via someone else's well and ingested without further treatment or disinfection. Subsurface treatment and discharge are no longer regarded as out-of-sight – out-of-mind.

Alternate wastewater treatment has endured years of benign neglect, in which funding for research on crucial questions was severely limited. It seems now to be recognized that on-site treatment is here to stay and is serving a fairly constant proportion (and thus, increasing numbers) of the U.S. population. Even the relatively unsophisticated septic system includes many unknowns, including pathogen retention and survival in the sludge layer, and effects of biomat formation and soil maturation on the effectiveness of subsurface treatment of septic tank effluents. Loading rates, surge loads, and many other factors that are harder to control in on-site systems than in urban systems, have still to be evaluated. Where a septic system is inappropriate or inadequate, much more sophisticated alternate technologies are required. There is no lack of innovation in this area, but risk assessment in such instances calls for consideration not only of what the alternate system was developed to do about pathogen containment, but how well it will function over years of use at hundreds or thousands of sites. And, given the growing shortage of water in the U.S., it is important to view every treatment technology as a means of preparing water for reuse. New organizational and regulatory approaches will be required to ensure that the right choice is made among alternate treatment systems and that systems are maintained and operated in such fashion that they continually achieve what they were designed to do. Performance standards for centralized wastewater systems are relatively easy to write and monitor, though not always easy to attain. Performance standards for on-site, decentralized systems will be very difficult to establish and even more difficult to enforce. It must also be recognized that pathogen control is a key feature of on-site treatment, but by no means the only item on the agenda. Laboratory and field research can supply more of the pieces that are missing from the puzzle, but others will require application of risk assessment (both absolute and comparative with alternate approaches) and inspired regulatory rule making. We hope that this conference affords another large step in that direction.

12. Research priorities, as seen after the conference

The conservative assumption is that the basic septic tank-soil absorption system is still the benchmark facility for on-site wastewater treatment. The new perception is that this is a mechanism of water recycling, and that the risk of returning used water to the aquifer (for eventual use by someone else) is what needs to be assessed. Furthermore, acceptable risk levels, rather than zero risk, need to be targeted with due awareness of attendant costs and benefits. Differences in orientation between scientists, regulators, industry, and the client public are real and must be recognized.

Risk assessment for on-site wastewater treatment by a septic tank-soil absorption system subsumes the likelihood that pathogens will be present in the wastewater and not be removed or inactivated by the various treatment steps before reaching someone's well. Further, the water from the well must be ingested by a susceptible person for illness to result; the distinction between what is called *drinking water* and what water (if any) people actually drink is too often ignored. It also should not be forgotten that solids retained in the septic tank are supposed to be removed periodically, and these, too, may contain pathogens.

The "A List," then, aims for a specific, quantitative risk assessment algorithm that can be applied to conventional, septic tank-soil absorption wastewater treatment systems. Some of the elements of this risk assessment pertain to the technology, and others are based on the human clientele. They are listed here in the order of events in wastewater treatment.

1. Incidence of infections, whose agents occur in wastewater, among the population served (human factor)
2. Delivery of the selected waste stream to primary treatment in the septic tank (technology)
3. Retention or inactivation of pathogens in the septic tank, as function of its design (technology) and maintenance (human factor)
4. Distribution and infiltration of pathogens into the soil treatment medium, as a function of the condition of the infiltrative surface and how the wastewater is applied (technology)
5. Retention or inactivation of pathogens in the vadose zone of the soil treatment medium (technology)
6. Transport and possible inactivation of pathogens in the aquifer (technology)
7. Abstraction and possible inactivation of pathogens from the aquifer via wells, springs, etc. (technology)

8. Ingestion of pathogens, with the abstracted groundwater, by susceptible people (human factor)
9. Transmission of pathogens via septage (technology, human factor)

Not enough is known about any of these elements to mount a valid risk assessment in the context of a specific on-site installation at this time. Those marked "human factor" are in the domain of social or medical scientists and are not topics of environmental research as such. The incidence of pathogens (1) and susceptibility to infection (8) are medical concerns; whereas maintenance of septic tanks (3), whether people drink water from their wells (8), and how septage is handled (9) are social or behavioral matters. Perhaps these last pertain to risk management, rather than risk assessment.

The remaining elements, first and foremost, involve soil — getting the pathogens and wastewater into the soil treatment medium, events in the vadose and saturated zones, and abstraction of groundwater for human use. **In all probability, the two most crucial of these are getting the wastewater into the soil (4) and treating it under unsaturated conditions (5).** The keys to eventual success are: soil classification criteria that apply directly to pathogen removal or inactivation, loading rates and depth requirements that derive from these classifications, and distribution-infiltration systems that ensure that the soil medium is functioning optimally in treating the wastewater. Validation of surrogates that can substitute for pathogens in field studies and give useful results is crucial.

The "B List" deals largely with comparisons (to the risks assessed via the work proposed above) of risks involved in the use of alternative on-site wastewater treatment systems. All of these are technological in nature and are listed here in the order of treatment events to which they pertain. These alternatives become attractive where the basic septic tank-soil absorption system is not appropriate.

10. Segregated waste systems that permit treating graywater separately, perhaps composting or incinerating human excrement rather than flushing it
11. Secondary or advanced on-site treatment and disinfection, often to permit surface discharge of the effluent
12. More efficient means of applying the waste to soil for treatment, modification of the soil to make it a more effective treatment medium, or substitution of other treatment media for soil
13. Cluster systems as means of serving larger numbers of people by scaled-up, on-site technology

How the high-priority topics should be studied raises further questions. Laboratory research can address one or a few variables at a time and provide fairly concise answers, but application of the findings in the field may yield unpleasant surprises. Field research subsumes many more variables, some of which may be totally out of control and random; but field work is absolutely essential to produce data that can be applied in valid risk assessments.

Where soil systems are concerned, it is important that characterization go beyond just classification, to integrate weather and other factors into performance evaluations. Clearly, this calls for interdisciplinary studies. It also calls for understanding of significant regional differences, at the same time that even-handed, national regulations are sought.

Because pathogens may not be spontaneously present in operating on-site systems — and field studies in which pathogens are introduced are generally illegal or unethical — it is important that surrogates be validated that can be used instead of pathogens and produce results that can be applied with confidence in risk assessment. No one surrogate agent can represent both bacteria and viruses; to the extent that protozoa may also be a concern, the challenge is greater still. Standardization of research methods would also be very helpful, but hard to achieve.

A good deal of needed information is probably in the literature but needs an intense search and good organization to enhance its usefulness. Beyond that, innovative means might be sought of “mining” results of research that has been performed and recorded, but not published in journals. These could be in reports of research done or funded by all levels of government, as well as unpublished theses and other academic documents. Most of these documents will not have been peer-reviewed, but current methods of “meta-analysis” will probably surmount some of the shortcomings of the reports.

13. Acknowledgments

It was a great pleasure to be asked to do this White Paper, which led me to revisit many happy years of collaboration in the Small Scale Waste Management Project of the University of Wisconsin-Madison, and some very stimulating interactions with Jim Kreissl of EPA. I thank Valerie Nelson of the Coalition for Alternative Wastewater Treatment and Tom Yeager of Kennedy/Jenks for pointing me in the right direction and Loret Ruppe of the UCD Department of Civil and Environmental Engineering and Stuart Oakley of California State University, Chico, for helping me get started. I am grateful to Chuck Gerba of the University of Arizona and Marylynn Yates of the University of California, Riverside, for serving as official reviewers and to the conference attendees who provided guidance. I thank Andrea Arenovski, the Project Coordinator, for helping me with finishing touches. Above all, I thank Leticia Chacon Rodriguez, who worked with me through all of this — especially the hardest parts.

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Appendix: Tabulations of research needs

Table 1. Research needs regarding pathogens in alternate wastewater treatment:

Research Topics	Strategic Focus	Tasks	Products	Uses
Septic systems (septic tank plus soil absorption field)	What are the basic mechanisms by which pathogens are contained or inactivated by basic septic systems?	Develop experimental methods to predict the fate of pathogens under a variety of conditions of use of septic systems	Data on the fates of various pathogens at each stage of wastewater treatment in septic systems, under a variety of operating conditions	To determine the conditions under which septic systems can be used, and with what density, without threatening public health
Innovative on-site wastewater treatment systems	What are the basic mechanisms by which pathogens are contained or inactivated by such systems?	Develop experimental methods to predict the fate of pathogens (surrogates) under a variety of conditions of use of innovative systems	Data on the fates of various pathogens at each stage of wastewater treatment in innovative systems, under a variety of operating conditions	To determine the conditions under which innovative systems can be used, and with what density, without threatening public health
Cluster systems for alternative wastewater treatment	How do cluster systems meet the requirements of pathogen containment or inactivation?	Develop predictive methods to determine the anti-pathogen effectiveness of cluster systems, based on established effects of unit processes	Data on how cluster systems will meet the pathogen containment or inactivation goals required to prevent disease transmission	To determine where the use of cluster systems is appropriate in serving wastewater treatment needs of small communities without endangering public health
Septage and other solids from on-site treatment	How can solids generated in on-site wastewater treatment be removed without threatening public health?	Conduct trials of disinfection or disposal of septage and other solids, from the standpoint of pathogen control	Data on the content of pathogens or surrogates in septage and other solids, as removed from the on-site system and after treatment and disposal by various methods	To guide regulation of solids disposal and integrate solids generation into operational plans for safe on-site wastewater

Table 2. Pathogens and septic systems (septic tank plus soil absorption field):

Subtopic	Strategic Focus	Tasks	Products	Uses
Septic tank	Removal of pathogens (bacteria, viruses, protozoa) from wastewater by retention of solids in septic tanks	Validate surrogates, especially for viruses and protozoa; determine retention in prototype and in-service units	Knowledge of the basic ability of septic tanks to retain pathogens shed in human waste	To evaluate wastewater treatment in the septic tank as a unit process for pathogen control
	Effects of sludge accumulation and surge loads on pathogen retention; inactivation of pathogens retained in sludge	Follow history of sludge retained in septic tanks; determine resuspension of pathogens in solids by hydraulic surges and by out-gassing	Knowledge of the stability of retained sludge as it affects long-term containment of pathogens, as well as the stability of pathogens in sludge	To determine the effectiveness of pathogen control by septic tanks in longer-term use, under real-world conditions
Soil field	Effects of clog zone (biomat) development on uniformity of wastewater infiltration to the soil medium, for pathogen retention in the soil; alternative, engineered infiltrative surfaces	Determine the development of a clog zone, as a function of time and loading, and how this affects entry of pathogen-containing septic tank effluent into the soil medium for treatment; evaluate alternatives	Knowledge of how a new system matures to reach its full pathogen-control capacity; reliance on natural biomat formation as an alternative to using engineered infiltrative surfaces to ensure success	To evaluate the pathogen retention of new, conventional septic systems and determine whether alternatives are needed to optimize wastewater entry into the soil medium

	Effectiveness of pathogen retention during wastewater passage through the vadose zone	Determine pathogen retention and inactivation in the vadose zone, as functions of depth of the zone, loading rate, soil texture, etc.; determine changes in soil performance as a function of length of service, and how these can be predicted during the initial soil assessment	Knowledge of how various classes of pathogens are retained by wastewater treatment in unsaturated soil systems; information about changes in soil treatment as a function of maturation of the absorption field	To enable more accurate decisions as to siting and sizing of soil fields, with a view to long-term effectiveness in containing pathogens
	Pathogen survival and transport in saturated soil	Extend studies of the fate of pathogens that reach groundwater	Data on pathogen travel with groundwater, including groundwater that rises into a formerly unsaturated zone in which pathogens had been retained	To enable sound decisions about well locations relative to soil fields, based on hydrogeologic measurements; to provide bases for managing groundwater that may compromise pathogen retention in soil fields
Catastrophic conditions	Pathogen mobilization in the event (e.g.) of flood or earthquake; restoration of normal operation after such events	Conduct model studies in which septic systems are inundated or seismically disrupted, determining the fate of retained pathogens and how to restore normal pathogen containment	Data on the possible public health effects of natural disasters that impact septic systems	To establish procedures for protecting public health in the event of natural disaster; to provide valid bases for comparison of on-site systems with centralized systems in case of disaster-induced failures, including what is required to restore normal operations

Table 3. Pathogens and innovative on-site wastewater treatment systems:

Subtopic	Strategic Focus	Tasks	Products	Uses
Wastewater to be treated — raw wastewater or septic tank effluent?	How is pathogen containment or inactivation by innovative systems influenced by pretreatment of wastewater in a septic tank?	Apply aeration, filtration, and disinfection methods to wastewater with and without prior septic tank treatment, determining how pathogens (surrogates) are affected by each unit process	Data on the anti-pathogen effectiveness of unit processes under various conditions of wastewater pretreatment	To decide how these unit processes can best serve to prevent transmission of various pathogens via on-site treatment systems
Failure mode analysis	How might pathogen discharges result from abrupt failures of sophisticated treatment system?	Run unit processes to failure, either through lack of maintenance or accelerated deterioration, with pathogens or surrogates in the wastewater treatment train; test treated wastewater before disinfection, to determine the pathogen discharges that would result of disinfection should fail	Data on "worst-case" failure of units that require maintenance and service to operate optimally, from the standpoint of pathogen release	To decide which innovative systems are best suited to control pathogens under suboptimal conditions of operation and maintenance
Catastrophic conditions	How might pathogen control in innovative systems be affected by flood or earthquake; restoration of normal operation after such events	Operate treatment units under normal conditions, loading each with pathogens or surrogates, then impose inundation or simulated seismic disruption to record the possibility of pathogen discharge and what is needed to restore normal function	Data on pathogen release from failures due to exogenous events	To plan restoration measures that will be required in the event of natural disaster and to select robust treatment apparatus that will minimize pathogen escape during such events

Table 4. Pathogens and cluster systems for alternative wastewater treatment:

Subtopic	Strategic Focus	Tasks	Products	Uses
Clustered disposal	Assuming that individual (household) systems are used to treat the wastewater, what special pathogen problems are associated with assembling the wastewater for disposal at a common site?	Model the anti-pathogen effectiveness of various individual treatment systems that may contribute to clustered disposal; model the various transport systems that may be used to collect the wastewater; do field studies of scaled-up soil infiltration treatments	Predictions of wastewater quality from a pathogen standpoint; predictions of pathogen loads arriving at the disposal site, based on the dynamics of the population served	To aid administrative rule-making regarding the safe use of cluster systems in congested areas and other situations where they are contemplated; siting and design criteria
Clustered treatment	Where the primary treatment facility serves several individual households or other establishments, what particular problems may arise regarding pathogen control?	Model wastewater blends and surge loads from multiple households, from the standpoint of pathogens in larger-than-usual on-site primary treatment systems	Predictions of pathogen control in scaled-up primary on-site treatment facilities	To provide bases for design, maintenance, and permitting of cluster treatment systems that will protect public health
Catastrophic conditions	Where small communities are served by cluster systems, how will natural disasters such as floods or earthquakes affect pathogen control, and how will such systems be restored after catastrophic disruption?	Model a variety of cluster systems and impose selected, plausible catastrophes on them, to determine the probable effects on pathogen control and the measures needed to restore pathogen containment after the event	Predictions (hopefully, accurate) of the potential public health impacts of catastrophes impinging cluster wastewater treatment systems	To guide the choice of a cluster system that will present the least pathogen-discharge threat in the event of foreseeable natural disasters and to plan for damage control and remediation in such events

Table 5. Pathogens in septage and other solids from on-site treatment:

Subtopic	Strategic Focus	Tasks	Products	Uses
Septage	Given that septic tanks retain solids that may be rich in pathogens (depending on the history of the household served), what are appropriate means of disposal that are economically feasible and protect public health?	Model various septage disposal or disinfection options, based on facilities available in specific areas (e.g., land spreading is not an option in a congested area); estimate the adequacy of each from the standpoint of pathogen control	Area-specific predictions of pathogen discharges that may result from various methods of septage disposal	To make regulatory decisions as to which septage disposal procedures adequately protect public health, while offering system owners economically feasible options
Solids from innovative treatment systems	What pathogen loads are likely in solids generated by aerated treatment systems or in materials backwashed from filters?	Model innovative systems, using surrogates for viruses and protozoa in wastewater, to determine what pathogen levels can be expected in solids produce in treatment	Predictive data on pathogen collection and inactivation in solids produced by innovative wastewater treatment systems	To determine whether these classes of solids can be disposed of within the regulatory framework devised for septage, or whether special rules (more or less stringent) are appropriate

Peer Reviews

The preceding White Paper, *Research Needs in Decentralized Wastewater Treatment and Management: Fate and Transport of Pathogens*, by D.O. Cliver was solicited for peer review. Reviewers comments are provided in this section.

C. P. Gerba
University of Arizona

This document is an excellent overview of the general areas and topics needed to better understand the fate and transport of enteric pathogens from onsite treatment systems. I think the key points to be considered are right on target. However, because research funds are always limited I think it is important that key research needs be prioritized. It has to be recognized that pathogen groundwater contamination by onsite systems will always occur. The goal of research in this area should be to minimize this impact. Pathogen contamination of groundwater is controlled by three factors:

1. The ability of the onsite system to remove pathogens

2. Reductions that occur in the vadose zone

3. Reductions that occur during lateral movement of the ground water

The first can be improved by better design of onsite systems and assessing how pathogens are removed by onsite systems of different design. Removal in the vadose zone is control by specific site characteristics. Removal of pathogens is expected to be greater in the vadose zone than the saturated zone of the subsurface. Ideally given enough depth all of the pathogens could be removed in the vadose zone. However, in most regions of the United States there is not great enough depth available to have a significant impact on pathogen reduction. Finally, the vertical distance of the point where the pathogens enter the groundwater and where the water is abstracted is another barrier that can be used to reduce pathogens.

To evaluate the reduction of pathogens by onsite treatment systems of different design standardized testing protocols are needed. How the tests are to be conducted and the length of study need to be defined. To reduce the cost of such testing surrogate test organisms should be developed to reduce the cost of such testing. To this end a group of surrogates such as coliphages, *Clostridium perfringens,* etc. should be compared with pathogen removal in pilot studies. Pathogens to be studied should include an enterovirus and an adenovirus. An enterovirus should be selected because of the ease of working with these viruses and the large data base available on removal by treatment systems. Adenovirus appear to be the longest surviving virus in the environment are very resistant to some disinfectant processes such as ultraviolet light and chloramines. Acceptable performance criteria for removal also need to be defined. To have an impact on virus contamination of groundwater onsite systems must reduce virus concentrations by at least 99% or more.

The vadose zone appears to be an important area in limiting the transport of pathogens. While in many or most cases it may not be possible increase the depth of this zone a better understanding of pathogen transport through this zone will aid in better design and criteria for onsite systems. Recent research by Jin et al., 1999 suggest that metal oxides on the soil surfaces and ionic strength play a major role in virus transport under unsaturated conditions. These factors need to be better understood and defined. The objectives of this work should focus on the impact of soil type, degree of soil saturation, virus or pathogen type, impact of rainfall, and depth of the vadose zone. The goal should be to define the degree of removal that can be expected by the vadose zone under a defined set of conditions.

In a nationwide wide study on the occurrence of viruses in groundwater (Abbaszadegan, 1999). It was found that 80% of the time a virus was detected in groundwater when a potential source was within 500 feet. This suggests that without knowing site characteristics at least 500 feet separation distance is necessary to ensure some reliable safety vertical distance between onsite systems and points of groundwater abstraction. Again, site characteristics play a major role in determining how far a pathogen may migrate from an onsite system. To aid the regulatory community and better define transport of pathogens under any given field condition the development of easy to use tracers for pathogens is probably needed. This may be accomplished by the development of a manual for tracer studies. The use of surrogates such as bacteriophages or the use of molecular fingerprinting techniques for source tracking could be described.

Finally, I believe a complete literature review of pathogens and onsite systems would be useful. This should be a critical review defining the current state of knowledge on the removal of pathogens by onsite systems and their potential for contaminating groundwater. A risk assessment approach should be used so that a quantitative approach as possible is used to define risks and ways these risks can be best reduced.

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Marylynn V. Yates, Ph.D.
Department of Environmental Sciences
University of California
Riverside, CA

Dr. Cliver has presented a clear, concise review of onsite wastewater systems and the potential for exposure to pathogenic microorganisms associated with those systems. In the course of this review, he has articulated a number of issues that need to be resolved in order to manage the risks associated with onsite wastewater disposal systems. Some of the issues are unique to onsite wastewater disposal systems, such as the extent of pathogen removal in the septic tank itself. However, many of the issues, such as the fate and transport of pathogens after release into the subsurface, apply to all systems that treat and dispose of domestic wastewater. Some of these issues are also pertinent systems for the treatment and disposal of animal wastes, as these wastes also contain microorganisms that are pathogenic to humans.

In order to minimize the risks from pathogens associated with onsite wastewater systems, the systems must be designed and located in an appropriate manner. This discussion will focus on the location of the systems, rather than the design of the actual physical system itself. The optimum situation would be one in which the information about the waste loading properties *(e.g.,* volume, temporal nature of waste flows), environmental conditions (*e.g.,* temperature, rainfall, evapotranspiration), and subsurface properties (*e.g.,* soil hydraulic properties, ground water properties) were input into a model (this term is being used in the broadest possible sense and does not necessarily mean a mathematical computer model). The model would then tell the user where the system should be located with respect to any receiving water bodies to minimize potential risk (the acceptable risk level must be defined by the user). Developing this model will require a variety of different types of activities. These activities may be categorized as follows:

- Basic (laboratory) research
- Applied (field) research
- Literature investigations
- Public education and communication

In the following paragraphs, examples of specific activities in each of these areas will be discussed.

Basic Research

As discussed by Dr. Cliver, a great deal of research has been performed to determine the factors that affect the fate and transport of microorganisms in the subsurface environment. In some cases, the information has been used to develop models to describe the behavior of microorganisms at that particular site. However, the nature of the information is such that it cannot be used to develop input parameters for models that can be *universally* applied. In other words, for a model to be universally applied, the environment must be fully characterized—including such information as soil texture, porosity, hydraulic conductivity, etc. In addition, the manner in which those environmental factors influence the behavior of the microorganisms must be known in a quantitative sense. It is not sufficient, for example, to know that temperature affects the length of time that microorganisms remain infective in the environment. What must be known is the exact relationship between the environmental temperature and the length of time a microorganism remains infective. To complicate matters further, this information must be known for all of the microorganisms of interest, unless a surrogate organism that is truly representative of all pathogens of interest has been identified. To date, much of the data has been collected using a variety of different microorganisms, under a variety of conditions, for a variety of purposes. In many cases, because it was collected for different purposes, it is impossible to compare the information obtained. In addition, many times the information is contradictory in nature. The contradictory information has led many to conclude that it will be impossible to be able to predict microbial behavior. However, it is unlikely that this is the case. It is much more likely that the inability to predict microbial behavior is due to a lack in our understanding of the factors that control their behavior at a *very basic* level. For example, exactly what happens to a virus particle as it sorbs to the surface of a soil particle? What happens to the very structure of the virus particle as it desorbs from a soil particle, or as it encounters an air-water interface? What enables some members of a virus population to survive for much longer periods of time than others? Once we determine, at a molecular level, what is happening, we may be able to predict the behavior of all microorganisms based on their individual characteristics.

Applied Research

An understanding of the molecular-level factors that control the interactions between the pathogenic microorganisms and the subsurface materials is critical to being able to predict their fate. However, we must also have an understanding of the larger-scale factors that control their fate. For example, it is well documented that rainfall can have an influence on the ability of microorganisms to be transported in the subsurface. An evaluation of that effect must be made at the field level and not in laboratory columns. Successful extrapolation of information obtained in laboratory columns to the field has generally been limited to qualitative and relative information, rather than quantitative information. For example, demonstration that a virus can be transported only 10 cm in a particular soil in a laboratory column cannot be accepted as proof that the virus will not be able to be transported only 10 cm in the field.

Another aspect of pathogen behavior that must be studied is in relation to the differences observed when studying pure suspensions of laboratory strains of organisms compared to (for lack of a better term) indigenous organisms in a cell-associated or organic material-associated state. For determining the basic-level interactions, it will be appropriate to use pure, laboratory stains so that confounding factors can be eliminated. However, in order to extrapolate that information to the field, studies must be performed with organisms in the state in which they will exist in the field.

While behavior in uniform porous media may be able to be simulated in the laboratory with some degree of success, it is essential to perform field experiments to examine the effects of soil heterogeneities such as worm holes, cracks, fractures, etc. These soil features may contribute to unexpectedly rapid transport of the microorganisms. Another critical feature that must be examined, as indicated by Dr. Cliver, is the role of the schmutzdecke in the fate of pathogenic microorganisms. The biological activity in this layer may play a role in accelerating pathogen inactivation. It may also trap pathogens, limiting their movement through the soil. On the other hand, microorganisms may be protected from environmental stresses by association with the schmutzdecke, prolonging their survival.

Overall, it will be imperative that the information learned in the basic research studies be “ground-truthed” through field studies.

Literature Investigations

As Dr. Cliver stated, it is reported that contamination from septic tanks is the cause of many waterborne disease outbreaks. However, the circumstances under which the contamination occurred are rarely stated. For example, was the septic tank functioning properly? Was it located in a manner and place in accordance with local regulations? In order to determine how risks from pathogens in onsite treatment systems can be minimized, it is important to assess the circumstances associated with the contamination events. If it is learned that the outbreaks were all associated with improperly functioning or located systems, the course of action to remedy the situation will be very different than if it is found that the systems were properly functioning.

Public Education and Communication

A critical component of risk management is communication. The risks associated with onsite wastewater systems must be put into perspective and communicated to the public in an understandable manner. For example, are onsite treatment systems more hazardous than other wastewater treatment systems? Having the data from the septic tank-associated outbreaks will be critical in answering this question.

The public must also understand the basis for the regulations governing the use and location of onsite wastewater systems. The basis for septic tank setback distances, for example, is a common question. The fact that most setback distances weren’t established using a systematic study of pathogen transport may make this somewhat difficult to handle, but it must be acknowledged. The implementation of new programs such as wellhead protection and source water assessment programs should be communicated so that the public is aware that there is a recognition that these systems are potential contamination sources. In addition, the recent proposal of the Ground Water Rule by the EPA, which was designed specifically to protect ground water supplies from fecal contamination, should be explained.

Research Needs in Decentralized Wastewater Treatment and Management: A Risk-Based Approach To Nutrient Contamination

Arthur J. Gold[1] and J. T. Sims[2]

August 1, 2000

[1]Department of Natural Resource Sciences, University of Rhode Island (agold@uri.edu).
[2]Department of Plant and Soil Sciences, University of Delaware (jtsims@udel.edu)

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Research Needs in Decentralized Wastewater Treatment and Management: A Risk-Based Approach To Nutrient Contamination

Arthur J. Gold[3] and J. T. Sims[4]

I. Abstract

In this White Paper we develop research priorities to improve risk assessment and management efforts targeted at nutrients from decentralized wastewater treatment systems (DWTS). We are concerned with human and ecosystem health at both the micro and the macro-level spatial scales. We focus primarily on the factors that control the movement of N and P from DWTS to ground and surface waters and the research needs related to controlling nonpoint source nutrient pollution from DWTS. At the micro-scale the exposure pathways include the system and the immediate surroundings, i.e., the subsurface environment near to the DWTS. The exposed individual or ecosystem at the micro-scale can be a household well, lake, stream or estuary that borders an individual wastewater treatment system. At the macro-level our focus is at the aquifer and watershed scale and the risks posed to downstream ecosystems and water users by nonpoint source pollution of these waters by nutrients from DWTS.

We recommend the following key research priorities be established to improve our ability to assess and manage nutrient risks from DWTS:

1. The development and use of common sets of methods and measured parameters for studying micro-scale nutrient dynamics and transport in conventional and alternative systems.
2. Increased emphasis on micro-scale research on the site and management factors that affect the risk of nutrient loss from DWTS. At the micro-scale, models can be evaluated effectively and wastewater plumes can be isolated with reasonable effort and expense.
3. Continued emphasis on rigorous field evaluations of nutrient removal in alternative systems subjected to a variety of loading rates, climatic and physical settings. Overview consensus studies need to be compiled that provide guidance to local communities on the designs most applicable to their area.
4. Additional research at "sink" or "hot spot" locations, such as streamside buffers, where groundwater flow is likely to interact with zones of high N and P transformation rates. We need to determine the factors that generate interaction of these zones with nutrient laden groundwater from DWTS and the capacities of these sites to reduce N and P flux from concentrated plumes of effluent.
5. The development of risk categorization models or site indexing approaches where the quantification and complexity of the models or indices match our ability to understand and to parameterize the factors controlling transformations rates in many different settings within a watershed. In particular, we suggest that the use of easily identified physiographic features (soils, hydrology, aquifer characteristics) be explored to classify and describe the vulnerability of aquifers and watersheds to nutrient losses from DWTS.

[3]Department of Natural Resource Sciences, University of Rhode Island (agold@uri.edu).
[4]Department of Plant and Soil Sciences, University of Delaware (jtsims@udel.edu)

II. Introduction: Problem Formulation

Nutrients originating from decentralized wastewater treatment systems (DWTS) can pose a risk to human and ecosystem health. Assessing the likelihood and magnitude of this risk is a formidable and complex challenge. However, a properly constructed risk assessment (Suter, 1993; US EPA, 1996) is essential if we are to design and implement practices for DWTS that minimize the impacts of nutrients on our environment. To do this successfully, we must carefully consider: (i) the specific risks posed by nutrients emitted by DWTS and the sensitivity of humans and ecosystems to these risks; (ii) the pathways by which nutrients move from DWTS to the sectors of the environment where the risk will occur (most often ground and surface waters); (iii) the micro- and macro-scale processes that affect the transport and transformations of nutrients once they are emitted from the DWTS and how this in turn affects risk; and (iv) the effects of current, or alternative, DWTS design and management practices on nutrient transport and subsequent risks to humans and ecosystems.

The nutrients of greatest concern to human and ecosystem health are nitrogen (N) and phosphorus (P), both of which are also essential nutrients for the growth and well-being of plants and animals. However, if N and P from DWTS reach surface waters and ground waters and expose humans and natural ecosystems to nutrient concentrations that are markedly above ambient levels, humans and ecosystems can be threatened. Human health is primarily at risk from high nitrate-N concentrations in groundwater used as drinking water, although in some cases (e.g. surface reservoirs) potentially carcinogenic by-products of algal blooms are also of concern. Elevated concentrations of N and P threaten both freshwater and estuarine ecosystems through accelerated eutrophication, which has many undesirable features, including frequent algal blooms, fish kills, increased turbidity and sedimentation, foul odors and surface scums, impaired recreational and navigational uses of the waters, loss of biological diversity, and habitat destruction.

The probability and extent of exposure of humans and ecosystems to nutrients emitted from DWTS is controlled by a combination of many factors, including localized site characteristics (i.e., soils, hydrology, slope), population density, system design and maintenance, proximity to receiving waters and watershed features. Most DWTS use a septic tank for pretreatment of raw wastewater, discharge the pre-treated wastewater into the subsurface environment and then rely on chemical, physical, or biological processes in subsurface soils for nutrient removal and/or retention. Thus, in contrast to other sources of nonpoint nutrient pollution, such as atmospheric deposition or the agricultural and urban use of nutrients in fertilizers, animal manures, and biosolids where nutrients enter the soil environment via dynamic, biologically active topsoils, DWTS usually release nutrients into low organic matter subsoils, below the rooting zone of most plants. Unfortunately, this often minimizes the likelihood of plant uptake and microbial transformations of N and P, both of which can reduce the potential for nutrient losses to surface and ground waters. Also, in many cases, DWTS generate concentrated plumes of effluent that migrate into the vadose zone and groundwater rather than diffuse inputs – potentially overwhelming the capacity of any chemical, physical, or biological nutrient removal mechanisms operative in the subsurface environment. The nutrient of greatest concern in the watershed also must be considered when evaluating risk and management of risk. This is because N and P are distinctly different in terms of their interactions with soils and microorganisms, their transport pathways in the subsurface (and surface) environments, and in their potential effects on humans and ecosystems. Phosphorus impacts from DWTS are most pronounced in systems plagued by hydraulic failure and in systems bordering surface waters. In contrast, offsite losses of nitrate-N are more often associated with groundwater flow and nitrate-N is known to travel long distances with minimal removal or retention occurring in groundwater aquifers. The nature of the receiving surface water body must also be considered in any risk assessment process. Nonpoint source pollution by P is usually more of a concern for freshwaters, where biological productivity is typically P-limited, than estuaries, which are typically N-limited. Finally, the design, age, maintenance program and many site-specific features can affect the performance of individual systems and thus the likelihood that nutrients will be discharged into the environment at acceptable, or unacceptable, concentrations.

In this paper we examine the risks of nutrients from DWTS to human and ecosystem health at both the micro and the macro-level spatial scales. We focus primarily on the factors that control the movement of N and P from DWTS to ground and surface waters and the research needs related to controlling nonpoint source nutrient pollution from DWTS. At the micro-scale the exposure pathways include the system and the immediate surroundings, i.e., the subsurface environment near to the DWTS. The exposed individual or ecosystem at the micro-scale can be a household well, lake, stream or estuary that borders an individual wastewater treatment system. At the macro-level our focus is at the aquifer and watershed scale and the risks posed to downstream ecosystems and water users by nonpoint source pollution of these waters by nutrients from DWTS. We analyze what is known about the effectiveness of current designs at mitigating these risks and our ability to predict the risk at both scales. Finally, we present a summary of the research priorities at both scales, based on a review of past research and our assessment of current environmental concerns.

III. Assessment and Analysis of Adverse Effects from Nutrients in Onsite Systems

III.A. Nitrogen (N):

The U.S. Environmental Protection Agency (USEPA) estimates that raw human wastewater generates approximately 2-8 kg of total N capita^{-1} yr^{-1} (USEPA, 1980). Using an average value of 4 kg capita^{-1} yr^{-1}, a density of 3 people per household (Valiela et al., 1997) and five houses per hectare (1/2 acre lots), the annual N loading rate from unsewered suburban developments is potentially 60 kg ha^{-1} yr^{-1}. This compares to N loading rates of 6–12 kg ha^{-1} yr^{-1} from atmospheric deposition (Howarth et al., 1996), and 100 to 200 kg ha^{-1} yr^{-1} applied as fertilizer to row crop agriculture (Keeney, 1986; Addiscott et al., 1991).

The drinking water standard for nitrate-N is 10 mg L^{-1} (USEPA, 1996). Excessive levels of nitrate-N can cause “blue-baby” syndrome or methemoglobinemia in infants and other human and ecological problems (Pierzynski et al, 2000). Total N concentrations in human wastewater range from 30-80 mg N L^{-1} (USEPA, 1980). In many areas, much of the total N converts to nitrate-N as the wastewater moves through the soil absorption field and into the groundwater. Thus, wastewater requires removal or dilution by uncontaminated ground waters to meet the USEPA drinking water standard. If a community drinking water system exceeds the USEPA drinking water standard, a series of actions are triggered, starting with public notification. If a State determines that the community system cannot meet the nitrate-N standard, treatment or abandonment of the water source can be mandated. At the discretion of the State nitrate-N concentrations not to exceed 20 mg L^{-1} may be allowed in certain non-community water systems (i.e., the water will not be available to children under

6 months of age, etc.). Although States are not empowered to regulate nitrate-N standards in individual water supply systems, some financial institutions make home loans contingent on the home water supply meeting the 10 mg L^{-1} nitrate-N limit, creating a de-facto form of scrutiny on individual water supplies. (Pers. Comm., R.N. Mendes, Office of Drinking Water Quality, RI Dept. of Health).

Risks to drinking water are linked to the intended use and relative importance of the receiving aquifer. Of greatest concern are unsewered areas overlying shallow, sandy, and relatively deep (>6 m of saturated depth) water table aquifers with potable water. Wells in these aquifers can provide substantial quantities of water for individual wells and community systems, and are a valuable resource that may be difficult to replace if contaminated by nitrate-N. Many of these areas have been mapped either as part of statewide "well-head" protection and groundwater recharge programs or through United State Geological Survey (USGS) water supply investigations. Locations with low yielding water table aquifers are of less concern, given the constraints on their use as drinking water supplies. Low risk aquifers are characterized by low tranmissivities generated by a combination of minimal saturated thickness and low hydraulic conductivities. In addition to low-yielding aquifers, confined aquifers that are protected from contamination by a impermeable, restrictive layer (e.g., clay or silt beds) are at lower risk to nitrate-N exposure from DWTS, which normally discharge above the restrictive layer.

Increased N inputs from watersheds have been implicated in the degradation of estuarine and marine ecosystems across the U.S. (Nixon et al., 1986), including shallow New England estuaries (Valiela, 1990; Lee and Olsen, 1985); the Chesapeake Bay (Shuyler, 1995); the Gulf of Mexico (Rabalais et al., 1996); and the Puget Sound (Inkpen and Embrey, 1998). Increases in N loading to estuarine and marine ecosystems have been linked to phytoplankton blooms (Elmgren, 1989; Nixon et al., 1986); nuisance seaweed growth (Sims and Price, 1998); increased growth of macroalgae (Harlin and Thorne-Miller, 1981); loss of seagrass and other submerged aquatic vegetation (Johansson and Lewis, 1992; Short and Burdick, 1996); increased deposits of organic sediment leading to loss of shellfish habitat (Lee and Olsen, 1995) and hypoxia (Valiela et al, 1992; Rabalais, 1996). In contrast to the long history of nutrient loading models used to address freshwater eutrophication (Smith, 1998), estuarine scientists are still exploring the mechanisms and linkages necessary to quantify the response of estuarine systems to different nutrient loading rates (Valiela et al., 1990; Nixon et al., 1986). Seagrass decline (Short and Burdick, 1996) has been statistically correlated with the density of unsewered houses in estuarine watersheds. Coastal estuaries respond to the mass yr^{-1} of N input (i.e., loading) and concentrations well below the drinking water limit (e.g,. 0.5 to 1. 0 mg N/L) can cause major ecosystem degradation.

<u>III.B. Phosphorus:</u>

There is no drinking water standard for P and there are no regulatory upper limits established for P concentrations in runoff, although USEPA is now in the process of developing a national strategy to identify regional nutrient criteria for different types of water bodies (e.g. streams vs. rivers vs. lakes vs. estuaries). These criteria will presumably be used by states and tribes to reduce the over-enrichment of surface waters, a key goal of the Clean Water Act (USEPA 1999). The USGS in its recent report *The Quality of Our Nation's Water* (USGS, 1999) reported that "..in most streams draining agricultural, urban, or mixed land use, concentrations of total P were greater than background concentrations and the

USEPA desired goal for preventing nuisance plant growth in streams". The current USEPA goal is 0.10 mg total P/L, similar to the 0.10 mg MRP/L standard (MRP=unfiltered molybdate reactive P) recently proposed by the European Community Urban Waste Water Treatment Directive (Withers et al., 2000). Stricter standards have been proposed in some other countries, such as Ireland, which established a "P target concentration" of 0.03 mg MRP/L for rivers (Water Quality Standards for Phosphorus Regulations,1998).

These proposed standards and other reported P concentrations in eutrophic waters (0.01 to 0.10 mg P/L; Correll, 1998) are very much lower than the P concentrations in the effluent from DWTS (11 to 31 mg P/L; Reneau et al., 1989; Crites and Tchnobanoglous, 1998). Additionally, the majority (~85%) of P in the effluent is present as soluble orthophosphate, the most biologically available and potentially mobile form of P. Annual P loading rates from DWTS to a watershed will depend on a number of factors, but primarily on DWTS density. About 170 L of wastewater are generated per capita, per day in the U.S. (USEPA, 1980). Assuming a total P concentration of 16 mg/L (Reneau et al., 1989), a density of three people per household (Valiela et al., 1997) and five houses per hectare ("half-acre" lots), the annual total P loading rate from unsewered suburban developments is potentially ~15 kg/ha/yr. This compares to P loading rates of <1 kg/ha/year from atmospheric deposition (Howarth et al., 1996), ~5 kg P/ha/yr in maintenance fertilization of lawns, and from 10 to 150 kg P/ha year as fertilizer or manure in agricultural production systems (Pierzynski et al., 2000; Sharpley et al., 1998).

Phosphorus is well-known to have many undesirable effects on aquatic ecosystems, primarily because it contributes to eutrophication of surface waters. Eutrophication has been identified as one of the leading causes of water quality impairment in the U.S. today (USEPA, 1996). Typical problems associated with eutrophic waters are: increased growth of undesirable algae and aquatic weeds; low dissolved oxygen levels after the death of algal blooms and nuisance aquatic weeds, which in turn can cause fish kills; turbidity and decreased light penetration through the water column that eventually leads to the loss of benthic plant and animal communities; sedimentation which negatively affects navigational and recreational uses of surface waters; and increased incidences of foul odors, surface scums, unpalatable drinking waters, and nuisance insect problems. Phosphorus loading models are a key facet of many lake eutrophication studies and have been used successfully to guide water quality management for over 30 years (Smith, 1998).

In contrast to N, P is not directly toxic to humans, but has been shown to be involved in several water quality problems related to eutrophication than can impact human or animal health. Examples include the formation of carcinogenic trihalomethanes during the chlorination of waters that have recently experienced algal blooms (Kotak et al, 1993); consumption, by livestock or humans, of waters containing cyanobacterial blooms or the neuro- and hepatoxins released when these blooms die (Martin and Cook, 1994); and, most recently, concerns about the effect on human health of neurotoxins and other toxic constituents released by dinoflagellates, such as *Pfiesteria piscicida*, that bloom in eutrophic coastal waters (Burkholder and Glasgow, 1997).

IV. Exposure Assessment: Micro Level Issues Related to Nutrients

IV.A.1.Nitrogen:

IV.A.1.a. Nitrate Dynamics in Conventional Septic Tank/Soil Absorption Systems:

The fate of N within septic tank/soil absorption systems has been documented in a number of thorough reviews (i..e. Siegrist and Jenssen, 1989; Reneau et al., 1989; Long, 1995; Crites and Tchobanoglous, 1998). Wastewater from households, schools, businesses and other sources that rely upon DWTS initially enters a septic tank where settling and volume reduction occurs as a result of anaerobic decomposition. Reneau et al. (1989) estimated that reductions of 40% of sludge volume, 60% of the biological oxygen demand (BOD), and 70% of the suspended solids occurred in the septic tank. Nitrogen in raw wastewater is subjected to mineralization and settling within a septic tank. Most nitrogen in septic tank effluent is in the form of ammonium-N and organic N. Through settling and periodic pumping of septage, septic tanks can remove approximately 5 – 15% of the incoming N (Laak et al., 1981; Pell and Nyborg, 1989b; Kaplan, 1991).

Siegrist (1989) concluded that soil absorption systems remove approximately 20% of the N in septic tank effluent. However, N removal appears to vary with site factors, such as soil wetness, water table fluctuations and soil texture. Studies conducted on systems located in sandy, well-drained soils show little N removal occurs in soil absorption systems and the underlying vadose zone (Walker et al., 1973b; Keeney, 1986; Lamb et al., 1990; Robertson et al., 1991). Nitrogen removal has been noted in systems located in finer textured soils and sites with fluctuating water tables (Walker et al., 1973a; Reneau, 1979; Cogger et al., 1988), but these types of sites may generate hydraulic failure of the DWTS and constrain the use of conventional designs.

Within the soil absorption system a host of potential transformations can occur as the effluent travels through the vadose zone and into the groundwater. During startup, N can accrete in the biological clogging mat (crust) that often forms at the interface of the native soil and the constructed absorption system. In systems that provide appropriate hydraulic function (i.e., long-term acceptance of wastewater), the lower sections of the crust are subjected to aerobic soil conditions and the rate of mineralization approximates the rate of accretion, minimizing long-term N removal by the clogging mat (Kristiansen, 1981). Clogging mat dynamics can create a misleading impression of N removal within the soil absorption field. In colder weather the mat tends to grow and accrete N and C due to lower microbial respiration rates, while in warmer weather, the mats decrease and contribute mineralized N to septic tank effluent entering drainfield soils. This can cause errors in annual removal estimates if simple mass balance studies that compare N in septic tank effluent to N in leachate from a drainfield are based on data for a single season of the year. It also argues for year round research efforts that better reflect the effect of seasonal changes on N transformation and release from clogging mats.

Immediately below the clogging mat, nitrification and ammonium adsorption by ion exchange reactions (rapid, reversible, electrostatic retention of the ammonium cation) are the primary transformation processes. Under aerobic conditions, nitrification is the dominant mechanism, with NH_4-N oxidized to NO_3-N. Nitrate is a soluble anion that readily leaches through soil. Under anaerobic conditions soil cation exchange sites can adsorb NH_4-N; however the adsorption sites on clays and organic matter can become saturated over time, limiting long-term removal of N by this process. In addition, during aerobic periods (i.e., falling water tables), adsorbed NH_4-N can be released into the soil solution, nitrified and become subject to leaching out of the vadose zone as NO_3-N.

Nitrate can be eliminated from soils and leachate by biological denitrification. This is a microbial respiratory process where nitrate-N serves as a terminal electron acceptor in the absence of oxygen. The process requires anaerobic conditions and an energy source (i.e., a readily decomposable carbon (C) source). Nitrate is reduced to either N_2 or N_2O gas during denitrification and is permanently removed from the leachate as the gaseous forms of N enter the above-ground atmosphere. Most studies have found that the extent of labile (i.e., biologically available) organic C controls denitrification when nitrate-N is abundant and oxygen is limiting or absent.

Denitrification is usually assumed to be of little importance within most of the soil absorption field, due to low concentrations of labile C in subsoils below the point of effluent discharge. Septic tank effluent does contain substantial concentrations of organic C; however, based on column studies, sand filter performance, and sampling wells immediately below the drainfield, the crust and upper portions of a soil absorption field are very effective at removing most of this C from septic tank effluent – minimizing this source of C for denitrification in the vadose zone and the groundwater below the absorption field (Robertson et al, 1991; Aravena and Robertson, 1998). Several studies suggest that anaerobic micosites within aerobic portions of the vadose zone and groundwater can cause localized denitrification that may generate substantial nitrate-N removal (Parkin, 1987; Chen and Harkin, 1998; Jacinthe et al., 1998). The nature and origin of these sites are not well established, but in deeper portions of the subsoil they appear to occur as a result of plant activity associated with root channels and root exudates, as well as from various soil forming processes.

Plant uptake of N from conventional soil absorption fields is expected to be minimal. Conventional drainfields are often placed deep into the soil profile to maintain gravity flow from the buried septic tank. As a result, the effluent is often below the root zone of most plants. This deep placement will also minimize hydraulic failure in distribution pipes due to root growth clogging. Where plant uptake has been observed, it has been limited to plants within 1 m of the effluent distribution line (Brown and Thomas, 1978; Ehrehfeld, 1987).

The dynamic and open nature of soil absorption fields creates methodological challenges and uncertainties for *in-situ* studies of N dynamics. The effects of dispersion, dilution from precipitation and groundwater, and spatial variability of soil properties and infiltration rates confound direct mass balance estimates of individual removal processes and overall N removal. Indirect approaches such as chloride:nitrate-N ratios are often used to separate concentration declines due to dilution from those associated with N removal processes. Chloride:nitrate ratios, however, are an indirect and inexact method for estimating N removal and can provide misleading results. It is never certain that the “upstream” and “downstream” monitoring locations are within the same flowpath. Samples in the vadose zone often display high spatial and temporal variability. As the effluent plume becomes dilute, small concentration changes due to spatial or temporal variability can result in large changes in these concentration ratios and markedly alter estimates of N removal.

IV.A.1.b. N Dynamics in Alternative Systems:

A number of alternative and innovative N removal systems have been developed and subjected to varying degrees of testing and evaluation. Most of these systems subject wastewater to an aerobic environment to promote nitrification, followed by an anaerobic zone where a labile organic C source is available to promote denitrification. Factors that affect both the nitrification and denitrification steps can limit N removal by this combined process. Microbially-mediated N transformation rates decline at low temperatures (Keeney, 1986), suggesting that colder climates might require designs with longer retention times than commonly used in milder locations. In addition to retention time, nitrification is often limited by insufficient aeration (low oxygen), while denitrification is limited by an inadequate supply of labile carbon.

<u>Nitrification component</u>: The nitrification/denitrification processes can occur in a series of distinct components (i.e., filters, tanks, drainfields) or within different portions of a single filter or constructed mound system. In component-based systems, aerobic tanks, single-pass filters and recirculating filters promote nitrification and in some cases, enhance N removal. Nitrate-N removal in these aerobic components varies from 10–50%, most likely occurring either in anaerobic microsites or in situations where the nitrified effluent mixes with anaerobic septic tank effluent. In all of these systems questions surround the extent of nitrification in cold conditions. These systems can also fail to generate nitrification if they become anaerobic, through clogging, which reduces the diffusion of oxygen to the site of nitrification. From a system maintenance perspective, however, anaerobic conditions generate obvious secondary attributes, such as odors and ponding that make gross failures relatively easy to diagnose and then correct. Systems such as these are now in wide-spread use and a growing body of information is emerging on their performance, costs, and maintenance requirements. Input/output mass balance studies are relatively simple for those systems where effluent collects in a dosing chamber or distribution box before discharging to the absorption system or the next component.

<u>Denitrification Component:</u> Aerobic components that generate nitrification also tend to reduce the amount of labile C in the effluent, thus limiting the extent of denitrification that occurs. Several research systems have mixed effluent from an aerobic component with continuous additions of methanol (Andreoli et al., 1979, Boyle et al., 1994) and ethanol (Lamb et al., 1990) to provide the labile C required for denitrification. Anaerobic filters, mixing tanks and soil zones have served as the labile C mixing/denitrification location. However, methanol and ethanol are flammable and potentially toxic and thus pose substantial handling and maintenance demands for small community and individual systems. To overcome this constraint, some systems use wastewater to provide a continuous, fresh supply of labile C in an attempt to sustain the denitrification of nitrified effluent. These systems may not achieve complete N removal, since a portion of the wastestream does not encounter the nitrification and denitrification steps. However, high removal rates (>70%) can be achieved. Examples include:

- Designs that use greywater as a carbon source: The RUCK system, a propriety system, separates greywater (high labile C:N ratio) from blackwater, which contains most wastewater N. The blackwater passes through a layered, aerobic intermittent sand filter that promotes nitrification and is then mixed with unfiltered greywater in an anaerobic media filter where denitrification occurs.

- Designs that use septic tank effluent as a carbon source: Septic tank effluent also appears to be a reasonable C source for denitrification. Substantial N removal has been observed in designs that mix effluent nitrified in an aerobic component with unfiltered septic tank effluent. Examples of these types of systems include the RSF-2 system (Sandy et al., 1988) and the Orenco septic tank/trickling filter system (Ball, 1995).

Rock-filled tanks have been used to provide anaerobic zones for denitrification, although some studies reported clogging, leading to hydraulic and treatment failure (Lamb et al., 1991; Boyle et al., 1994). This problem appears to have been overcome with designs that use open mixing tanks or packing material with greater porosity. In addition many studies reported problems with pump failures; however, great strides have been made in improving the reliability of pumping systems in the past five years, suggesting less risk from this type of failure in the future.

IV.A.1.c. Soil Based Systems:

Nitrogen removal has also been observed on several modified soil absorption systems. Mound systems and "at-grade" systems are often used where the water table is near the surface. The systems rely on nitrification to occur in the aerobic conditions created in the media immediately below the distribution system. Substantial denitrification may occur subsequently if the nitrified effluent encounters a supply of labile C, either through anaerobic microsites (Chen and Harkin, 1998) or as the effluent passes through the original top soil. However, in almost all systems, the supply of labile C is not replenished, so questions arise concerning the longevity of nitrate-N removal, warranting evaluation of long term performance. Shaw and Turyk (994) concluded that minimal nitrate-N removal occurred in mounds and at-grade systems located over sandy media.

Soil absorption fields that rely on shallow narrow drainfields or subsurface irrigation systems may hold promise for nutrient removal and warrant additional research (Rubin et al, 1995). Effluent is expected to have undergone some form of pretreatment to minimize clogging, thus N is likely to be in the nitrate form. Because N is discharged into upper 30 cm of the surface, it is subjected to the full range of N dynamics that occur within the root zone. Vegetated soils can retain N for prolonged periods if the soil is accumulating organic matter, but this is a finite removal process, since the rate of mineralization is related to the extant pool of organic matter. Nitrate can potentially undergo denitrification in anaerobic microsites, in fine textured soils or soils with shallow water tables. This could be a long-term removal process, since the active root system in the upper portion of the soil continually replenishes labile C, through root exudates and root turnover. Denitrification should be not presumed to be a major removal mechanism in all sites. In well-drained, sandy soils, virtually no denitrification has been observed when secondary treated effluent has been discharged to the surface of forest soils (Barton et al., 1999)

IV.A.1.d. Wetland Based Systems:

A substantial research base is developing on N removal in subsurface wetland treatment systems. (Kadlec and Knight, 1996). In particular, lined, cell-based systems are well-suited for mass balance research studies. If N removal is primarily occurring through plant uptake and immobilization in organic matter, we may see seasonal differences in removal and long term saturation of the system. If nitrification and denitrification are the dominant removal processes, the systems have greater potential for long term removal. Aerobic pretreatment may strongly enhance the extent of denitrification in wetland systems.

IV.A.2. Phosphorus:

Phosphorus discharged into the environment from DWTS may pose a threat to water quality and/or human health if transport processes exist to deliver P to surface waters. Inputs of P to DWTS include human wastes (primarily organic P) and domestic inorganic sources of P, such as laundry detergents. Effluents from DWTS contain a mixture of solid and dissolved forms of organic and inorganic P. Most DWTS rely upon chemical reactions (precipitation and adsorption) of wastewater P with soils in drainfields to prevent P from entering shallow ground waters and subsequently discharging into surface waters. The effectiveness of soils, and underlying aquifer materials, in attenuating P movement to waters depends upon a number of factors including their chemical and physical properties, the chemical properties and loading rate of the wastewater, site hydrology, proximity of the site to surface waters, and the design and management of the DWTS. Understanding and managing the risk of P to water quality and human health therefore requires that we not only fully understand those factors that control P retention and release by soils, but that we carefully consider the hydrologic factors and management practices that affect P transport at both local and watershed scales. At the smaller, local scale, we are most concerned with short flow paths that directly and rapidly deliver P discharged by DWTS to nearby streams, rivers, ponds, lakes, or estuaries, either naturally or as a result of system failure. Natural examples mainly include subsurface flow paths such as tile drains, man-made ditches, and soil or hydrologic conditions (restrictive subsoil barriers, shallow water tables) that promote rapid lateral discharge of ground waters (and thus dissolved P) to surface waters. System failures that result in surface ponding of wastewaters can also result in P losses either through (i) direct runoff of ponded effluent to a nearby stream, or (ii) by enrichment of the soil surface with P, which can then be lost, as either particulate P or dissolved P, in subsequent rainfall events or snowmelts. At larger spatial scales, we are primarily concerned with the broader effect of multiple DWTS on the enrichment of ground water aquifers by P leaching downward from drainfields or from soils where wastewaters are applied by other methods (e.g. sand mounds). In these situations flow paths are deeper and transport distances from the DWTS to surface waters are longer.

IV.A.2.a. Phosphorus dynamics in DWTS soils and ground waters

The fate and transformations of P in DWTS have been investigated rather extensively because of public concerns about surface water eutrophication. A number of studies in the past 25 years have elucidated the major processes involved in the retention and release of P in DWTS effluent by soils in the vadose zone and in underlying ground waters. Our current understanding of these processes, at the micro-scale, is illustrated next using a conventional gravity flow system (septic tank, distribution box, and subsurface soil absorption system). Major differences in P transformations and transport processes for other types of DWTS are considered as appropriate.

Within a septic tank most organic P and polyphosphates are converted to orthophosphate (PO_4^{3-}) during the decomposition process. It has been estimated that as much as 48% of the influent P is removed by settling and subsequent pumping of septic tanks (Pell and Nyberg, 1989a). Phosphorus dynamics are strongly related to the uniformity of effluent distribution within a drainfield. In conventional designs, effluent from the tank is discharged into a distribution box and then into trenches or tiles within a drainfield. In a properly functioning DWTS the effluent would next be distributed uniformly within the drainfield where it would then percolate into the vadose zone of the soil and gradually downward to underlying ground waters. In practice, most systems discharge effluent unevenly into the drainfield, often resulting in localized areas with highly elevated effluent (and thus nutrient) loading and other areas where little or no effluent enters the soil in the vadose zone. Uneven distribution of effluent can increase the likelihood of nutrient leaching into groundwaters for several reasons. First, areas with excessive loading can remain under saturated flow conditions longer, promoting deeper percolation of water and nutrients. Second, the capacity of the soil to remove nutrients by chemical or biological means can be exceeded in these areas of elevated nutrient loading. And, finally, the full renovative capacity of the drainfield soil for nutrients is not used when only small portions of the drainfield continually receive the majority of the effluent. Therefore, to assess the risk of P from DWTS impacting ground and surface waters, we must not only understand the biogeochemical reactions of effluent P in soils and groundwaters but how these reactions are affected by site hydrologic factors and system management.

IV.A.2.b. Biogeochemical reactions of P:

Phosphorus in the effluent is primarily found as soluble orthophosphate (~85%) with the remainder as organic and inorganic particulate P in the form of suspended solids. Soluble ortho-P in the effluent can either be: (i) *precipitated* in the soil as a discrete, sparingly soluble mineral phase by reaction with other ions in the effluent or in the soil solution, most commonly aluminum (Al), calcium (Ca), or iron (Fe). Precipitation reactions are rather permanent, but can be reversible, resulting in dissolution of the mineral phase and release of P into solution, particularly if significant changes in pH, redox potential, soil solution composition of Al, Fe, and Ca, or ionic strength should occur; (ii) *adsorbed* to soil colloids by the formation of a strong, but slowly reversible, chemical bond between orthophosphate and clay minerals, Al or Fe oxyhydroxides, or solid phase calcium carbonate ($CaCO_3$). Adsorption reactions typically have at least two kinetic phases - a rapid initial phase that is complete within a matter of hours, followed by a much slower phase that can persist for months and nearly double the amount of P adsorbed in the initial phase. During the slower phase initially adsorbed P may also gradually be converted into less soluble precipitated forms of Al-P, Fe-P, or Ca-P; (iii) *leach* downward in the soil with the percolating effluent, through natural pores, fissures, or cracks. Note that precipitation and adsorption of P can also occur during the leaching process and after soluble P has leached into a lower horizon or into the saturated ground water zone; (iv) *immobilized* biologically by uptake by plants (somewhat uncommon since effluent is usually discharged below the rooting zone of most plants) or microorganisms in drainfield soils. As with N, P that is immobilized may later be mineralized into a soluble form and re-enter the soil solution where it can then be precipitated or adsorbed. Unlike N, however, there are no gaseous pathways for P loss from soils. Particulate P in the effluent can either be mineralized into soluble ortho-P (organic particulate P) or be physically retained in the soil (inorganic particulate P) where P solubility will then depend upon the same factors that control the dissolution of precipitated P or desorption of adsorbed P.

Past research has shown that most drainfield soils are highly effective at retaining effluent P within a short distance of the point of effluent discharge, although the mechanisms and permanency of P retention will vary, particularly as a function of soil reaction (acidic vs. calcareous soils) and the redox status in soils and ground waters at the site. Typical mass balances for P have shown that from 60-95% of effluent P is found in soils within a few meters of the drainfield, even years after system start-up. Other research has identified the existence of a highly P-enriched "rapid transformation zone" immediately adjacent (< 30 cm) to the point of initial effluent infiltration that can retain much of the P entering the drainfield soil. Robertson and Harman (1999) reported that 85% of the total P added in sewage effluent was retained within the vadose zone and that most of this P was found within 30 cm of tile infiltration lines. However, more variable retention of P (23-99%) in the vadose zone was reported by Robertson et al. (1998) for ten mature septic systems with rather widely varying soil properties, illustrating the importance of characterizing the P sorption capacity of soils with depth in the vadose zone when siting DWTS. Whelan and Barrow (1984) and Whelan (1988) reported that P retention in the vadose zone of acid and calcareous soils could be predicted with reasonable accuracy by laboratory measurements of a soil "P sorption characteristic". Some studies, however, have shown that the effectiveness of drainfield soils at retaining P declines with time as the soil adsorption sites become saturated, mineral phases attain an equilibrium state with P in the effluent, and hydrologic flowpaths become altered due to clogging, which in turn decreases the uniformity of effluent infiltration into drainfield soils (Magdoff et al., 1974; Lance, 1977; Nagpal, 1985). "Resting" of DWTS (stopping effluent discharge) for several months has also been shown to regenerate some of the P sorption capacity in the vadose zone and thus extend the site life of the DWTS (Hill and Sawhney, 1981; Sawhney and Starr, 1977). Therefore, to minimize the risk of nonpoint source pollution of ground and surface waters by P, it seems clear that we must understand not only how the site-specific chemical and hydrologic characteristics that affect P retention in soils and groundwaters differ between sites, but also how they change with time.

The exact fate(s) of P in the effluent, initially and in the long-term, depends upon a number of complex, interacting factors, such as: (i) the chemical composition of the effluent itself (pH, elemental composition, alkalinity, ionic strength, redox status); (ii) the chemical and physical properties of the soils in the drainfield - which vary spatially (it is common to find different soil series in individual or adjacent drainfields that vary widely in physico-chemical properties), and which also change with depth (soil horizons also vary markedly in factors related to P retention, such as pH, the content of clay, Al/Fe oxides, and organic matter, and physical factors such as aggregation, porosity, and the existence of preferential flowpaths), and with time as function of effluent dosing rate and frequency; and (iii) the geochemical conditions in the underlying ground waters, particularly the pH, redox status, the P mineralogy in aquifer materials, and major ion chemistry.

In acid soils, ortho-P in the effluent is either adsorbed rapidly by Al and Fe oxyhydroxides or precipitates because the effluent is supersaturated with respect to the solubility product of Al-P (variscite: $AlPO_4 \cdot 2H_2O$) or Fe-P (strengite: $FePO_4 \cdot 2H_2O$; vivianite: $Fe_3(PO_4)_2 \cdot 8H_2O$) minerals. In calcareous soils precipitation of Ca-P minerals (e.g., hydroxyapatite: ($Ca_5(PO_4)_3OH$)) generally should predominate because the effluent and/or soils are supersaturated with respect to the solubility of these minerals. The kinetics of Ca-P precipitation in drainfield soils, however, are not well understood. Some studies have suggested that even though the soil is supersaturated with respect to crystalline hydroxyapatites, more soluble Ca-P minerals (e.g., • -tricalcium phosphates) may form initially and maintain soil solution or groundwater P concentrations at higher values than predicted from standard hydroxyapatite solubility equilibria (Robertson et al., 1998).

Phosphorus mineral equilibria are pH and redox dependent and well established for soils. In general, the highest concentrations of soluble P will occur in near neutral soils where P solubility is controlled by Ca-P; as pH values decrease to less than 6.0 soluble P concentrations decrease due to adsorption reactions and precipitation of Al-P and Fe-P (Reneau, et al., 1989; Robertson et al., 1998) . The effect of pH on P solubility in soils (or in effluent plumes in ground waters) can, however, depend on the redox status at the site. Research has shown that P solubility under oxidized conditions decreases from concentrations of ~5-10 mg P/L at pH > 6.5 (P solubility controlled by Ca-P minerals) to < 0.1 mg P/L at pH < 5.5 (P solubility controlled by Al-P and Fe-P minerals) (Robertson et al., 1998). Under reducing conditions, much lower concentrations of soluble P are found in near-neutral soils (~1 mg P/L or less) because of greater precipitation of Fe-P or adsorption of P by by $Fe(OH)_3$. In contrast to oxidized conditions, P solubility tends to increase slightly under reduced conditions as soils become more acidic. The high degree of alkalinity present in most effluent, however, makes it unusual to find soils that are both highly reduced and highly acidic. The general trend observed for the combined effect of pH and redox on P solubility is that oxidized soils (and plumes) tend to vary rather widely in P concentration as a function of pH, while reduced soils tend to have lower and more stable P concentrations across the pH range typically found in drainfield soils.

Precipitation of P most commonly occurs within a short distance (< 1m) of the point at which the effluent enters the soil because these depths are the most saturated with Al, Fe, Ca, and P from the effluent. At distances farther from the point of effluent discharge adsorption processes predominate over precipitation reactions. Thus it is common to find P solubility in drainfield soils controlled by mineral equilibria in the shallower depths close to the point of effluent discharge, where soils are highly saturated with P. At deeper depths and farther lateral distances from the point of discharge, where soils and ground waters are enriched with P, but not to the point that all adsorption sites are saturated and new mineral phases are forming, adsorption-desorption reactions predominate. With time, as the adsorption capacity of soils becomes saturated, P concentrations in the soil solution increase to values that can be of environmental significance. Phosphorus solubility can, of course, change with time due to alterations in site geochemical conditions. Acidic leachate at a site can dissolve Ca-P minerals and the carbonates that are capable of P sorption and the onset of reducing conditions can promote the dissolution of Fe-P minerals, all of which can result in increased P solubility.

IV.A.2.c. Site Hydrologic Factors and System Management:

In general, the strong affinity of soils and aquifer materials for P, either through adsorption or precipitation reactions results in significant retardation of P transport in vadose zone soils by leaching and by saturated flow in underlying groundwaters. Phosphorus leaching and lateral movement in the unsaturated zone is usually minimal at distances more than 1-2 m from the trenches or tiles where the effluent is discharged. Similarly, studies of the chemical composition of effluent "plumes" in groundwaters beneath septic system drainfields typically report retardation factors for P of 10 to 100 (i.e., the effluent plume has moved 10 to 100-fold further than the P plume). Thus the travel time of P from the point of effluent discharge to surface waters is often years, or even decades. Situations that result in more rapid P transport to surface waters are reasonably well-understood and generally related to siting of DWTS in unfavorable soils and hydrologic settings. The most common example is a subsoil condition (e.g., high water table, impermeable soil horizon) that promotes rapid lateral movement of effluent. The severity of this problem can be compounded if an artificial drainage system (e.g. tile drains or ditches) has been installed at a site to reduce soil wetness by enhancing shallow ground water flow to surface waters (Reneau, 1979). Overcoming the transport of P by saturated flow to nearby tiles or ditches will require larger distances of separation between drainfields and the tile lines, or the use of surface mounds to initially receive the effluent. Both practices will increase the interaction of the effluent with the soil and thus decrease the concentration of P entering the adjacent tile lines or ditches. Other situations where P transport from the DWTS to shallow ground waters or surface waters is enhanced include highly leachable soils with low P sorption capacity (e.g. deep sands with low concentrations of Al and Fe oxides) and soils with well-established preferential flowpaths (e.g. "biopores" caused by plants, earthworms, etc; natural cracks and fissures; or hydrophobic zones). Further exacerbating these situations is the placement of DWTS in close proximity to surface waters, most commonly observed in areas where homes or businesses are located close to the shores of lakes, streams, or rivers.

The management, and design, of a DWTS also affects the potential for P discharge to surface waters. Two common problems are (i) surface ponding of effluent, either due to poor siting or management of the system, which enhances the likelihood of surface runoff of soluble and particulate P from the effluent, or later loss of P-enriched surface soils by erosion; and (ii) uneven distribution of effluent in the drainfield, which creates zones that are saturated with P, as opposed to systems designed, or managed, to evenly distribute effluent (and P) throughout the vadose zone. Even distribution (lateral and vertical) results in the full use of the adsorption capacity of a much larger soil volume, and thus a longer DWTS. site life More even interaction of effluent with soils (i.e., through such techniques as pressure distribution) will also maintain P concentrations in the soil water and percolate at lower values than situations where soils are saturated with P to the point that concentrations in the soil water and percolate differ little from those in the effluent. One approach to enhance P removal from DWTS effluent in soils with limited infiltration capacities is the use of sand filters or re-circulating sand filters (RSF) (Pell and Nyberg, 1989a). These systems can increase the interaction of effluent with the filter media, enhancing P retention, particularly if a low-pressure, re-circulating system is used in which the effluent is collected and evenly passed through the filter media multiple times before discharge into the drainfield (Gold et al., 1992). It may also be possible to amend filters with materials that are highly reactive for P (e.g., calcite, dolomite, de-watered water treatment residuals and red "mud"), increasing the capacity of the filter to renovate effluent and thus extending its longevity (Chowdrey, 1975; Sikora et al., 1976). Since even amended sand filters have a finite capacity to retain P, they should be engineered to allow for replacement or renovation as leachate P concentrations increase to values of environmental concern.

IV.A.3. Research Priorities - Micro-scale Exposure Assessment

IV.A.3.a. Common Field Research Approaches To Characterize N Dynamics: Currently, due to a lack of common methodologies and measured parameters, it is difficult to compare the research results on nutrient losses between different site conditions and different alternative systems. We recommend that long-term mass balance studies are essential to further our understanding of the risks posed by conventional systems and the risk reduction that can be achieved through alternative systems. *In-situ* studies should be encouraged that track transformation and removal processes in different settings and in different alternative systems (Chen and Harkin, 1998; Johns et al., 1998). In addition any future work must carefully describe site conditions (i.e., background nutrient levels, soil properties, water table dynamics). Additional research into the efficacy of 15N enriched ammonium-N for mass balance studies could yield useful insights for future research.

IV.A.3.b. Geochemical Modeling Of P Dynamics: We need to further improve our ability to model the geochemical transformations of P in the vadose zone and ground waters and to integrate this with site hydrologic factors to predict P transport. Recent, highly intensive monitoring and research projects, such as those conducted by Robertson et al. (1998), suggest that our understanding of P sorption and mineral equilibria can be used to explain, with reasonable accuracy, the attenuation of P in drainfield soils and in underlying ground waters. Studies such as these have, however, also raised questions about the short and long-term kinetics of P retention and release, the effects of changing site properties on P equilibria (and sorption-desorption), and the interaction of site hydrologic factors with P movement in soils and shallow ground waters. It seems unlikely that widespread, intensive monitoring of P movement in ground water plumes and vadose zone soils will be possible. Therefore, a multi-disciplinary effort to develop a model (or adapt an existing model) to predict P movement by subsurface pathways, as a function of effluent characteristics, soil properties, aquifer geochemistry, site hydrology, system design and management should be a high research priority. Since past research has identified the situations where P mobility is likely to be greatest (e.g., shallow soils with low P sorption capacity, high water tables, impermeable subsoil layers, in close proximity to surface waters sensitive to eutrophication) it should be possible to focus this modeling effort in a manner that will have the greatest risk assessment benefits. Geochemical modeling results need to compared against field data and against the results obtained from simpler indices of nutrient retention based on soil and site characteristics (see below).

IV.A.3.c. Nutrient Removal and Site Characteristics: Given the large variability in N and P removal among conventional systems, we recommend that additional research is needed to help us rapidly identify those sites that possess an innate capacity for nutrient removal and thus are at less risk for offsite movement. Currently, we lack understanding on the range of soil features and water table dynamics that generate a high capacity for nitrification/denitrification in soil absorption fields. We have many unanswered questions in this regard. What site factors and soil characteristics produce denitrification microsites within predominantly aerobic media? What is the range of soil textures and saturation that can promote denitrification without generating hydraulic failure? Can we establish the efficacy of coupling sites at risk for clogging and hydraulic failure with some type of aerobic pretreatment and thereby enhance denitrification and promote long term wastewater acceptance? Evaluation of existing data in varying risk scenarios (Hoover et al., 1998) might identify soil and site characteristics where N dynamics need to be characterized in greater detail as well as sites where research is adequate for risk management.

IV.A.3.d. Site Indices Of Nutrient Retention: We suggest that there is a need to integrate our current knowledge on N and P transport from DWTS to surface and ground waters into a rapid, systematic approach that can be used to identify sites that are most suitable, or are unsuitable, for various types of wastewater treatment systems. An agricultural analogy is the "Phosphorus Site Index" now under evaluation by the USDA Natural Resources Conservation Service (NRCS) and many U.S. states as a means to rapidly identify "critical source areas" in watersheds where the combination of soil and hydrologic properties at a site (erosion, runoff, drainage, proximity to surface waters, etc.) and site management (P fertilization practices, current soil P status, etc.) combine to make an agricultural field highly likely to deliver P to surface waters (Gburek et al., 2000; Lemunyon and Gilbert, 1993). It seems likely that a similar approach could be used with DWTS, both to rate the potential of current systems to be significant sources of P and N to receiving waters and to site new systems properly (including the selection of the most effective system for the site of interest). Site properties such as sediment grain size, carbonate mineral content, water table dynamics and electron donor content could be readily incorporated into site indices at the regional scale. At the local scale site properties can be combined with design parameters such as the effluent distribution system, effluent characteristics and the soil volume available for nutrient transformations to provide a more refined performance index. This again requires a multi-disciplinary effort and should be closely integrated with the geochemical/hydrologic modeling mentioned above.

IV.A.3.e. Alternative Systems: Important information gaps still plague assessment on the effectiveness of "alternative systems" at reducing the likelihood of N and P transport to surface waters. Research in this area should focus on both modification and/or remediation of existing systems to improve their effectiveness or extend their site life (e.g., changes in system management that alter effluent properties, installation of pre-treatment systems) and innovative designs that are specifically tailored for new systems in areas where reducing the likelihood of N and P movement to receiving waters is a high priority. In general we need to focus on site factors and design factors that control the variability and long term performance in nutrient removal within and between different alternative system designs. Some specific research areas include:

- <u>Aerobic Filters and Tanks</u>: Additional research is needed to understand the reasons for the variability noted in nitrate removal in aerobic filters and tanks. Factors such as loading rate, retention time, effluent strength, textural distribution of the media and temperature need to be considered. The value, economics, and practicality of "dosing" septic tanks with chemical additives to precipitate additional P is also in need of evaluation.

- <u>In-Ground Wetland Systems:</u> Research questions involve the processes responsible for nitrogen removal in wetland systems and the long-term effectiveness of wetlands at sequestering P. In particular, the factors that control seasonal and long term performance of the extent of plant uptake, immobilization, nitrification/denitrification, and the sorption and desorption of P by redox-sensitive soil colloids (e.g., Fe-oxides).

- Mounds and At-Grade Systems: As with all types of soil based systems, research is needed to evaluate the effects of site conditions on N and P removal in mounds and at-grade systems. In particular, what are the site factors that promote denitrification in the vadose zone and shallow groundwater and what is the source and expected sustainability of labile C necessary for longterm denitrification? Can mounds and at-grade systems be amended with by-products that enhance the retention of P in insoluble forms?

- Rootzone systems: Research is needed on the effects of different site conditions on N and P removal dynamics. What conditions promote denitrification and microbial immobilization of P? How long can a rootzone system tie up N and P in soil organic matter before the system is saturated?

IV.B. Exposure Assessment: Macro Level Issues Related to Nutrients:

IV.B.1. Nitrogen:

Great uncertainty surrounds the fate of N in groundwater. At both landscape and watershed scales it is very difficult to track the fate of nutrients along the flowpath from a DWTS to a receiving surface water. Within ground waters, nutrient plumes from DWTS are subject to dispersion/dilution and their flowpaths can be strongly influenced by the presence of layers or lenses of media with elevated hydraulic conductivity (Freeze and Cherry, 1979). Thus, documenting the flowpath of groundwater plumes can be a daunting task that requires an intensive, multi-level monitoring network. For example, Robertson et al. (1991) used between 250 and 500 individual sampling wells to examine the fate of septic system plumes in a sandy aquifer over distances of 20–130 m. In addition, groundwater flow rates are often very slow (i.e., 1–5 cm/day), minimizing the usefulness of introduced tracers for landscape scale studies. A number of studies have used differences in the natural abundance ratio of 15N/14N to track different sources of N within a watershed (Flispe and Bonner, 1985; Doll, 1997); however, there is great controversy over the efficacy of this method, due to the variation in the ratio induced by nitrogen transformations in the soil (Keeney, 1986). Additional confounding factors, such as other sources of nutrients or additional inputs of chloride (the common conservative tracer used to track the plume) also argue against the likelihood of advancing our knowledge of groundwater nitrate dynamics through the widespread use of well network studies that track existing septic system plumes over long distances.

Nevertheless, there is a substantial, existing knowledge base that can be used to help guide future research. Several studies have shown that groundwater nitrate-N concentrations rise with increasing density of unsewered residential development (Bicki and Brown, 1991; Valiela et al., 1992; Winter, 1999). Climatic factors, such as the timing and extent of precipitation and evaporation, and the nitrate-N losses associated with the surrounding land uses influence the response of groundwater nitrate-N concentration to unsewered housing density (Cogger, 1991). A number of studies suggest that N removal cannot be simply related to residence time or travel distance. Instead, N removal depends on the specific characteristics of the receiving aquifer and more specifically with the characteristics that occur in *selected environments* along the groundwater flowpath.

IV.B.1.a. Aquifer Characteristics That Affect Vulnerability to Nitrate-N Contamination:

Aquifer characteristics influence the risk of nitrate-N contamination and suggest an opportunity for the development of site indices of aquifer vulnerability. Several studies have reported substantial groundwater denitrification in aquifer zones dominated by pyrite rich deposits. In these zones sulfide or ferrous iron may serve as the energy source for denitrification, rather than labile C (Kolle et al., 1985; Pederson et al., 1991; Postma et al., 1991; Korom, 1992); however this is expected to occur in selected settings and is not likely to be important in most aquifers. We encourage close collaboration with USGS to identify anaerobic aquifers that contain a suitable reservoir of electron donor constituents (e.g., trace quantities of organic carbon, sulfide minerals or ferrous iron) and hold the potential for groundwater nitrate-N removal.

Limestone areas are at one end of the spectrum of risks associated with aquifer or watershed exposure to nutrients generated from DWTS (Keeney, 1986). The soils are often shallow and groundwater can flow rapidly in preferential pathways through cracks and fissures. Septic system leachate has been found to move in groundwater from a source area to surface waters in a matter of days (Dillon et al., 1999), limiting the magnitude of any transformations that might occur along the flowpath.

Aerobic, sandy, water table aquifers also appear to have minimal capacity for nitrate-N removal. Denitrification is expected to be limited by a lack of labile C and an absence of reducing environments (Starr and Gillham,.1993). Declines in nitrate-N concentrations are largely attributed to dilution and dispersion. However, dilution/dispersion does not always generate rapid declines in groundwater nitrate-N concentrations. Intensive, groundwater studies in Ontario (Robertson et al. 1991) found effluent moved in narrow plumes with minimal dispersion and dilution for >100 m. In addition, other researchers have noted that nitrate-N entering groundwater from the unsaturated zone often moves in the upper portions of deep aquifers, rather than mixing evenly throughout the entire saturated depth (Hill, 1982; Spruill, 1983; Perkins, 1984). This limits dilution and suggests a high risk to shallow wells. Conversely, these observations suggest that the risks of groundwater contamination by nitrate-N may be less than expected in deep wells that draw water from lower portions of aquifers. Also, if nitrate-N enriched plumes remain in the shallow groundwater, this increases the potential for nitrate-N removal -- if the water table approaches the surface as it discharges to streams and surface water.

IV.B.1.b. Watershed Issues: "Hot Spots" in the Landscape:

There is an emerging consensus that our understanding of watershed N dynamics is beset by uncertainties surrounding the role of "sinks" – areas within a watershed that are capable of removing nitrate-N from ground waters (Jordan et al., 1997; Howarth et al., 1996; Valiela et al., 1997). Of particular note is that localized "hot spots" with high N removal rates can occur within an aquifer and that these locations can account for much of the nitrate-N removal within a watershed – if nitrate-N laden groundwaters traverse these locations. By focusing research on "sink" locations, we may be able to advance our overall understanding of watershed risks posed by nutrients from DWTS.

Substantial rates of groundwater denitrification have been observed when groundwater plumes approach the ground surface (i.e., flowpaths occur in the upper groundwater in aquifers with shallow, i.e., < 1 m, water tables). These locations often have anoxic or hypoxic groundwater, and elevated "soil" organic matter or DOC (Starr and Gilham, 1993; Robertson et al., 1991). Areas with shallow water tables can be identified through soil survey maps, based on soil drainage class or hydric soil characteristics; however, we need to couple these areas of high potential for N transformations with knowledge regarding their likelihood of intercepting nitrate-N laden groundwater.

IV.B.1.c. Streamside Buffers and Groundwater Nitrate Removal:

In humid areas, where annual precipitation exceeds annual evapotranspiration, groundwater will enter streams, rivers, lakes and estuaries as "baseflow" or groundwater recharge (Winter, 1999). Because groundwater often approaches the surface as it moves towards surface waters there are areas in the near stream environment that may be the site of nitrate-N removal from discharging groundwaters. These areas include: (i) riparian zones, i.e., lands that border streams and surface waters and (ii) hyporheic zone, i.e., the sediments that constitute the stream bed.

Riparian Zones: There is a substantial body of research documenting groundwater nitrate-N removal in riparian zones (Pavel, et al., 1996; Hill, 1996; Correll, 1997;. Lowrance, 1998). The extent of removal may be influenced by the hydrology, soils and vegetation of the riparian zone. Removal can occur through plant uptake, immobilization in organic matter or denitrification. In certain settings these streamside zones have been found to be a major sink for groundwater nitrate-N leaving upland agricultural and suburban lands. Their preservation, protection and restoration could be a key factor in sustaining or restoring watershed functions in certain watersheds. (Gilliam et al. 1997).

Riparian zones display a great variation in groundwater nitrate-N removal. Groundwater nitrate-N removal appears to be limited to riparian zones where the water table is shallow and organic deposits accumulate in surface soils. Soil mappers often use the hydric classification to identify these types of soils. Conversely, riparian zones with deep water tables and non-hydric soils may not serve as groundwater nitrate-N sinks (Correll, 1997).

Flowpaths influence the extent of groundwater nitrate-N removal in riparian zones (Hill, 1996). Substantial nitrate-N removal has been noted where nitrate-N laden groundwater flows through the upper 1 to 2 m of soil – while minimal removal has been observed when groundwater moves at greater depths below the soil and upwells directly beneath streams and other sources of surface water. If groundwater emerges in surface seeps upgradient of riparian wetlands, surface flow can occur rapidly (i.e., 1–2 hours) across the riparian zones, minimizing the potential for N removal. Within riparian zones research is needed on the factors that control the depth of the biologically active zone (i.e., water table dynamics, soils, geomorphology, type of vegetation, age of vegetation) and the relationship between the width of different riparian settings and groundwater nitrate-N removal.

Hyporheic Zone: Considerable upwelling of groundwater can occur directly into streams from deeper groundwater. Some stream sediments have substantial deposits of anaerobic organic matter and can be a location of high nitrate-N removal rates for groundwater discharging into the stream (Robertson et al., 1991). Other studies have suggested that the hyporheic zone is a minor sink for nitrate-N (Hill, 1997). More research is warranted on N dynamics in hyporheic zones. Particular questions concern: denitrification rates and retention time within the organic sediment, and the factors that cause upwelling to bypass organic sediment deposits and move into the stream through more mineral sediment. Also, we lack information on stream characteristics that are associated with organic sediments (i.e., stream order, flow rates) and the relationship between organic sediments and geomorphology, soils and land cover.

IV.B.1.d. In-stream nitrate-N removal:

Understanding the factors that influence "in-stream" nitrate-N removal can aid in our ability to manage the risks posed to estuarine waters by nutrient loading from DWTS. Constructed wetlands can generate substantial denitrification and have been suggested as a means to control N export from the Mississippi basin to the Gulf of Mexico (Mitch et al., 1999). Reestablishing estuarine health through N control by wetlands restoration in the Mississippi basin may demand substantial (0.7–1.8 % of the basin) land conversion. Clearly, in-stream wetlands restoration and performance holds promise and warrants additional research attention on such issues as design factors influencing removal (i.e., retention time; depth; plant materials) and long term performance.

The result of recent USGS stream monitoring and modeling (Sparrow Model) also stress the importance of in-stream nitrate-N dynamics to the delivery of land based N to coastal waters. Alexander et al. (2000) concluded that nitrate-N removal is higher in small streams than large rivers. They theorize that denitrification in the bottom sediments of in small, shallow streams can be a significant source of nitrate-N removal. In larger streams they suggest that the proportion of interaction between stream and bottom sediments is too small to have notable effects on nitrate-N dynamics.

IV.B.1.e. Modeling aquifer and watershed risks from septic system N:

The consequences of unsewered developments have long been examined with nitrate-N loading models. These models relate population density and the extent of fertilization by various land uses to groundwater nitrate-N concentrations in underlying aquifers. A variety of studies have shown that these predictions can capture general trends and approximate actual groundwater quality. Models have been used in New England, California and other locations (Frimpter al., 1990; Hantzsche and Finnemore, 1992). In most cases, septic system loading models are comparable to models that have been used to relate N loading by various agricultural practices to water quality (Gorres and Gold, 1996). Quantifying the contributions of septic systems to the total loading of nutrients from a watershed should follow a similar approach and have similar levels of uncertainties as the approach used in documenting nutrient loading from agricultural settings. For both of these nonpoint sources of nutrients, there is comparable consensus on the extent of nutrient losses that enter the upper portion of the vadose zone (i.e., the zone beneath the rootzone). However, septic systems differ from cropland in their potential to create narrow plumes of nutrient rich groundwater, rather than the diffuse loading expected from row crop agriculture. These concentrated effluent plumes may be important to the estimate of removal that occurs along the groundwater flowpath from the septic system to the receiving stream or surface water that connects a septic system to the rest of the watershed.

In models such as these, long-term aquifer concentrations are estimated at the aquifer-scale. The models stratify a recharge area into distinct land uses and compute an area-weighted mean concentration in the aquifer from the annual recharge and the nitrate-N loading expected from the vadose zone of each strata. Generally, nitrate-N is assumed to be conservative in the vadose zone and groundwater. In general, for both agriculture and DWTS, the greatest uncertainty in modeling is associated with the extent of N removal in the vadose zone and in the groundwater. Valiela et al. (1997) employed this type of model to estimate N loading to estuaries, with an additional assumption (based on interpretation of published literature from a variety of aquifers) that linked groundwater nitrate-N loading to distance between source areas and surface waters. Valiela et al. (1997) noted that the largest source of uncertainty in the model was the amount of nitrate-N lost in aquifers by processes that could not be quantified.

Nitrogen loading models, however, can exhibit great errors at the watershed scale. Several recent landmark studies have found that in watersheds ranging from local scales (i.e., 1000 ha) to regional scales (i.e., the Mississippi Basin) riverine discharges account for less than one-third of the anthropogenic loading of N to the watershed (Jordan et al., 1997; Howarth et al., 1996). Incorporating a riparian zone function into watershed models may improve model accuracy. Recent advances in GIS techniques now permit automated assessment of drainage pathways and may be useful for estimating the interaction of nitrate-N laden groundwater with streamside sinks. The potential removal associated with different locations could then be factored into the models used to estimate/predict watershed risks from DWTS

IV.B.2. Phosphorus:

The more effective attenuation of P transport (relative to nitrate-N) from DWTS to surface waters by soils and aquifer materials has resulted in fewer macro-scale concerns about P impacts on most surface waters, and thus fewer watershed scale research efforts to quantify P losses. In most cases, the general opinion on the impact of P from DWTS on water quality has changed little in the past 25 years. Jones and Lee (1979) assessed the effects of P from DWTS on ground water quality in northwestern Wisconsin from 1972-1976 and stated "..No evidence for phosphate transport from septic tank effluent was found in any of the monitoring wells, even though this is a sand aquifer with a relatively high groundwater velocity" and "..in general, phosphate will not be transported from septic tank wastewater disposal systems and thereby contribute to excessive fertilization problems". The authors speculated that a very limited number of water bodies directly adjacent to septic tank disposal systems might be at risk. Gilliom and Patmont (1983) conducted a similar study in the Puget Sound watershed in Washington and developed a mathematical analysis (Monte Carlo simulation) of P transport from DWTS to a small lake. They concluded that "..movement of more than1% of effluent P to the lake was rare" and that any P loading to the lake was mostly associated with "septic systems in wet areas that may contribute P to the lake by both shallow groundwater flow and the surfacing of septic effluent and subsequent movement to the lake by overland flow". Chen (1988) investigated P movement in ground waters from 17 septic tank disposal systems located near the shores of eight lakes in New York State. All systems showed "good removal of ortho-P". Groundwater in three of the 57 wells monitored exceeded the current USEPA water quality goal of 0.10 mg P/L; one site was located on a steeply sloping (>10%) soil, and the other on a soil with a very shallow water table. Reneau et al. (1989) reviewed the literature on P transport from DWTS to ground and surface waters and stated "..the limited movement of P away from on site wastewater disposal systems is well-documented" and that "..most field studies indicate that P contamination is limited to shallow groundwaters adjacent to the systems". As noted earlier, Reneau et al. (1989) identified coarse-textured soils with low P sorption capacity, poorly drained soils, and soils with poor effluent distribution as situations with the greatest likelihood for P loss. Weiskel and Howes (1992) monitored "near-field effluent" and groundwater quality in a densely populated (~10 houses/ha) coastal watershed served by DWTS (Buttermilk Bay, Massachusetts). Virtually all (99.7%) of the effluent P was retained in the aquifer at this site. Some "near-field" (5 m downgradient) enrichment of groundwaters with P was noted and attributed to reducing conditions induced by DWTS effluent. The authors concluded that while "..septic systems are clearly a major potential source of N and P to coastal waters..", septic effluent was a "minor source" of P to coastal waters. Finally, Robertson et al. (1998) conducted a detailed study of 10 "mature" septic system plumes in central Canada. Six of the 10 sites had P plumes > 10 m in length with P concentrations elevated about 2 orders of magnitude (0.5 to 5.0 mg/L) compared to natural background concentrations. The authors concluded that "..phosphate plume velocities are substantially retarded compared to groundwater velocities at all sites (R=20 to 100).." but felt that P migration velocities at some sites (calcareous sands) were fast enough to be of concern, given current minimum setback distances established for septic systems relative to surface waters. In general, it appears that the consensus of scientific opinion is that the long distance transport of P from DWTS to surface waters is a much lower risk than nitrate-N transport, except in certain, well-defined situations.

Based on this research, and other studies such as the “micro-scale” research cited earlier, the major “macro-scale” environmental issues with regard to P and DWTS today are: (i) siting considerations related to the proximity of the DWTS to surface waters, such as any site properties that will facilitate more rapid P movement to surface waters. Examples include a better understanding of site hydrology and soil/aquifer geochemistry, both of which affect P retention and the rate of P movement in the landscape; (ii) density of DWTS in a watershed, which relates to annual loading and water body sensitivity to P. For example in Delaware where total maximum daily loads have been established for the Inland Bays watershed (a national estuary), reductions in P loadings of 40-65% of present values will be required for these estuaries and their tributaries to meet “fishable” and “swimmable” criteria under the Clean Water Act. Thus, the long-term concern is whether the current, (or future, as coastal development proceeds) loading of P to shallow ground waters will eventually deliver, in base flow, P in excess of the TMDLs for the watershed; (iii) system design and management, particularly as this affects the likelihood of system failures which can result in more rapid, surface transport of P. Or, the value of innovative designs for new systems that can more efficiently retard P transport and/or remediating existing systems to improve their effectiveness in removing P from ground water discharge.

IV.B.3. Research Priorities: Macro-Scale Exposure Assessment:

IV.B.3.a. Indices Of Aquifer and Watershed Vulnerability To Nutrient Losses: In the near-term, we believe that substantial benefits for risk assessment will result from the development of indices that classify N and P removal potential based on physiography, mineralogy, and hydrologic characteristics in a watershed. These indices could guide state and federal regulatory agencies in the development of the pollution control strategies required by TMDL agreements to reduce nonpoint source pollution of surface waters by nutrients from all sources. We question the capacity and practicality of spatially-explicit, i.e., "distributed models" to quantify the fate and transport of nutrients from DWTS in a watershed, given the uncertainty due to spatial variability and the gaps in understanding of factors controlling transformations rates in many different settings within a watershed. We recommend careful examination of the costs and benefits associated with aggregated, decision-support models (or indices) as compared to the high investments associated with more complex and highly explicit models.

IV.B.3.b. Cost Effective Research Methods: Improved research methods are needed to develop cost-effective, *in-situ* techniques to evaluate low rates of groundwater N and P removal within discrete aquifer locations . Low transformation rates can account for substantial removal, given the long retention time experienced by groundwater moving from source areas to surface waters or wells. A key aspect in this regard is need for methods to assess the rate and extent of reversibility of P attenuation by soils and aquifer materials. Additionally, the high spatial variability (vertical and horizontal) associated with groundwater nitrate dynamics demands techniques that can generate timely insights into potential hotspots of nitrate removal. Well injection techniques offer a promising option (Trudell et al, 1986; Gillham et al., 1990; Nelson et al., 1995; Istok et al, 1997), particularly those that that track the fate of isotopically enriched nitrate in the spike plume.

IV.B.3.c. Stream-based Nutrient Sinks: We recommend that estimates of streamside and "in-stream" N and P removal be incorporated into watershed models – and research encouraged to improve the state of knowledge on these topics. Specific research areas include:

- The reversibility of P retention by stream-bed materials, given the redox-sensitive nature of many P sorption and precipitation reactions.

- Site factors and management practices that influence the capacity of stream side buffers and wetlands to remove groundwater N and P in both concentrated and diffuse leachate plumes. The long-term fate of P in wetlands (source vs. eventual sink) is an area of particular importance, since removal of P by harvesting plants or dredging wetland sediments is very unlikely to occur.

- Mappable indicators (i.e., soils, geomorphology, landuse/land cover) that can identify stream reaches bordered by high N and P retention zones and to then couple those indicators with regional physiographic features (soils, hydrology, aquifer characteristics) that classify and describe the probable interaction of groundwater N and P from DWTS to the retention zones around the streams.

- Riparian zone models to extend the insights from intensive and costly watershed riparian zone studies to other areas. In addition, more qualitative indices of risk reduction from streamside locations should be investigated based on available digital spatial databases.

IV.B.3.d. Nutrient Dynamics In Aquifers: Further research is clearly needed to evaluate the long-term fate and transport of N and P in aquifers, particularly given the TMDL process now underway in many U.S. states. It could encompass plume dynamics and characteristics that generate different biogeochemical transformations in groundwater. In addition, these studies may explore the potential for identifying N from DWTS based on the ratio of the stable isotopes of N. We caution that these are large and complex task that may not yield timely results. We encourage insights from these types of studies be incorporated into the development of indices of aquifer and watershed vulnerability.

V. Conclusions and Implications

Accurately assessing the risk of nutrients from DWTS to human and ecosystem health is a critical environmental issue today. We recommend that new research on DWTS nutrient issues focus on improving our understanding of exposure risks, rather than on "dose-response" relationships of exposure vs. human and ecosystem health. We believe that exposure risks represent a tractable research arena. The groundwork is established that will permit the research community to generate timely and cost-effective returns from investments in risk exposure research – particularly at the micro-scale. As ecosystem scientists reach consensus on "dose-response" relationships, we encourage EPA to incorporate those findings into the overall risk management strategy for DWTS.

Our analysis of the past research on N and P movement from DWTS to ground and surface waters has identified a range of specific, critical research needs with regard to exposure assessment at both the "micro" and "macro" scales (see above). Based on these research needs, we recommend the following key research priorities be established to improve our ability to assess and manage nutrient risks from DWTS:

1. Increased emphasis on research at the micro-scale (rather than the macro-scale) on the fate and transport of N and P from DWTS. We suggest that research focus on field efforts to assess nutrient dynamics in conventional and alternative systems - both within and in the immediate vicinity (i.e., within 10 m) of DWTS. At the micro-scale, models can be evaluated more effectively and wastewater plumes can be isolated with reasonable effort and expense. In addition, studies can be conducted that will provide timely improvements in our understanding of the site and management factors that affect the risk of nutrient loss.

2. The development and use of common sets of methods and measured parameters for studying nutrient dynamics and transport at the micro-scale. Currently, due to a lack of common methodologies, it is difficult to compare the research results on N and P losses

between different site conditions and different alternative systems.

3. Additional research at "sink" or "hot spot" locations on the landscape where groundwater flow is likely to interact with zones of high N and P transformation rates. Of particular interest are ecosystems that develop at the land/water interface. We need to determine the factors that generate interaction of these zones with nutrient laden groundwater from DWTS and the capacities of these sites to reduce N and P flux from concentrated plumes of effluent. A better understanding of the factors controlling the reversibility of P attenuation at sink locations is also a high priority for research.

4. A critical analysis of current approaches to estimate N and P loading at the watershed scale is needed, given the pressures on state regulatory agencies to develop TMDLs and pollution control strategies that are source-specific. This requires at least statewide, or regional, interactions between scientists and regulatory agencies to assess past research and evaluate the validity of models now being used to estimate nonpoint source contributions of N and P from DWTS. Research is needed on the efficacy, uncertainty and practicality of spatially-explicit, i.e., "distributed", modeling approaches vs. "lumped" modeling approaches that aggregate watersheds into large units.

5. We encourage the development of risk categorization models or site indexing approaches where the quantification and complexity of the models or indexes matches our understanding of factors controlling transformation rates in many different settings within a watershed and our ability to obtain the level of information needed to parameterize the model. In particular, we suggest that the use of easily identified physiographic features (soils, hydrology, aquifer characteristics) and system design and management practices be explored to classify and describe the vulnerability of aquifers and watersheds to nutrient losses from DWTS.

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Appendix I: Research Needs In Decentralized Wastewater Treatment and Management: A Risk-Based Approach to Nutrient Contamination

Table I-1	Research to Improve Micro-Scale Assessment of Exposure Risk

Strategic Focus	Tasks	Product	Uses
What micro-scale site characteristics affect the variability and long term performance of nutrient removal from DWTS? What is the range of soil textures and saturation that promote denitrification without generating hydraulic failure? Can we establish the efficacy of coupling sites at risk for hydraulic failure with aerobic pretreatment and thereby enhance denitrification and long term hydraulic performance?	Develop and encourage the wide-spread use of common field research methods that track transformations and removal processes of septic tank effluent throughout the system and the micro-scale environment. Develop and test models to predict P movement by subsurface pathways, as a function of micro-site characteristics and system design.	Reliable and comparable data on the relationships between nutrient removal and micro-scale characteristics. Indices that rate the exposure risks from conventional technologies at different locations. Indices that rate the risk reduction associated with pretreatment at different locations.	To prioritize locations that warrant alternative technologies for new systems and for upgrades. To link location characteristics with different degrees of pretreatment required to minimize risks of nutrient losses.

Table I-2	Research on Alternative Systems

Strategic Focus	Tasks	Products	Uses
What factors affect variability and long term performance for nutrient removal in different alternative DWTS? What conditions and designs promote denitrification in aerobic filters and pretreatment tanks? What is the long term removal expected from plant uptake and microbial immobilization in rootzone systems and wetland systems? What is the viability of dosing DTWS or amending filters with chemical additives to precipitate P?	Develop common approaches to quantify and track transformations and removal within different components of alternative systems. Continued emphasis on rigorous field evaluations of alternative systems subjected to a variety of loading rates, climatic and physical settings. Overview studies to compile, analyze and report on the research results.	A comparable dataset on the fate and removal of nutrients in alternative systems. Consensus on the range of expected treatment and sources of variability from different designs subjected to a variety of loading rates, climatic and physical settings.	To develop guidance to local communities on designs most suited for the conditions in their area.

Table I-3	Research Related to Landscape Sinks of Nutrient Removal

Strategic Focus	Tasks	Products	Uses
What is the role of in-stream and streamside removal in reducing watershed nutrient loads from DTWS? What are the site factors and management practices that affect the capacity of streamside areas to remove nutrients in concentrated plumes and diffuse groundwater flow? Are there mappable attributes that relate to the streamside characteristics that generate high nutrient removal capacities?	Conduct streamside field research that evaluates the effects of micro-scale site factors on nutrient transformations and removal. Develop and test models that explore the effects of streamside characteristics and management on nutrient removal. Develop and test indices that use spatial attributes to identify streamside locations with high potential to function as nutrient sinks.	Data on the role of site characteristics on the extent of DTWS nutrient removal in different types of streamside locations. Decision support models that extend the results of intensive field research to untested locations. Indices that rank the likelihood that nutrient sinks are associated with specific stream reaches.	To identify streamside areas that should be protected or restored to maximize nutrient retention in a watershed. To identify upland locations where the risks of nutrient losses are minimized by streamside nutrient sinks. To target remediation and nutrient removal technologies to upland locations that drain to stream reaches not protected by streamside nutrient buffers.

Table I-4	Research Related to Watershed and Aquifer Vulnerability to DTWS Nutrients

Strategic Focus	Tasks	Products	Uses
Can we determine the nutrient dilution and removal capacities of different types of aquifers? Can we predict the extent of interaction between nutrient laden groundwater from DTWS and the biologically active zones of streamside nutrient sinks?	Develop cost-effective in-situ techniques to determine low rates of nutrient removal within discrete aquifer locations. Develop and test indicator-based models of the fate of nutrients in aquifer and watershed different characteristics (soils, physiography, aquifer features, geochemistry) and compare the results to more explicit models. Gather available information on the fate of nutrient plumes in different types of aquifers.	Data on nutrient removal and nutrient dilution capacities of different types of aquifers. Indices that rank the vulnerability of different aquifers and watersheds to nutrient inputs from DTWS.	To target micro-scale investigations and management efforts to existing systems located on high risk aquifers and watersheds. To target community based strategies that seek to minimize the effects of new DTWS on vulnerable aquifers and watersheds.

Peer Reviews

The preceding White Paper, *Research Needs in Decentralized Wastewater Treatment and Management: Fate and Transport of Nutrients*, by A.J. Gold and J.T. Sims was solicited for peer review. Reviewers comments are provided in this section.

R. B. Reneau, Jr.
Virginia Tech

This is an excellent document and the authors are to be commended on a detailed assessment of the fate and transport of nitrogen (N) and phosphorus (P) from decentralized wastewater treatment systems (DWTS) and the subsequent risk to human and ecosystem health. The use of micro- and macro-level spatial scales is particularly applicable to fate and transport of nutrients since siting, design, and management of DWTS are important factors in controlling risk.

The review and interpretation of data, particularly for conventional systems, is very comprehensive. The authors have identified several significant knowledge gaps where information is needed to properly assess environmental and human risk with increased use of DWTS. This is particularly true for alternative DWTS.

Each of the five key research priorities listed in the abstract is an area where our knowledge with respect to nutrient fate and transport is incomplete. Research in these priority areas will improve our ability to assess and manage risks from DWTS.

Assessment and Analysis of Adverse Effects from Nutrients in Onsite Systems

1. Additional information on the fate of N and P for several of the alternative systems listed in this section would assist in identifying research needs.

   a. Wetland based systems can be used as an example of the need for further review of alternative systems. General information is given concerning N cycling in these systems, but no references are included. There are several very comprehensive references that deal both with the fate of N and P in wetland systems and the design of wetland systems for nutrient reduction. One example is the book by Kadlec and Knight (1996) entitled “Treatment Wetlands.”

   b. Several articles are referenced for aerobic filters and tanks. More information on concentration ranges and removal of N and P in these systems as influenced by system type, climatic conditions, and time in operation would be very beneficial.

2. There are several places where references or additional references would benefit the reader.

   a. An example can be found on page 4. “Increased N inputs from watersheds have been implicated in the degradation of estuarine and marine ecosystems across the U.S. (Nixon et al., 1986), including shallow New England estuaries (Valiela, 1990; Lee and Olsen, 1985); the Chesapeake Bay and Delaware’s Inland Bays; the Gulf of Mexico (Rabalais et al., 1996);

and the Puget Sound (Inkpen and Embrey, 1998).” I was particularly interested in the reference for the Chesapeake Bay and Delaware’s Inland Bays.

b. Another example can be found on page 5. “Through settling and periodic pumping of septage, septic tanks can remove approximately 5-10% of the incoming N.” Several references are available that address this topic. Pell and Nyberg (1989) reported an average loss of 17% while Lakk (1982) estimated that about 10% were removed.

3. On page 7 the authors state that, “Conventional drainfields are usually designed to place effluent below the root zone of most plants, primarily to minimize hydraulic failure in distribution pipes due to root growth and clogging.” In my experience, subsurface absorption systems are not deliberately placed below the root zone. Normally when DWTS are placed deeper in the soil profile it is to maintain elevation differences for waste to move via gravity through all the system components or where more favorable soil properties are present at a deeper depth.

Research Priorities – Micro-Scale Exposure Assessment

The authors have done a superb job of interpretation of the information presented and formulating research priorities.

Common Field Research Methods to Characterize N Dynamics

The ability to construct long term mass balances are essential to further our understanding of risks posed by conventional systems and the risk reduction that can be achieved through alternative systems. Perhaps employing common goals would be more descriptive. The same research methods may not be applicable to all situations. Also, improved research capabilities and techniques will hopefully be developed with experience.

^{15}N can be an extremely useful tool in studying the fate of N in different settings and with different alternative systems. ^{15}N techniques offer a means for testing existing concepts and developing new and sound principles for assessing and managing risk. However, use of ^{15}N as a tracer has a unique set of questions that need to be addressed prior to its use to collect long term mass balance information for DWTS. These questions are related to the interchange of N between the soil inorganic and organic pools. This occurs because mineralization and immobilization occur simultaneously and organisms do not discriminate between ^{15}N and ^{14}N. It is possible that data collected with ^{15}N techniques, at some sites, may potentially have as much or perhaps more error than conventional mass balance procedures.

Geochemical Modeling of P Dynamics

I agree with the authors that such modeling effort should be focused on sites with the greatest risk benefits. However, these models should be tested against field data from a number of sites prior to implementation.

We should also determine if such modeling will improve our ability to make risk management decisions based on P compared to a simpler process based on soil and site characteristics. Such a simple process might consist of factors such as the P sorption maximum for the soil, soil volume available for sorption, characteristics of the effluent, and effluent distribution system.

Nutrient Removal and Site Characteristics

An extremely important area of research. Prior to conducting long-term mass balance studies using tracers, perhaps existing field research data could be evaluated more rigorously using a risk based approach similar to the one proposed by Hoover et al. (1998). Evaluation of existing data in varying risk scenarios might identify soil and site characteristics where N dynamics need to be characterized in greater detail as well as sites where research is adequate for risk management.

I agree with the authors that more information is needed on fate of N, particularly the potential to enhance denitrification where highly treated wastewater is applied to soil. There is certainly the potential for operating these systems in such a manner that denitrification losses may be increased.

Site Indices of Nutrient Retention

Perhaps the research area with the highest priority. However, development of site indices will be dependent on the success of the Nutrient Removal and Site Characteristics research.

Alternative Systems

Alternative DWTS will probably be used more routinely at sites that pose the greatest risk to humans and ecosystems. Such sites include high density housing, soils with inadequate renovation capacity, and proximity to vulnerable receiving environments. The single greatest challenge in managing risk at these and other sites may be the fate of nutrients, particularly N.

I agree with the authors that there is a need to fill information gaps with respect to the fate and transport of N and P. Because of these gaps perhaps a more intensive review of literature on alternative systems is needed prior to identifying specific research priorities.

Additional review of alternative systems should include systems used to produce highly treated effluent and systems used to control distribution of these effluents in soil. If an adequate database is not available for the range in N and P concentrations from these systems over varying operating conditions, then information should be assembled to answer these questions. Systems such as recirculating media filters, peat based filters, wetlands, sequencing batch reactors, aerobic package plants, and others are being used more frequently to treat wastewater from both individual homes and clusters.

Also, there is increased use of distribution systems such as drip irrigation that afford better control over rate and timing of effluent application than can be achieved with the more commonly used distribution systems. Control of factors such as where effluent is placed in the profile and rate, duration, and uniformity of application of effluent will impact the fate of N and P and subsequently the ability to manage risk. Most of these systems will apply wastewater that has undergone varying levels of treatment. There is currently research being conducted with some of these systems. Results from these studies might assist in prioritizing research.

Research Priorities – Macro-Scale Exposure Assessment

The integration of macro-scale information into research priorities by the authors is excellent. However, Total Maximum Daily Load implementation may be seriously hampered by the difficulty in determining N sources using the proposed methodologies.

An additional research priority for consideration is:

Explore the potential for identifying N from DWTS based on the ratio of the stable isotopes of N. The key word is ***explore***.

This suggested research priority is based on the following information.

Research by William Showers (Doll, 1997) suggests the ratio of the stable isotopes, ^{15}N and ^{14}N, can be used to trace N sources in the Neuse River Watershed. He has observed that N originating from municipal wastewater treatment plants, fertilizer producers, swine and poultry waste facilities, urban storm water runoff, and crop runoff each have different isotopic ratios. Since many N sources in a watershed can be identified by their stable isotope ratio, this suggests that DWTS may also have a unique isotopic ratio or signal.

Riparian Zones

Pavel et al. (1996) has conducted research that might benefit the authors in their evaluation of N removal in selected riparian wetlands.

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W.D. Robinson
Research Associate Professor
University of Waterloo

The White Paper entitled “Research needs in decentralized wastewater treatment and management: Fate and transport of nutrients” (Gold and Sims, 2000) reviews our current understanding of the processes that govern nutrient (N & P) migration from septic systems (DWTS) and provides suggestions for future research focus. The paper firstly summarizes the health and environmental risks associated with excess anthropogenic loading of nitrogen and phosphorus; nitrogen primarily as a risk to drinking water supplies and phosphorus as a risk to surface water quality. The paper then discusses the loading potential of DWTSs in relation to other sources such as atmospheric deposition and agricultural practices. Transport mechanisms are discussed with an emphasis on sinks or “hot spots” that may naturally attenuate sewage-derived nutrients. Alternative technologies for enhancing nutrient removal by engineered means are also reviewed. Finally, a set of research priorities are put forward to enable more effective risk management of DWTSs including: 1) development of standardized testing protocols, 2) continued model development at the microscale, 3) continued development of alternative technologies for enhanced nutrient removal, 4) field investigations that focus on sink hotspots, and 5) development of risk models that incorporate site indexing approaches.

In general, this is a very comprehensive document that displays a thorough understanding of the important issues associated with nutrient loading from septic systems. It provides an unusually broad review of the topic, complete with an extensive and up-to-date reference list. Moreover, it is well written using easy to understand language, and as such, it should be a valuable reference document for a wide range of practitioners from engineers and scientists to regulatory personnel.

Comments on suggested research priorities:

1. **Hot Spots.** As the authors suggest, there is now substantial evidence that specific zones exist where much more intensive transformation/attenuation of nutrients occur. Examples include the vadose zone sediments within 1-2 m of the tile lines for P and riparian/stream bed discharge zones for both N and P. Ongoing research should certainly focus on these areas.

2. **Site Indexing.** The authors suggest that site by site assessment of risk will continue to be difficult due to cost factors, suggesting instead, that risk could be more easily addressed at the regional scale by using indexing schemes that incorporate the important landscape characteristics (e.g. soils, hydrology, sediment type). In general, I agree with this approach as there appear to be a number of easily quantifiable parameters that may be reasonably consistent at the regional (aquifer) scale that substantially affect the risk of nutrient transport. For example, in other than permeable sand and fractured media environments, it is unlikely that P will migrate more than a few meters from the tile lines. In addition, in permeable sands, preliminary evidence now suggests that P is much less mobile in non-calcareous environments where acidic conditions may be encountered (Reneau, 1989; Robertson et al., 1998). For nitrate, it has been shown that aquifers which contain a suitable reservoir of electron donor constituents (e.g. trace quantities of organic carbon, sulfide minerals or ferrous iron), may be substantially less at risk from nitrate contamination than other aquifers (Kolle et al., 1985; Pedersen et al., 1991; Postma et al., 1991). Properties such as sediment

grain size, carbonate mineral content and electron donor content, are all characteristics that could be readily incorporated into site indices at the regional or local scale.

3. **Standardized Methods.** This is a noble goal but would likely be difficult to initiate considering the diverse background of the many stakeholders involved with DWTSs. However, standardization with respect to certain parameters such as sediment and soil properties, which are easily measured and comprehended by persons with a variety of backgrounds, would perhaps be possible.

4. **Continued Development of Alternative Technologies.** I agree that alternative systems hold considerable promise for achieving improved “at source” nutrient removal and that use of such systems should be promoted in high-risk landscapes. I find it curious however, that relatively simple and low cost alternative systems have now been available for more than a decade (e.g. “red mud” system for P removal, Chowdery, 1975; and Ruck system for N removal, Laak, 1981) and yet I am not aware of any widespread implementation of these systems. Obviously, the appropriate incentive structure has not yet been put in place to promote their use and further development.

Minor Editorial Comments:

Missing references; P. 7 Parking, 1987; Jacinthe et al., 1998; p. 9, Ball, 1995; p. 19, Gilbert, 1993; Magdoff et al., 1974; Lance, 1977.

Also include Laak (1981) in reference to Ruck system p. 8; include additional references for wetland systems, p. 9; and sulfide based denitrification p. 17 (i.e. Kolle et al., 1985; Pedersen et al., 1991; Postma et al., 1991).

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Economics of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits

A white paper for the
National Decentralized Water Resources
Capacity Development Project

by
Carl Etnier, Agricultural University of Norway
Valerie Nelson, Coalition for Alternative Wastewater Treatment
Richard Pinkham, Rocky Mountain Institute

August 6, 2000

Economics of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits

Carl Etnier,[1] Valerie Nelson,[2] and Richard Pinkham[3]

August 6, 2000

ABSTRACT

This paper describes important direct and indirect costs and benefits to be considered in decision making about decentralized wastewater treatment, as well as decision-making structures which in the future would integrate public health, environmental, engineering, and socioeconomic risks and benefits. Its purpose is to identify and prioritize research gaps in these areas: estimation of direct and indirect socioeconomic costs and benefits associated with various risk-reduction strategies; models and methods of risk assessment and decision making; and risk management options at the individual home or community level that are practical, politically acceptable, and cost effective. Thirty three possible research projects are identified as important to decision making for decentralized wastewater treatment. Six of these are prioritized, based on their anticipated usefulness in providing basic information to the field, information to assess new directions the field is taking, and information to overcome existing obstacles to decentralized treatment. The six prioritized research projects separately address:

- the importance of national performance standards for the diffusion of decentralized wastewater treatment technology;
- hydrological impacts, and their associated economic implications, of wastewater treatment choices;
- lifespans, failure rates, and risks associated with decentralized and centralized solutions;
- economies and diseconomies of scale in different types of wastewater systems;
- cost effectiveness of management systems, including performance-based codes; and
- compatibility of decentralized treatment with smart growth.

[1] Department of Agricultural Engineering, Agricultural University of Norway, Ås, Norway (carl.etnier@itf.nlh.no).

[2] Coalition for Alternative Wastewater Treatment, Gloucester, Massachusetts (valerie508@aol.com).

[3] Rocky Mountain Institute, Snowmass, Colorado (rpinkham@rmi.org).

CONTENTS

1 INTRODUCTION

Domestic wastewater is treated to reduce risks to health and the environment, as well as to manage natural resources. Health risks are primarily the spread of water-borne diseases or toxins. Risks to the environment are primarily those of eutrophication, along with toxicity to aquatic organisms. Resource management issues entail long-term maintenance of (primarily) agricultural productivity through replacement of nutrients removed. While the resource management aspect has been little emphasized in North America and Europe in the last century, it is regaining significance.

Costs of wastewater treatment decisions include direct financial expenses incurred for treatment system components and management. Costs may also be indirect, in the form of financial expenses incurred as a result of wastewater treatment decisions, and non-monetary, like ecosystem degradation. Costs include initial capital costs and long-term operating and maintenance costs. **Benefits** of wastewater treatment are primarily non-monetary, like health and environmental protection, though some benefits can more easily be measured in monetary terms, like avoided costs of water supply when treated wastewater is used for irrigation.

The costs are borne by and benefits accrue to different actors, who may be the property owner, the wastewater utility, or other groups in society. A crucial part of fair risk management decisions is understanding who pays the costs and who reaps the benefits of a given decision.

This white paper addresses the issues of costs and benefits of decentralized wastewater treatment at the micro and macro scales. By decentralized wastewater treatment, we mean treatment "at or near the source" of the wastewater (Crites and Tchobanoglous 1998). We give an overview of what the main issues are in each of these areas, what is fairly well understood, and what needs more development.

<u>Micro scale costs and benefits</u>

The micro scale is the level of the individual property, most often a single-family house. The costs include such things as the direct costs of installing, operating, and maintaining a wastewater treatment system, and the indirect costs of homeowner inconvenience or discontent with the system. Benefits include increased property values, protection of wells and prevention of surfacing or backup of effluent into the home (with the associated onsite health issues), opportunities associated with reuse of wastewater onsite or nearby, and off-site public health and environmental risk reduction (externalities). The ability to provide good wastewater treatment can also be crucial to obtaining a building permit for a site, so the option value of building is also a benefit in such cases.

<u>Macro scale costs and benefits</u>

The macro scale is a larger area, such as a watershed or a community. Costs at the macro scale include the direct, monetary costs of building and maintaining systems. Benefits can include economic benefits from increased commercial, industrial, and residential development, pollution reduction, help in enforcing development plans, maintenance of local vegetation and wildlife, nutrient recycling to agriculture, public health, and many more.

Economic costs function as a broad barrier or inhibitor to more aggressive risk reduction measures at both the micro and macro scale. Homeowners will almost always choose the least-cost, least intrusive onsite system permitted by the regulations (Sandison et al.1997), and regulators are constrained by political opposition to tighter regulations which would cost more money to, or require more oversight of, large numbers of homeowners. At the macro level, many small communities suffer with substandard onsite systems, because the costs of onsite upgrades or sewering are prohibitive. On the other hand, some communities choose to install sewers rather than upgrade onsite systems at a lower financial cost, since the non-monetary benefits and increased property values from sewers are perceived to be significant.

From a societal perspective, a key question is whether the current regulations and practices represent an appropriate balancing and distribution of benefits and costs. Risk-based assessments should lead to a more efficient allocation of economic and social resources over time. The following types of questions are important to various decision makers:

- What improvements in health, the environment, and resource management are to be expected from greater investment in improved domestic wastewater treatment?
- Would resources spent by homeowners, communities, or the nation on improved decentralized treatment technologies and management provide greater improvements to health, the environment, and resource management than resources spent on sewer systems, stormwater remediation, agricultural waste treatment, drinking water filtration systems, or, for that matter, on transportation safety programs, public health programs, etc?
- What wastewater treatment investments are likely to give the greatest benefits, and to whom?

Some more specific questions include:

- Are there pre-treatment technologies, management protocols, or better use of soils treatment which are more cost effective than conventional septic systems and which might be utilized more widely?
- Could revisions in onsite codes lead to more cost-effective risk reduction, in particular from a shift either to a performance code or to higher treatment effluent standards in sensitive areas?
- What are the financial risks of building centralized versus decentralized systems when there is uncertainty in the geographical distribution and amounts of growth expected or the environmental risks of treatment decisions?
- What financial and environmental benefits are achieved by reusing treated domestic wastewater near where it is generated, either in the house or on the grounds?

Finally, it is important to consider the political feasibility of decisions. It might not be possible to muster the political will to carry out the decision which is theoretically best, and the way that decision is made may significantly affect whether there is sufficient political will to carry it out. Decentralized wastewater treatment involves a large number of small actors. It is generally easier to make changes in treatment technologies or management when homeowners and other stakeholders understand what problems there are with the present system and are meaningfully involved in deciding which options are chosen for solving those problems.

Relationship to the other white papers

The other topic-specific white papers presented at this conference address public health and environmental risks. This paper addresses the range of other considerations that Dan Jones (2000) mentions as important in his overview paper for the National Decentralized Water Resources Capacity Development Project on integrated risk assessment and risk management—a range including both direct economic costs and other socioeconomic aspects of decisions, like water reuse and community values.

2 COSTS AND BENEFITS AT THE MICRO SCALE

Costs at the micro, or household, scale include design and installation costs, long-term operation and maintenance costs, and possible inconveniences to homeowners. In addition to sanitation and environmental protection, direct and indirect benefits may include the value of retaining the water onsite, and the possibility of development.

This section of the paper addresses only those benefits and costs pertaining to individual properties. Of course, many costs and benefits at this level depend on decisions made by the community, and individual systems affect costs and benefits experienced beyond individual property lines. Those interrelationships are treated more fully in the macro section.

2.1 Direct costs

2.1.1 Design and installation costs

A number of studies suggest an average cost of $2,500-3,500 for conventional septic systems, with costs in the southern states as low as $1,500 and costs in more urbanized East Coast and West Coast states as high as $6,000-8,000 or more. Also, costs can vary tremendously depending on the site conditions, with factors like steep slopes and a need to pump wastewater uphill increasing costs, sometimes dramatically. Costs of advanced or alternative technologies, such as mound systems, intermittent or recirculating sand filters, peat or other media filters, aerobic treatment units, disinfection, etc. are typically $5,000-10,000 higher (Chaffee 1999; Sandison et al.1997; Hoover 1997; Nelson et al. 2000). These costs are easily quantified by engineers who contact equipment manufacturers and are familiar with local construction costs. However, costs may be artificially high if contractors charge more for installing new types of systems or if they engage in price gouging in more affluent areas.

2.1.2 Long-term costs

The proper cost figures for comparing onsite systems are from present value calculations of initial installation costs plus long-term maintenance and repair costs. The U.S. EPA generally suggests that planners use a twenty-year time horizon. Costs may include planned expenditures for septic tank pumping, inspections, electricity, and replacement of sand or other filter media, and unplanned repairs or replacement due to failure of the system.

Operating, maintenance and repair costs are a significant portion of the present value figures (in some cases such as aerobic treatment units with air pumps, they exceed the initial installation costs), but estimates vary widely among onsite experts (Sandison et al.1997; Hoover 1997).

2.1.3 Tools

North Carolina State University has developed a predictive cost-estimation model COSMO (Costs of Onsite Management Options), which can predict costs for 46 different combinations of pre-treatment options, distribution technologies, and other units which fit together in a treatment train, with varying loading rates, design flows, etc. Costs of equipment, materials, labor, etc. can be varied with local conditions and the characteristics of the site (Hoover 1997). The COSMO model includes precise schedules for various maintenance and planned replacements (e.g., media in filters). It is useful to policy makers, individuals, contractors, and system designers.

2.2 Effect of long-run performance on costs

Major uncertainties exist about long-run performance and, therefore, about total cost projections for various systems. From an engineering risk management perspective, an onsite system designer should have the following information:

- likely scenarios and time frames for failure, or breakdown in treatment capacity, for each of the technologies and/or designs being considered;
- routine maintenance schedules that would lower the risks of such failures in treatment of each system;
- costs of major repairs or replacement when failures do occur and costs of routine maintenance to prevent such failures;
- costs of installing and maintaining remote monitoring systems and benefits of information collected from them; and
- value (reduced failure risk vs. costs) of preventive, planned replacement of pumps, filter media, and other components.

It is generally understood that all onsite systems, including conventional septic systems, typically require only routine maintenance, such as septic tank pumping, but advanced treatment systems are more prone to failure. The dominant risk management policy currently is the inspection by the county health agent or other regulatory personnel at the time the system is installed. Since many failures are attributed to flaws in system design and installation, this approach is undoubtedly cost effective. In most states, ATUs and some other units also come with mandatory two-year inspection and maintenance requirements, again with the understanding that many of the failures occur within the first two years. Some communities and some states require long-term periodic inspection and maintenance programs for some systems, including the time of real estate transfer inspection approach. The preponderant long-run responsibility for inspection and maintenance, however, continues to rest with the homeowner.

Given the high costs of long-term professional (non-homeowner) inspection and maintenance programs, the following research questions are particularly crucial:

Research project: Failure rates and long-term performance. Good estimates of failure rates and/or long-run performance of various technologies or designs are not available, nor are causes of failures well-documented, in particular for technologies developed in the last five years. Minimally maintained advanced pre-treatment units generally have twice the failure rate of minimally maintained conventional septic systems (Gunn 1998; Ingram et al. 1994; Water Resources Research Institute News 1992; Nelson et al. 2000). However, definitions of failure differ among various studies. Widely accepted definitions of different types of failure plus a systematic study of long-term performance with respect to these failure types would have great value to the onsite designer.

Research project: Benefits and costs of mandated inspection programs. How much would long-term professional inspection and maintenance programs reduce risks to public health and the environment, and at what cost both in dollars and in alienation of homeowners? (See 2.3.1, below.) Conventional designs are conservative and able to withstand a certain amount of abuse and neglect. Homeowners typically repair hydraulic problems leading to sewage back-up or surfacing. However, hydraulic failure is not always a sufficient indicator of system performance; a system may not adequately treat wastewater indefinitely. Analysis of systems that are near sensitive surface waters or ground water, or that have mechanical and electrical components prone to failure, will likely show that the benefits of professional oversight exceed the costs.

Research project: Reliability and effectiveness of remote monitoring technologies to reduce engineering risks of treatment failure. Remote monitoring and control has the potential to reduce both the risks of treatment failure and the socioeconomic costs of professional maintenance, but technologies and approaches need to be examined (Tchobanoglous 1999). Anecdotal evidence indicates that remote monitoring can reduce the maintenance and repair costs for some systems, but what needs to be monitored and what system types give net returns on the investment in monitoring technology? And can monitoring reduce the political resistance of homeowners to maintenance programs? For remote monitoring uses which are not currently cost effective, how much would the cost of remote monitoring have to decrease for them to become cost effective?

2.3 Non-monetary and indirect costs and benefits

2.3.1 Convenience, intrusion, and other homeowner concerns

Non-monetary costs of various onsite technologies or designs and of professional inspection and maintenance programs are substantial. Various case studies (Sandison et al. 1997; Nelson 1999; Washington State Department of Health and Puget Sound Water Quality Authority 1996; Herring 1996; Dillman 1999; Wayland Wastewater Management Committee 1995; Otis et al. 1981; Uhren 1991; Ingram et al. 1994; Piluk 1998; Jantrania et al.1998; Nelson et al. 2000) indicate that the following factors are critical in the acceptability of onsite systems and management to homeowners:

- difficulty of and time spent on maintenance;
- inconveniences of limitations on garbage disposals, washing machines, etc.;
- opposition to new public bureaucracies;
- private property rights and resistance to intrusion of outsiders into backyards;
- aesthetic concerns about mounds, risers, etc.;
- equity issues, or unwillingness to pay for someone else's system; and
- voluntary risk vs. involuntary risk—generally people tolerate higher voluntary risks, or those perceived to be under their own control.

One quantification of these attitudes is found in an Infiltrator Systems survey in Massachusetts. Sixty-one percent of respondents preferred homeowner responsibility for installation, maintenance, and repairs of the septic system, compared to 34% in favor of a professionally managed and guaranteed or insured septic system with a standard monthly fee (Infiltrator Systems 1999). The survey also showed that homeowners believed that homes connected to sewers would be worth about $9,000 more than homes on individual septic systems.

Research project: Documentation of non-monetary costs. Additional surveys of homeowner preferences would be useful. How much would homeowners be willing to pay to avoid limitations on their water use, garbage disposals, etc. and to avoid having regulators or other maintenance personnel on their property? Answers will be very site specific, so it is also useful to consider how to regularly incorporate such surveys into the planning process, and how the education and planning process can be structured to motivate people to accept the ways onsite systems diverge from so-called "flush and forget" sewers. A second research approach would be to examine housing markets through the method of hedonistic pricing. How much lower are housing prices, all else being equal, for homes with frequent inspections and limitations on quality of life compared to homes with no outside maintenance requirements? What contributes to making this difference higher in some places than others?

Research project: Effective education campaigns. Various studies suggest that when a community has a valued resource, such as a lake, drinking water supply, or estuary, homeowners are more willing to spend money and accept mandated long-term maintenance programs (Eddy 1992; Eddy 1993; Breisemeister 1996). Training centers and other programs have developed onsite system educational campaigns. A study of the effectiveness of educational campaigns in changing behavior would be useful. In addition, advertising campaigns for other public benefits have lessons for wastewater planners (Earle 2000).

2.3.2 Effects on property values

Concerns over ground water contamination often lead to large-lot zoning in unsewered areas, with 2-5 acre lots not uncommon. Higher value accrues to the owner of undeveloped land (and the relevant taxing authority) if it can be developed more densely. This is indicated both by the literature on real estate value (Wentling et al. 1988; Boykin and Ring 1993) and the practical experience of a real estate agent contacted (Anderson 2000). The question of permitted density is as much a political question as a technical question; it may be that adequate protection to the public health and the environment would be provided by conventional onsite systems at higher development densities than is permitted in many areas. Where that is not the case, any wastewater treatment technology which makes denser development acceptable can increase land value.

Politically accepted wastewater treatment can make the difference between whether a property may be built on at all, or even whether an existing house must be destroyed. In Massachusetts, the local authorities prescribed that a house on an eight acre lot must be destroyed at the time of land sale, because the soil's hydraulic gradient connected with a nearby town's well field. An advanced onsite system with blackwater composting and a recycle loop for the graywater satisfied the authorities, who allowed the lot to be sold with the house intact (Schönborn 2000).

While small-lot zoning realizes greater value per acre developed than large-lot zoning, the cost per lot is less, all other things being equal. This contributes to making the homes more affordable.

2.3.3 Value of reusing water onsite

Many decentralized wastewater treatment systems provide potential for reuse of water at the individual property or neighborhood scale. While graywater reuse is certainly much more common than blackwater reuse, advanced decentralized wastewater treatment systems make blackwater reuse a viable opportunity under proper conditions. Typically, treated effluent is used to irrigate vegetation or support landscape water features such as ponds and wetlands. Doing so may involve various monetary and non-monetary costs, and provides a variety of benefits.

Costs in addition to the decentralized treatment system may include storage tanks or ponds; distribution lines; drip emitters, soaker hoses, sprinkler heads, or other irrigation devices; pumps; permits; disinfection; signs; fencing; operational costs such as pumping energy; cleaning and other maintenance of system components; and monitoring and supervision. These costs are largely well-understood and easily quantified by engineers familiar with the relevant technologies and local conditions. Frequently these costs can be co-mingled with, or are already borne by other systems; for instance, irrigation systems supplied by conventional water sources, rainwater collection, or stormwater detention. For some decentralized wastewater systems, effluent dispersal via irrigation may be integral to final treatment or effluent disposal and thus a core cost rather than an add-on cost.

Benefits may include: avoided purchases of water from local water providers; avoided pumping costs for self-supplied systems; avoided water treatment costs for self-supplied systems; development of a higher-value landscape where water is costly or in short supply, with concomitant increases in property values plus intangible benefits to property owners; maintenance of landscape during drought periods when irrigation with first-use water is restricted, including avoided costs of replacing plantings that would otherwise die; and fire-fighting supply for systems with storage capacity, which may lead to reduced fire insurance premiums. Current understanding of and approaches to valuing each of these potential benefits ranges from straightforward to hazy. For instance, avoided water supply costs are easily quantified for local conditions.[1] Some other benefits are harder to quantify and rarely evaluated.

Research project: Data and guidelines for valuing water reuse. Guidelines for estimating and including hard-to-value benefits when costing-out treatment system alternatives would be useful to system designers, and their application would improve the economic attractiveness of advanced decentralized wastewater treatment systems. Property value improvements could be estimated with real estate appraisal techniques and hedonistic valuation approaches. Reductions in fire insurance premiums have probably been applied for rainwater collection systems and could be similarly applied for wastewater treatment effluent storage.

Research project: Risk-based guidelines for appropriate reuse conditions. Risk assessments to define the risks of reuse of treated blackwater under various physical conditions, including soil parameters and physical configurations such as surface vs. subsurface irrigation are needed. This will enable development of appropriate regulatory responses, and comparison of the economic values of reuse with any risks involved.

2.4 Incorporating monetary and non-monetary costs and benefits into an onsite system risk-based design.

While much of the emphasis on onsite system design has historically focussed on system performance and risk reduction, economic factors and homeowner preferences in practice are also of major concern from a public policy perspective. The Washington State and Rhode Island training centers have installed a wide range of conventional and advanced treatment systems, in particular in remedial situations with non-conforming lots, and have been incorporating cost and homeowner preferences into their design process. Anish Jantrania has also been attempting to lower costs for advanced treatment installations in Virginia. For example, strict requirements in the onsite code may be reduced somewhat if costs are prohibitive.

Research project: Guidelines for and effects of code variances. These efforts should be examined and formalized in guidance documents, which also document the additional risks, if any, these exemptions entail.

2.5 Possible developments and their effects

In general, the conventional septic system with homeowner maintenance appears to be a highly effective mode of protecting public health and the environment where soil and site conditions meet the requirements of the state or county codes. Advanced treatment systems are substantially higher in monetary and non-monetary costs (inconvenience and intrusiveness) to the homeowner, and may be justified only in instances where risks to public health and the environment are higher. However, clear data are lacking. In the absence of hydraulic failure, conventional septic systems may cause pollution which is undocumented and unregulated. Even hydraulic failures are not non-compliant in some jurisdictions, and many jurisdictions have no way of detecting hydraulic failures other than homeowner self-regulation.

However, if markets for advanced systems are broadened and standardized, existing firms may expand dramatically, or large firms may enter the field, and the treatment performance of systems may increase substantially, while the costs of such systems may fall dramatically, and approach more closely the costs of a conventional system. Costs of remote monitoring may also fall significantly, so that intrusiveness on private property can be minimized. Wide production and distribution networks could create substantial economies of scale. Tighter regulation of pollution from onsite systems could also greatly increase the demand for improved technology.

Factors in the market which may forestall such technology developments and cost reductions include: lack of complementary products or services (such as onsite construction companies or management entities); lack of marketing and distribution channels; absence of agreement on "standards" or "rules of the game"; fragmentation in regulatory requirements; superior substitute products; and defensive responses by threatened competitors (Clerico 1998; Lindell 1997; Ball et al. 1997; Porter 1980; Nelson et al. 2000).

Research project: Cost structures and leverage points for advances in technology. A systematic study of possible future cost structures would be useful. What are the various cost profiles of typical treatment systems, including: design, installation, tanks, pumps, pipes, transportation, and profit? How much would each of these factors be reduced with mass production and distribution? If costs could be reduced substantially, what steps should be taken to encourage such a market evolution to occur?

2.6 Critical benefit and cost analyses at the micro scale

Risk-based decision making at the micro level would include estimates of risk reduction to public health and the environment, along with consideration of the homeowner's monetary and non-monetary costs and benefits, plus other impacts off the lot (externalities). A comprehensive risk assessment/risk management study for a prototypical lot would include:

- health risks to the family and beyond the property line, from surfacing or backup, well contamination, ground water contamination which could affect off-site drinking water supplies, shellfish beds, recreational waters, etc.;
- risks to the environment, including largely off-site externalities like eutrophication of lakes or estuaries, increase in storm flows, saltwater intrusion in ground water, mortality for plant and animal species, etc.;
- installation and long-term maintenance costs of various pre-treatment or soil-based system designs or equipment;
- homeowner preferences for various pre-treatment or soil-based systems and maintenance programs (based on convenience and/or intrusiveness issues);
- possible value of wastewater reuse to the homeowner;
- likelihood of system dysfunction (including homeowner misuse) and risk to public health or the environment from such failures; and
- costs and benefits of remote monitoring or other methods of early detection of incipient treatment failure.

An assessment would weigh all such factors with respect to the most promising options identified. A risk-based decision for a particular lot may be made for the system which has the highest benefit-cost ratio, where benefits include public health or environmental risk reduction and possible wastewater reuse or property value improvements, and costs include direct monetary costs of installation and maintenance, along with non-monetary costs. However, such calculations require that risk reduction figures be converted to dollars, and this has proven to be difficult in many fields, and even attacked as methodologically illegitimate (e.g., Sagoff 1988; O'Neill 1993).

An alternative decision-making structure would be to define risk reduction targets for various parameters, such as nutrient reduction or pathogen reduction, and then to identify the least-cost method to meet those targets. For example, if a rough estimate of risks suggests a 10-10-10-1 (mg/l BOD_5-TSS-NO_3-P) effluent standard should be met, what is the least costly option to meet that standard? The U.S. EPA has suggested that ground water standards become the basis for risk-reduction strategies. What are the least costly means to meet those standards?

Once the basic model of risk-based decision making is developed, then factors which vary by site can be introduced to see how benefit-cost calculations are affected. For sites which are more vulnerable to risks to public health and the environment, either on the lot or off the lot, then the technology options to manage those risks will change, likely increasing the cost and need for management oversight. Sites may be more vulnerable either because of proximity to wells or other sensitive resource areas, or because of problematic site or soil conditions.

Risk-based decision making at the micro scale is, in all likelihood, not possible without considering macro scale issues. The total risks to public health and the environment are to some degree cumulative. Density of housing and treatment choices for nearby housing units will affect the amount and kind of treatment necessary on a given lot to avoid exceeding a specified level of risk. Furthermore, costs of reducing the total risks a given amount vary greatly from lot to lot, raising questions of how risk-reduction resources should be allocated at community, watershed, and higher levels. Risk analysts and managers should recognize the interrelationships of micro and macro decisions and effects in their studies and action plans.

Research project: Development of a prototype for decision making. Is it feasible to develop a general risk-based decision-making framework for a standard lot and a variety of higher-risk situations? If so, it would be a valuable contribution to the field, because it would provide the structure for examining a wide variety of technology and management options within a common framework. Such a model could help address a number of questions. For instance, what are the costs of and benefits of risk reduction from the installation of specific technologies, such as watertight (vs. non-watertight) septic tanks, effluent filters, washing machine filters, pre-treatment units, shallow drainfields, etc.? How do the benefits vary with the particular site conditions of the lot and/or the other pre-treatment or soil-based systems being utilized in the treatment train? What effluent quality is a cost-effective goal for pre-treatment units and when is disinfection advisable? Under which circumstances would costly professional inspection and maintenance programs be required for homeowners, in terms of engineering, public health, or environmental risk reduction achieved? Under what circumstances are strict, costly requirements for nutrient reduction or disinfection justified?

Research project: Model code development. A risk-based decision-making assessment framework, if developed, could also be utilized to assess various regulatory (risk management) proposals. Specific issues could include separation distances, seasonal high ground water, additional treatment requirements in sensitive areas, etc. Are some requirements more costly to the homeowner, and to society at large, than can be justified by the expected risk reduction achieved? Alternatively, are there provisions which could be introduced into the code, in particular for high-risk lots, which would reduce risks at minimal costs? What provisions of the code would reduce risks at comparable costs for risk reduction from agricultural waste management, stormwater management, transportation safety, public health measures, etc.?

Research project: Performance-based codes. A benefit-cost study of a performance-based code would be a worthwhile exercise. What specific risk reduction benefits could be achieved by adhering to a performance-based analysis of a site in contrast to implementing requirements of a well-designed prescriptive code? Costs of site analysis and design would be substantially higher than for a prescriptive code. Monitoring expenses, either at the point of release into soils, or in particular after soils treatment, would be very high as well. Regulatory oversight costs would increase (Crosby et al. 1998; Smithson 1995; Gunn 1998; Nelson et al. 2000). Are the risks of failure higher when greater discretion is given to the designer? From a societal perspective, widespread implementation of a performance-based approach would require a diversion of skilled manpower into the onsite field and/or a substantial upgrading of skills of personnel already in the field. Would these costs be justified?

Research project: Technological and regulatory standards for market stimulus. Technology-driven markets expand when consistent standards evolve, either through consensus in the field or dominance by one large manufacturer. Standards in the onsite field have the potential to increase homeowner and regulatory acceptance of new technologies, and to create conditions for substantial innovation, mass marketing, and economies of scale (Hoover et al. 1998; Ruskin 1999; Jantrania 1999). The U.S. EPA (1999) has recommended ground water performance standards after soils treatment be the goal, but this approach does not help to achieve consistency in requirements for pre-treatment units. Conversely, effluent standards may not allow for sufficient consideration of additional treatment in the soils. A project to develop a consensus on standards would be an important contribution to technological innovation and market expansion.

3 COSTS AND BENEFITS AT THE MACRO SCALE

3.1 Point of departure

Community wastewater facilities planning has typically included three phases: (1) needs assessment, (2) development and screening of alternatives (particularly regarding problem areas or areas of special concern); and (3) overall, integrated evaluation of alternative plans and their area-specific sub-plans. A final recommendation, for example, would be based on a showing that the selected plan “is the most economical means of meeting the applicable water quality and public health requirements, while recognizing environmental and other non-monetary considerations" (Arenovski and Shephard 1995).

While the facilities planning process is sufficiently flexible and open-ended to include a full range of risk-reduction strategies and criteria, decentralized wastewater experts have argued that in practice the planning process has been excessively narrow in scope. Technologies more applicable to large urban systems have been carelessly recommended to small communities, while more affordable technologies such as community sand filters, pressure collection systems, cluster systems, and remedial onsite upgrades have been given little attention (Kreissl and Otis 2000). Further, inadequate consideration has been given to wastewater reuse, ground water recharge, wellhead protection, and other watershed needs and values.

The facilities planning process has also not typically addressed alternative means to reduce risks of water pollution, including stormwater remediation, repair of leaking sewer pipes, point-source upgrades, improved farming practices, and others. A proper comprehensive water quality protection plan would identify those measures which have the largest impact on reducing risks to public health and the environment at the least cost, and projects could be prioritized over time accordingly. The U.S. EPA's watershed initiatives and Total Maximum Daily Load (TMDL) agreements take this approach. EPA has also encouraged small towns to integrate risk assessment of air quality, solid waste, toxic waste, and other problems, and to move on high-priority, high-impact, cost-effective projects first. The improvements in efficient targeting of risk reduction monies should, however, be compared to the increased costs of planning.

Finally, the needs assessment phase of the project has typically over-simplified both the sources of pollution and the levels of pollution. Often problems such as elevated nitrogen levels in ground water or contaminated shellfish beds are attributed to septic system contamination, without investigation of sources. There are instances where sewers have been constructed, and the improperly diagnosed water quality problems remain. In other instances, insufficient water quality monitoring has been conducted to establish that serious threats to public health or the environment exist (U.S. EPA 1994). The benefits from improved accuracy in determining pollution sources need to be weighed against the costs of investigation.

This section of the paper describes socioeconomic impacts and other benefits at the community or macro level, as well as risk-based decision-making models and management tools which provide the means to conduct a full and comprehensive facilities planning process.

3.1.1 Benefit-cost analysis in the U.S. EPA's *Response to Congress*

The U.S. EPA has recently addressed many macro-level benefits of decentralized wastewater treatment in a report to the House Appropriations Committee (U.S. EPA 1997). The report affirms the ability of decentralized systems to meet federal and state water quality standards in most cases, and pointed to a number of benefits that decentralized treatment provides better than centralized treatment.

First among these benefits is the ability to minimize the impact of transporting water long distances, either in the form of interbasin transfers or movements downstream in the same basin.

Most decentralized systems “inherently include on-lot water reuse and ground water recharge” (U.S. EPA 1997) While most reuse is achieved by whatever vegetation happens to take up the water discharged, treatment for human reuse onsite in industrial, commercial, and even residential settings is practiced. Decentralized systems allow this to occur without the costs of two sets of pipelines: one carrying wastewater away from the site, the other carrying the treated water back to the site.

The report also points to advantages of decentralized systems in ecologically sensitive areas, where ground water recharge is an issue, shellfish beds need protection from effluent, or where the very construction of a centralized infrastructure can be significantly environmentally disruptive.

3.2 Scale issues

3.2.1 Economies and diseconomies of scale in wastewater treatment systems

Many factors affect the economics of choosing between centralized or decentralized wastewater treatment options. This section discusses factors related to system scale that can be addressed via fairly straightforward engineering economics approaches. The following sections consider additional factors in planning, policy, finance, and other areas that do or should affect system choice.

Major scale-dependent cost components for wastewater systems are the capital costs of treatment facilities, capital costs of collection systems, operating and maintenance costs; and administrative and managerial costs. We discuss each cost component below, briefly touching on scale economies and diseconomies for each.

Capital costs of treatment facilities

Treatment facilities have long been thought to enjoy economies of scale—costs decrease per capita as the population served, and thereby the facility size, increases (Hovey et al. 1977, citing Tihansky 1974). While decentralized wastewater treatment units are quite different in configuration, their capital costs, at least for advanced systems, appear to be higher on a per capita basis than those of medium to large-sized conventional plants. One diseconomy in centralized facilities compared to decentralized ones is that they must be sized to allow for infiltration into the sewer lines, a design parameter that can be reduced or avoided in decentralized facilities where infiltration is reduced or eliminated due to smaller diameter pipes, shallower trenching, or use of pressurized sewers (Otis et al. 1981).

Capital costs of collection systems

Collection system economies are highly dependent on land use density; capital costs per connection for systems at typical low suburban development densities are significantly greater than costs at higher densities typical of close-in urban development. (Adams et al. 1972). In part because density typically decreases as service area size increases,[2] collection systems often experience diseconomies of scale—costs increase per capita with increasing system size (Hovey et al. 1977, citing Adams et al. 1972). Because collections systems often account for 70–90 percent of total wastewater system capital costs (Ache 1999), diseconomies of scale in a collection system can offset or overwhelm economies of scale in treatment plant size (Adams et al. 1972). Analyzing the tradeoffs between treatment economies and collection system diseconomies is a key task of regional wastewater facility planning (Whitlatch 1997).

Operating and maintenance costs

Operating and maintenance costs of conventional treatment plants decrease per capita as number of persons served increases (Hovey et al. 1977, citing Tihansky 1974). Reasons include efficiencies in staffing, and bulk purchase of chemicals, replacement parts, and other supplies. Decentralized technologies are significantly different in their treatment plant O&M requirements. These costs are often quite low, though experience shows that decentralized systems may show low O&M costs because O&M schedules and expenditures are woefully inadequate. Collection system O&M is much higher per capita for centralized facilities than decentralized facilities, due to the costs of sewer inspection, cleaning, and repair. In total, Ache (1999) indicates that O&M is thought to be more expensive per capita for centralized systems (except perhaps for very large urban systems) than O&M for decentralized systems, though it is not clear whether this conclusion pertains to recommended or typical real world levels of decentralized system O&M.

Administrative and managerial costs

Management of a wastewater system usually benefits from economies of scale. Many key functions and personnel can almost be viewed as fixed costs, invariant to the size of the system. It is precisely the inability to support these costs that results in the poor performance of many decentralized and small conventional systems, and economies and increased quality in administration and management are a frequent argument for the efficacy of centralized systems. However, ongoing efforts in the decentralized arena to develop institutional models that create economies of scale by consolidating administration and management (and many O&M functions as well) of multiple small and decentralized systems under one management authority should provide significant cost savings and improved management.

Summary

In the authors' view, the prevalent belief in the wastewater management profession is that conventional onsite systems are a cost-effective wastewater treatment solution in rural situations, but are to be replaced with sewers when problems develop or density increases. Other onsite options and cluster systems are typically considered much more expensive on a per capita basis than centralized systems. This belief, and a range of significant administrative, regulatory, cultural and other considerations discussed elsewhere in this paper, supports the usual preference for centralized systems in many small communities and on the fringes of urban areas. Yet this belief about system economics may be significantly flawed. A number of studies show lower costs for advanced onsite and cluster systems in many small community and urban fringe situations (Ache 1999; U.S. EPA 1997).

Following are some additional scale-related considerations in wastewater system planning.

Synergies with other water-related infrastructure

A recent study of the economics of scale for water-related systems in Adelaide, a medium-sized city in South Australia, showed that small-scale systems, averaging perhaps 2,500 connections per wastewater treatment plant, are likely to be cost competitive with much larger plants on the basis of whole system analysis of the wastewater system, and more competitive if the wastewater system is designed to realize synergies with localized stormwater treatment, and local reuse of treated sanitary or storm sewer water (Clark et al. 1997; Clark 1997).[3] Heaney et al. (1999) have also recently posited that neighborhood-scale systems may be optimal for wastewater and integrated wastewater/stormwater systems.

Sewer rollbacks

Within existing urbanized sewer systems, there are opportunities to develop new satellite treatment facilities with or without local wastewater reuse, that have the benefits of reducing over-capacity in sewer lines and which reduce point-source discharges from existing treatment plants. In Mobile, Alabama a national demonstration project will involve removal of wastewater from an existing sewer line, treatment in several parallel cluster-size filter units, and reuse to irrigate a new recreational park.

Cluster systems

Newly-emerging cluster system technologies increasingly appear to be cost effective in high-risk areas, where conventional septic systems are not adequate. If the choice is to install individual pre-treatment units, pressure-dosed drainfields with pumps, and other advanced technologies at each individual home, or instead to cluster treatment at a nearby site for a group of homes, a cluster system has a number of advantages. Typically, a septic tank and pump would be installed at each lot, along with a pressure collection system for liquids. The savings would be from dividing pre-treatment unit and maintenance and monitoring costs among several or many homes. Land costs or unavailability of land for a cluster system could eliminate this advantage (Lindell 1997). Cluster systems appear promising in many ways, though a number of uncertainties remain. How should such systems be regulated? Will consumers accept them as an alternative to self-controlled onsite systems or conventional sewer systems?

Research project: Cost components across scale. It would be a very useful contribution to the literature and to developing practice in the profession to have a thorough compilation of the cost components of centralized and decentralized (onsite and cluster) systems, illustrating cost ranges across system scale and across important situational parameters, and with a review of which components are most ripe for cost savings given likely and reasonably expected improvements in technology or institutions.

3.2.2 Resilience and failure rates

Historically, user fees have not provided for the inevitable major repairs or replacement of centralized systems. An entire generation of Clean Water Act-financed central treatment plants and many collection systems are now reaching the end of their useful life, or facing huge backlogs in unperformed maintenance (Allbee 2000). The risks are sewer line infiltration; sewer overflows; contamination of drinking water supplies, lakes, streams, and estuaries; violations of point-source discharges to surface waters; and other problems. At the same time, many onsite systems are failing.

Research project: Infrastructure lifespans and associated risks. Documentation of the actual lifespan of different wastewater infrastructure solutions and the effects and risks associated with their deterioration would be an important contribution. Facilities planning generally considers a twenty-year lifespan for projects. It would be useful to document the actual lifespans of both decentralized and centralized systems, so that proper planning for their replacement could be made.

There is some evidence that leaking sewer pipes are a much greater source of drinking water and surface water contamination across the country than are septic systems (Gerba 1998), presumably because sewer lines have tended to be located in streambeds, or are deeper below ground and more likely to affect ground water, and provide no septic tank-like treatment.

Research project: Comparative analysis of risks of system failures. A comparative analysis of the risks associated with routine failures would be useful. When a large sewer system and treatment plant flood or otherwise breakdown, huge volumes of wastewater may be released to nearby surface waters. How do these risks compare to flooding/saturation of onsite leaching fields and to continuous breakdown of a small percentage of onsite systems scattered throughout the community?

Research project: Manageability of failures. A properly managed wastewater utility would seek to prevent and/or repair breakdowns as soon as possible. What are the comparative costs of maintenance, the relative risks of breakdown, the costs and timing of major system repair, the likelihood that major repairs would be financed, and the relative risks from inaction for different wastewater treatment systems—onsite, cluster, and centralized? Also, while budgeting and rate-making techniques to provide adequate funds for maintenance, repair, and replacement are relatively straightforward, many wastewater management entities do not use them. Why not, and how could implementation of adequate financial structures be increased?

3.2.3 Positive and negative externalities

3.2.3.1 Compatibility with "smart growth" objectives

As mentioned above, some homeowners are willing to pay substantially more for the conveniences, non-intrusiveness, and perceived reliability of a sewer system as compared to individual onsite systems, in particular those that require professional oversight. Cluster systems may be effective at mimicking a sewer system, in that treatment and management functions are off-site. However, homeowners do not have the same "flush and forget" mentality about cluster systems as about larger, centralized systems, and they sometimes worry about system abuse by neighbors.

Residents are also typically very concerned about the impacts of wastewater infrastructure on community character. A sewer system will likely promote more and higher-density growth, which entails a loss of open space, more traffic, higher taxes to pay for more schools and services, and changing cultures, politics, etc. Residents will often oppose a sewer if they can get by with existing septic systems, but will support a sewer (in spite of the impacts on growth) if their only other option is highly-managed and expensive onsite systems.

Finally, residents will be more likely to support wastewater infrastructure projects that protect assets that are of great aesthetic or recreational value to the community, such as lakes, harbors, or beaches, regardless of the results of risk assessment.

As we saw above (2.3.2), building densely on a property makes the property more valuable than less dense development. A similar principle holds on a community scale. Studies in the literature of fiscal impact analysis have looked at the cost to society of compact versus sprawling development, and found that compact development is less expensive than “leapfrog development,” where cities spread in bounds, leaving unbuilt-on parcels inside the fringe for quite some time. The savings come from capital costs of the transportation and public works infrastructure, plus energy use and other operating costs for transportation, and schools (Real Estate Research Corporation 1974; Burchell and Listokin 1995; both cited in Edwards 1997; U.S. EPA 1996). It seems reasonable to believe that many of the advantages of compact over leapfrog development also are true for small-lot versus large-lot zoning. Wastewater systems—be they sewers or advanced onsite systems—that make small-lot zoning feasible can, then, reduce the cost of providing services to the homeowners.

Research project: Impacts of lot sizes. In areas zoned for development into residences with decentralized wastewater treatment, it would be useful to compare the fiscal impacts of various lot sizes, the demand for lots of various sizes when a choice is given, along with the costs of the treatment systems needed for each alternative.

Some studies show that the value of cluster housing with open space is as much as 50% higher than for conventional development patterns. Homeowners prefer ponds and wetlands, and areas which support hiking, boating, bird watching, etc. (U.S. EPA 1996). Environmental protection can also follow from compact development. A South Carolina study of both a high-density neighborhood and a conventional subdivision showed that runoff from the subdivision was 43% higher in volume, and levels of nitrogen, phosphorous, and chemical oxygen demand were higher in this runoff (U.S. EPA 1996).

Control of sewer line extensions is used as a growth management tool by a number of communities, and some people see advanced onsite treatment systems as permissive of increased or random development. Concerns about the impact of the availability of advanced onsite wastewater treatment systems on development patterns in areas which have weak zoning and land use planning institutions have held up the permitting of these systems in more than one state. In Wisconsin, for example, the debate delayed a new septic code for much more than a decade.

Research project: Relationships of decentralized systems to “smart growth.” It would be useful to document where advanced onsite treatment systems have been used to promote “smart growth” objectives, or at least have been compatible with them. One of the key factors is the strength of the local or state planning institutions. Where growth is regulated by genuine planning, and the septic code is not used as a surrogate for the difficult decisions planning requires, then changes in the code to permit more advanced wastewater treatment need not affect growth patterns. In areas identified as success stories for smart growth, what institutions have fostered this success? Has the septic code affected growth patterns in these areas in any way? If so, how? Is decentralized advanced treatment permitted in these areas? If not, what effect would changes in the code to permit advanced treatment have on growth? Answers to these questions could help overcome opposition to improvements in decentralized waste treatment among opponents of sprawl.

3.2.3.2 Fairness

Choices between centralized, decentralized, and cluster treatment systems can raise a host of equity and fairness issues. For instance:

- Decentralized treatment is constructed on a unit-by-unit basis as properties are built. Centralized treatment plants and collection systems are built with excess capacity for future expansion. Depending on the financing mechanism, this may produce a subsidy by present ratepayers to those who move in later on. This is not uncommon in large infrastructure projects, however, and may be readily accepted in communities where growth is being encouraged. In other communities, where an influx of new residents is resisted, imposing extra costs on present residents is likely to be contentious.
- In some places, where strict requirements on new onsite treatment systems are necessary due to the loading produced by inadequate or failing existing systems, new growth may, in effect, subsidize previous development.
- Sometimes the rich are the beneficiaries of publicly funded infrastructure, as when sewers are constructed to serve lakeside properties that only the affluent can afford, while other nearby, less-affluent areas go unsewered.
- Onsite systems must be built for the particular influents associated with the attached home or facility. In cluster and sewered systems, some users—e.g., restaurants, schools, and industry—may load the system out of proportion to other users such as residences, raising questions of whether pre-treatment should be required, or differential pricing instituted.

Besides questions over the fairness of who pays and who benefits, questions will arise as to whether is a subsidy is justified or not. For some communities, subsidizing wastewater management for sensitive resource areas, or for important employers, may be reasonable. Also, the related issue of the amount and sharing of fixed or step-function costs can arise. When the choice of treatment scale affects the density of development, it may thereby affect the number of residents available to pay for roads, schools, and other infrastructure. This may lower costs per resident, or increase them if a threshold is reached where larger roads, schools, and so on are required.

Research Project: Equity implications of wastewater treatment choices. How wastewater system choices affect growth and the sharing and fairness of system costs is a topic that comes up in many communities. To our knowledge, there has been little systematic work to clarify the potential issues, the conditions in which certain types of conflicts arise, and the pros and cons from an equity perspective of different financing and institutional arrangements. A survey of the range of issues and available approaches to resolving fairness questions, appropriately illustrated with case studies of real conflicts and adopted solutions, would valuably inform program development efforts by wastewater and community infrastructure financing entities and allow professionals and communities to avoid commonly occurring equity issues and related political pitfalls in the interrelated tasks of choosing physical configurations and methods of paying for them.

3.2.3.3 Value of releasing water locally

3.2.3.3.1 Water reuse

Water reclamation and reuse is a viable source substitution strategy for communities facing water supply constraints or high costs for new supply. Many nonpotable uses can be supplied by reclaimed water, including landscape irrigation, industrial processes and cooling, construction, vehicle washing and other cleaning, toilet and urinal flushing, fountain supply, and more. Reclaimed water is also used to maintain landscape ponds, aquaculture systems, wetlands, and other environmental amenities and resources. The U.S. EPA (1992) and Asano (1998) provide many water reuse case studies. Reuse can be accomplished by dual distribution systems for centralized treatment and return to points of use, or by onsite reuse systems which avoid the expense of dual water supply piping. The potential economic costs and benefits of onsite reuse are discussed earlier in this paper from the micro perspective. Macro-level analysis must also include the costs and benefits associated with source substitution and environmental uses of reclaimed water. The relative suitability of onsite, cluster, and centralized systems in providing those benefits at least cost depends greatly on specific local conditions. Facility planning processes should incorporate reuse configurations, costs, and benefits, but typically do not.

3.2.3.3.2 Pollution abatement

Land application of treated wastewater is sometimes a viable strategy for utilities to avoid or reduce advanced treatment needs attendant to point source discharges to surface waters. In such situations, application rates are often much greater than in cases of wastewater reuse for irrigation, as the primary purpose is avoiding a discharge to surface water, rather than source substitution. The U.S. EPA (1992) has documented land application for pollution abatement for centralized wastewater systems.

Research project: Avoided treatment costs through decentralized systems. It would be useful to clarify circumstances in which decentralized systems, as an alternative to centralized treatment and high-rate land application, could help a utility avoid increased treatment costs by reducing surface water discharges. Such work should also address how this alternative would affect risks to the environment or public health.

3.2.3.3.3 Mitigation of hydrologic problems

The role of water *supply* systems in altering ground water tables and surface water flows is widely recognized. Less well known are the ways centralized wastewater systems contribute to the problems caused by supply withdrawals or themselves cause significant hydrologic changes:

- Collection systems prevent local recharge of ground or surface waters when they transport locally withdrawn water disposed to sanitary sewers to out-of-basin treatment plants, or dispose of treated effluent via ocean, lake, or distant downstream outfalls.
- Collection systems experiencing significant infiltration intercept natural ground water flows and transport water to distant discharge points, often significantly reducing local ground water recharge and stream base flow support.
- Combined sewer systems also intercept runoff that would otherwise add to streamflows between the point of sewer inflow and the eventual treatment plant discharge, or move the water out of basin.

Centralized sewer systems contribute to significant hydrologic problems in many areas, and have motivated steps to redress these problems in some.[4] Besides the hydrologic effects, sewers can produce or contribute to a variety of economic costs associated with alterations in hydrologic regimes:

- increased water pumping costs as ground water tables recede;
- increased water treatment needs with saltwater or brackish water intrusion to aquifers or coastal streams;
- reductions in property values where water supplies are lost or threatened;
- replacement of local ground water supplies with other sources if ground water becomes too contaminated or deep;
- increased POTW treatment costs as assimilative capacity is reduced when stream flows decrease;
- reduced fishery and recreation value due to reduced stream flows or compromised water quality because of reduced flows;
- loss of wildlife habitat and water filtering services as wetlands and riparian zones decline.

In some areas recognition of the hydrologic issues has led to consideration of increased use of decentralized wastewater treatment systems.[5] However, none of the persons contacted for this study were aware of any studies of the economic impacts associated with hydrologic alterations produced by sewer systems. Nor did they know of any efforts to assess the costs decentralized wastewater treatment systems can avoid through reducing alterations to hydrologic regimes by facilitating water reuse to displace water supply withdrawals; avoiding use of large collection systems and the out-of-basin transfer of I&I water and locally supplied sanitary water associated with them; and directly recharging local ground water, via soil absorption as a part of treatment or as a result of reuse. In summary, while the water *quality* implications—positive and negative—of wastewater systems are the subject of a voluminous environmental and economic literature, the issue of the hydrologic impacts of sewerage appears to be in early stages of recognition by wastewater professionals, regulators, and environmentalists.

Research project: Hydrologic and economic effects of sewerage systems. Research to scope the nature and extent of this issue from hydrologic and economic perspectives, to clearly articulate how decentralized wastewater treatment systems can reduce hydrologic impacts, and to develop economic valuation methodologies and examples could spark important advances in environmental management. Values could be extrapolated from water supply studies that place costs on reductions or replacement of water supplies. A substantial body of literature on non-market valuation of the assimilative, recreational, and habitat values of instream flows could also be applied.

3.2.3.4 Monetary and non-monetary value of reusing wastewater nutrients in agriculture

In Scandinavia, concerns about the sustainability of sanitation systems have led to a profusion of newly developed wastewater technologies in the last decade (e.g., Kløve et al. 1999; Staudenmann et al. 1996) A central design criterion for many of these technologies is returning the nutrients in wastewater to agriculture—from whence they came. The focus has largely been on the 5 kg N and 1 kg P that the human body discharges each year (Swedish EPA 1997). The costs and environmental impact of transporting these nutrients from where people live to agricultural land are a major concern, as they are contained in about 400 liters of urine and feces (ibid.), before being diluted even further with toilet water. There has been a premium, then, in developing systems that minimize toilet water use, with separate collection of the blackwater or its urine fraction, which contains most of the nutrients. In addition to the above references, some of the systems are described in Etnier et al. (1997) and Lange and Otterpohl (1997).

Etnier and Refsgaard (1998) have analyzed the costs and performances of these systems in rural areas, and found that nutrient recycling systems were the most effective in preventing pollution from nitrogen, phosphorus, and organic matter, and also the most cost effective in most situations. By going beyond the annual cost of installing and operating a system, and looking at its effect in preventing pollution, it is possible to create a cost-effectiveness index. This type of index can be very useful to planners and regulators working in a watershed context.

Included in these calculations was the value of the urine and feces to the farmers spreading them. While the cost of the fertilizer replaced is relatively low (less than $5 per person annually at U.S. prices), it is a benefit that deserves to be highlighted. While sludge from wastewater treatment plants is often applied to agricultural land, it retains much less of the nitrogen than these decentralized, nutrient recycling systems. Blackwater treatment systems also retain much of the organic matter for reuse in agriculture, but the authors found no satisfactory way to set a value on the improvements in soil tilth.

A number of studies have been done in Sweden using life cycle assessment (LCA) to assess the environmental impact of nutrient recycling systems systems, including the benefits of not manufacturing the fertilizer replaced. While Tillman et al. (1996) did not find a substantial benefit from avoiding fertilizer manufacture, members of the same department worked with a refined methodology to find that replacing fertilizer manufacture did have a notable effect on the total environmental benefits of the nutrient recycling systems (Bengtsson et al.1997). While the first proto-LCA study of wastewater treatment that we are aware of was done in the United States (Antonucci and Schaumburg 1975), we are aware of only the single use of this methodology to assess the environmental benefits and costs of wastewater alternatives. Note that LCA reveals many environmental costs that are incurred off-site, sometimes hundreds or thousands of miles away.

Research project: Implementation of LCA methodologies. It would be valuable to use LCA methodologies to elucidate the environmental impact of various decentralized treatment alternatives used in the U.S.

3.2.3.5 Feedback available from small systems

Feedback loops are important for the proper management of any system (Senge 1993). Small, decentralized systems offer feedback possibilities not available in larger systems. Small industrial wastewater treatment systems with influent from only one company can make it possible to detect problems in the industrial processes that might go unnoticed if the wastewater were leaving the site untreated.[6]

In centralized treatment systems, feedback loops are harder to come by. While some wastewater utilities employ sewage sleuths to track down the largest sources of heavy metals or other contaminants, it is not feasible to be aware of the contributions of each user. Small blackwater or urine systems, on the other hand, lend themselves very well to preventing contamination of products to be land applied, as the tanks can be tested before collection. If the content of hazardous chemicals is too high, then the tank's contents may be rejected, and the user will have to pay a much higher fee for collection—perhaps even having it treated as hazardous waste (Skjelhaugen 1999). A similar system could be implemented for septic tanks. To our knowledge, this sort of system has been proposed but never implemented anywhere. Institutional details like what tests to perform, how to perform them, whether to test all tanks or take random samples, and the costs of these measures need to be addressed.

Research project: Quality assessment of source-separated wastewater for agricultural application. In Sweden and Norway, blackwater and urine separation systems are already in use. Farmers' organizations in those countries are also very aware of chemical constituents in centralized wastewater treatment biosolids, and wish to avoid applying these potentially hazardous biosolids to the soil. It would be useful to evaluate the efficacy of source separation and testing schemes. For instance, working from standards that the Scandinavian farmers' organizations set, the contents of a broad selection of blackwater and urine tanks should be analyzed to see whether they meet these standards. At the same time, low-cost testing methods and testing frequency protocols should be identified and evaluated. Together, these initiatives will give a basis for determining the risks, costs, and benefits of testing source-separated wastewater products for agricultural use, and a rationale for developing comparisons of source separation and centralized biosolids processing in this country.

3.2.3.6 Other externalities

In Sweden, starting in the fall of 1999, the farmers' union put a year-long moratorium on accepting sewage sludge on their fields, because of fears of heavy metals and halogenated compounds. Swedish treatment plants are landfilling or storing the sludge, and investigating installing or using incinerators. This is a cost associated with a system designed to take care of wastewater, without fully considering in what form the products will leave the technosphere and re-enter the biosphere, and who will be affected by this. Where septic tank sludge is treated in central wastewater treatment plants, this, too, is affected by the moratorium. Costs instituted by something like a farmers' moratorium are difficult to foresee but can be quite high if they occur. They can include landfilling fees, extra transportation, incineration fees, or even all the costs of designing, permitting, constructing, and operating a new incinerator. These costs may be avoided if the system is designed from the beginning to meet the needs of the end user, the farmer.

Another type of externality of centralized treatment is the secondary cost of laying and repairing the sewer network. When roads are torn up to repair pipes, this causes downturns in business, delays for travelers, and sometimes damage to vehicles driving over roads under construction.

3.2.4 Market and other possible distortions

In the energy sector, years of subsidies to fossil fuel-based and nuclear energy have slowed the implementation of conservation programs and renewable energy sources (Lovins 1979; Lovins and Lovins 1980). To our knowledge, no similar survey of the wastewater field has been done on the effect of public spending and regulations on the costs of centralized vs. decentralized treatment systems. Factors to include in such a calculus are:

<u>Public spending</u>

- How much public research and development money is allocated to developing technology for centralized vs. decentralized systems?
- How are the costs of land used incorporated into the cost of centralized and decentralized treatment?
- How have construction grants and interest-subsidized loans been allocated to centralized vs. decentralized systems?

<u>Regulation</u>

- How do regulations for decentralized treatment compare with those for centralized treatment in the same state or county: stricter, less strict, similar?
- Are the regulations for centralized treatment more homogeneous, nationally, than those for decentralized treatment? If so, has this been a barrier to the spread of decentralized technologies?

Research project: Opportunities and barriers in public spending and regulation.
Documentation of what role public spending and regulation have had in promoting or hindering the use of decentralized treatment is important background information for assessing its costs and competitiveness with centralized technologies.

Local political and financial pressures can also distort decision making about system options. For instance, as many cities lose population and tax base to suburban development, they can face difficulties in recovering sunk costs of public services, including those of large centralized wastewater systems. Extending or planning services such as wastewater collection can provide a rationale and tool for annexation that offers new revenue sources. If pressures for annexation are strong, decentralized wastewater treatment options for the growing outer areas may be overlooked.

3.2.5 Financial planning and risk

Apart from physical and managerial economies and diseconomies of scale, project size can also affect the financial economics of capital investments. Scale affects the time at which costs must be incurred and how they can be recovered. And scale affects uncertainty and risk—and thereby financing costs—in many ways. The following discussion of these issues draws heavily on rapidly developing experience and theory in the electric utility industry, as summarized in Lovins and Lehman (1997). Electric utilities increasingly are turning away from large generating units and towards small-scale systems (combined cycle gas turbines, wind power, photovoltaics, and other technologies) for financial and other reasons. While the theory of capital expansion planning is essentially the same across utilities, some of the advanced risk adjustment concepts that have gained currency in the energy industry are only now beginning to be utilized in the water and wastewater industries (Rubin 2000). Further efforts to research and apply these concepts to the economic evaluation of decentralized wastewater technologies may reveal decentralized systems are more cost-competitive than currently thought.

It is important to note that these concepts require a utility or utility-like institutional structure. They may apply to development of cluster systems as an alternative to development or extension of centralized systems for a small community or an urban fringe area. They may be less relevant to dispersed onsite systems, built for and managed by the homeowner, where a utility or similar institution does not exist and may not be appropriate. (However, even dispersed rural onsite systems may be amenable to utility management. Some rural electric cooperatives, for instance, are initiating efforts to bundle onsite system services with other services such as electricity, water, and telecommunications.)

3.2.5.1 Value of deferring large investments

Engineers and planners recognize well the time value of money. All other things being equal, one would rather obtain a given amount of revenue now than in the future, or expend a given amount in the future rather than now. To reflect this, a discount rate is applied to equalize costs and revenues in the future with those in the present and provide a uniform basis for comparing projects with differing revenue and cost streams over time. In many cases it may be economically prudent to use decentralized technologies to delay implementation of centralized systems, even if the decentralized technologies are more expensive on a per capita basis and/or will be replaced by the centralized system. If the centralized capacity expenditure is sufficiently large and can be delayed sufficiently long, the present value of deferring the investment may be larger than the smaller near-term expenditures on decentralized capacity. Of course, if delaying a centralized capacity expansion would significantly increase its costs—for instance, by requiring more expensive sewer line trenching due to disruption or avoidance of other infrastructure installed in the interim—deferral may not be prudent. Evaluations of deferral value must be carefully made on a case-by-case basis. Following are two interesting cases of deferring large investments in centralized systems.

<u>Cluster systems for an urban fringe area</u>

In Alabama, the Mobile Area Water and Sewer Service utility (MAWSS) has recently elected not to extend sewer lines to a large, rapidly growing area located outside of the main watershed it serves. Instead, MAWSS is participating with developers in the costs of building cluster systems to serve growth outside its sewer collection area, assuming management of the facilities, and charging hooked-up residences and businesses as if they were sewer customers (albeit at a higher rate currently, as MAWSS does not yet have a good estimate of the actual management costs). In the future, the utility may sewer the area by connecting the clusters, or it may leave the area on cluster treatment. It will not have to make this decision for many years (Steves 2000).

Blackwater separation to extend life of centralized systems

In certain circumstances, it may make economic sense to hook up the blackwater from sewered houses to a decentralized system. Where sewers are in need of renovation but the sewers are hard to find, it may be possible to treat the blackwater separately and use the sewers for graywater. There have been discussions in the town of Ås, Norway, about doing just that. With solids content of the blackwater removed, the thought runs, the existing sewer pipes may not have clogging problems and their useful lifetime may be extended significantly.[7]

This could also be a model for a two-stage decoupling of a centralized system. It is quite conceivable, when the present sewer line becomes unfit for even graywater, that it would be less expensive to install onsite systems for graywater treatment than to install new sewers for just the graywater.

Research project: Prototype cost analyses of blackwater separation for sewer life extension. A possible research project would be to examine areas where sewer pipes need renovation and do cost estimates on alternatives using separate collection of blackwater and maintaining the flow of graywater in the present pipes. Savings may be possible here.

3.2.5.2 Value of small, incremental investments

The value of decentralized units is not only in deferring large centralized capacity investments. Decentralized units can also replace centralized capacity. One might think that the financial costs per dollar borrowed for wastewater treatment capacity would and should be the same whether capacity is provided by centralized or decentralized means. However, providing capacity by different approaches involves different types and degrees of risk. Risk is the heart of finance: the higher the risk, the higher the return required, and thus the higher the cost of capital. Differing risks for different types of treatment systems result in—or *should*, if providers of capital are astute—differing financial costs.

Readers will quickly recognize that because many decentralized technologies are not yet well-developed or generally well-managed, they involve a higher degree of risk than centralized technologies. This section of the paper draws attention to far less-recognized ways in which the scale of systems, rather than the type technology itself, affects risk. These factors tend to tilt in favor of decentralized systems.

The goal of physical capacity planning is to provide a needed service at an affordable cost. Successful capacity planning might also be defined as planning that minimizes future regret—regret that one built too much, or too little, or the wrong kind of capacity. The more closely planners can approach the ideal of “build-as-you-need, pay-as-you-go,” the lower the potential regret, and the lower the economic risk.

How closely one can match the size and type of demand growth depends on several characteristics of the units one can build: size, lead time to bring on-line, and flexibility in operations and upgrades.[8] These factors affect cost because of several types of risk inherent in capital project planning and finance: forecasting risk, financial risk, and the risk of technological or regulatory obsolescence.

3.2.5.2.1 Forecasting risk

The longer the time lag in planning and building capacity, the farther into the future a demand forecast must be made, and therefore the greater the uncertainty inherent in the forecast. The worst case scenario is if demand fails to materialize. Consumers pay a price penalty to cover the unused capacity, and/or the utility may be plagued by difficult financial conditions if rate increases to cover the loss in projected rate base are politically problematic. Longer range forecasts necessary for large plants are moderately to significantly riskier than short-range forecasts for implementation of small technologies.

Related to the uncertainties in demand forecasts is the carrying cost of capital tied up in unused capacity. Small, rapidly implemented units of capacity allow closer matching of a demand curve over time, while “lumpy,” slower units greatly overshoot initial demand within each unit’s time block and leave capacity idle until demand can “grow into it” at the end of the time block. Moreover, adding one more unit to 100 similar small ones rather than one large unit to two similar big ones causes an incremental capitalization burden of 1%, not 33%. If smaller increments can be added sequentially over a period of years to match the physical capacity of a large unit, they will strain a utility’s financial capacity far less at any point in time.

3.2.5.2.2 Financial risk

Because small units strain financial capacity less—that is, they require less working capital—they reduce the risk of default and thus may reduce the cost of capital. Also, short lead-time units expose a utility to the financial costs of construction delays and cost escalations far less than large, slow-to-build units.

Shorter time-to-availability also allows a utility to hook-up new growth more quickly, which can produce higher revenue returns over time, depending upon local regulatory and political conditions. This is analogous to the real estate development economics of building houses that are each sold as finished, with the sale proceeds helping to finance the next house, versus tying up larger amounts of capital in a large apartment complex that can’t provide rental revenue until it is completed. (This effect is most important to private utilities that are not allowed to “rate base” an asset until it is “used and useful.” Public utilities may be able to pre-charge for assets that are not yet in service.)

Small, fast units can also give a utility a longer “breathing spell” after the financial strain of a costly, prolonged project by, for instance, allowing a utility to avoid new sewer line extensions. Utilities can also use decentralized technologies to “leap frog” politically or financially risky areas, avoiding sewer extension costs and still producing revenue while systems are planned and negotiated for intervening, sensitive areas (English 2000).

Experience in the electric utility industry shows that the advantages described so far, and a number of other aspects of the reduced risk and lower carrying cost of smaller, faster units, mean that the construction costs of large, long-lead time plants must be considerably less on a per unit basis than small, short-lead time plants for larger plants to be the economically prudent choice. Research into the applicability of these concepts to choices of system scale in wastewater capacity planning would be very useful. It may be the case that apparently more expensive decentralized wastewater treatment technologies can compete with apparently less expensive conventional technologies when these sorts of effects are accounted for.

3.2.5.2.3 Risk of technological or regulatory obsolescence

Rapid technological change is endemic to most industries today, and marked change in the technology available to the wastewater sector is now occurring (Tchobanoglous 2000). In a period of flux, the smaller and faster to implementation the units ordered, the less risk. Less capital is tied up in technologies at risk of obsolescence; a larger fraction of capacity at any time will use the latest, most competitive technology; and the associated organizations can learn faster and drive continuous improvement (rather than shelving the lessons learned from one giant project for engineers to dust off a decade or two hence when the next large project is built). While the value of the resulting risk reduction is hard to quantify, because the nature and size of technological risk is by definition unknowable, it is prominent in the thinking of strategists in such industries as telecommunications, information technology, and energy, and should be so in the water and wastewater industries. The overall thrust of changes in these other industries is away from centralization and towards smaller technologies along with distributed intelligence and control. It seems likely these and other developments will increase the viability of small-scale, decentralized technologies for wastewater management, thereby increasing the significance of decentralized technologies for addressing the risk factors discussed above.

The cost, siting, and even practical availability of technologies depends significantly on regulatory requirements, tax rules, and other public policy, and ultimately larger developments in society, culture, and politics. Continuous conflicts between various groups amidst evolving social, environmental, and economic concerns make future regulations unpredictable in detail (though perhaps predictable in a general trend toward higher standards, however implemented), and thus an important source of risk. In meeting increasing demands, the advantage would appear to rest with centralized systems. Retrofitting a central plant is a single project that may offer significant planning, design, managerial and construction efficiencies over retrofitting each of many decentralized systems. And it may be politically easier to tighten requirements at the plant level than at the household level. However, these advantages would be reduced or overcome in situations such as the following:

- Limited space at centralized facilities precludes upgrades that require spatial expansion of the facility.
- The need to upgrade performance is not uniform across the watershed or service area. For instance, upgrades to a centralized facility may be required because of insufficient assimilative capacity at the discharge point, while upgrades to a decentralized system may not be needed because local assimilative capacities associated with dispersed,

nonpoint discharge are sufficient. Moreover, upgrades to a centralized facility would have to accommodate all constituent-of-concern sources within the sewershed, while if the corresponding area were served by onsite systems, many of those systems may have adequate discharge areas and soil conditions and not require upgrades. It may only be necessary to retrofit a small number of units that are producing a disproportionate impact on the watershed, or those affecting a portion of the watershed of particular concern, though some of the savings may be offset by additional costs in determining which units to retrofit.

- The problem is from leaking sewer pipes.
- Advances in decentralized technologies mean that future growth can be accommodated with improved decentralized systems, versus being "locked in" to an already-chosen centralized technology. Because decentralized technology is in many ways an "infant industry" with high potential for rapid improvement, this factor should not be overlooked. The flexibility of small, rapidly implementable units could be a key advantage for decentralized systems in times of change.
- Plant expansion triggers reevaluation of existing facility permits, allowing regulators to change discharge standards, resulting in additional modifications to the facility (English 2000).

Research project: Responsiveness of system types to changing performance demands. Further elaboration and illustration with case studies of the relative advantages and disadvantages of centralized and decentralized systems when performance demands change would be useful. Such research should also address how management systems and regulatory institutions can best develop to take advantage of distributed intelligence and control when small-scale systems are coupled with advanced information and communications technologies.

3.2.5.2.4 Valuing incrementalism: available tools

Other industries are beginning to apply a variety of sophisticated analytical tools to address risk and put a value on modularity, short lead times, and flexibility. These techniques go far beyond engineering approaches that place judgmental probabilities on uncertainties in order to develop probability-weighted recommendations. For instance, option theory—a tool used in financial investment management—allows one to value options one may exercise in the future, such as may be allowed by a finely incremental capacity expansion strategy. Rather than assessing the net present value of decisions envisaged now, option theory assesses the additional value of flexible choices now to delay a decision until more is known. Decision analysis is another potential tool. Rooted in operations research, management theory, and financial analysis, it uses an elaborate simulation of millions of possible decision trees to determine the optimal decision under each set of possible outcomes, and thence the optimal decision policy to pursue under the assumed uncertainties. The procedure requires significant analytic effort and may not be appropriate for most wastewater service providers.

The point here is simply that alternatives are available to the engineering profession's traditional tendency to approach risk by overbuilding. Given the huge current gap between water and wastewater infrastructure needs and actual spending (Allbee 2000), overbuilding is no longer financially tenable. Its costs will be increasingly questioned, and alternatives increasingly needed.

Research project: Data and methodologies for assessing economic risks and values associated with system scale. As technology and management systems for decentralized wastewater treatment improve, it would be useful to develop planning and economic analysis tools that can adequately account for any financial advantages incremental implementation of decentralized technologies may offer. Research to comprehensively compile and illustrate ways the scale of wastewater systems affects their costs and risks would be a valuable step toward development of such tools. Such research would consider forecasting risk, financial risk, risk of technological obsolescence, and risk of regulatory obsolescence. It would also incorporate as appropriate the other scale-related factors discussed earlier in this paper.

3.3 Methods and models to assess impacts and costs of wastewater treatment

A comprehensive facilities planning process must include both a needs assessment and consideration of alternative risk-reduction strategies. Typically, engineers have provided only a simplistic assessment, and have given short shrift to technologies other than central sewers and activated sludge treatment plants. What models and methods can facilitate development of a more appropriate plan? These models must take into account not only the risks to public health and the environment, but also the monetary and non-monetary costs to the community. Models should also be developed which can assess the environmental and public health risks of those options which generate wastewater reuse values or which benefit from long-term financial and risk planning approaches.

3.3.1 A catalog of models.

The water quality field in general has recognized the importance of developing transport models for nonpoint source pollution, as well as user-friendly models and tools for testing the impacts on public health and the environment of remedial measures. These models should distinguish among agricultural, wildlife, fertilizer, sewer leakages, and onsite systems, and other sources. But, because the onsite field has focused so much on the single-family home and the uniform code approach, very little work has been done on cumulative impacts of septic systems. Several researchers have been attempting to fill this gap.

The MANAGE (Method for Assessment, Nutrient-loading, And Geographic Evaluation of watersheds) model is a relatively simple and low-cost approach to predicting nitrogen impacts from septic systems based on soil types, density of housing and other land uses, and allows for consideration of options including restricted development, advanced treatment in new housing, and advanced treatment in remedial upgrades of existing systems. The model has typically led to the establishment of advanced treatment “zones” surrounding critical drinking water and sensitive estuary or other resource areas. Within these zones, dependent on soil type and depth to ground water or ledge, technologies must be installed which meet specific nitrogen and fecal coliform removal standards.

Mike Hoover of North Carolina State University has developed a risk-management methodology for the identification of “management control zones” in a community within which various advanced treatment standards are required. These zones reflect the sensitivity and importance of drinking water sources, estuaries, and other natural resources, and treatment standards vary for nitrogen or phosphorous removal, disinfection, etc. Matrices of treatment zones vs. treatment standards, and treatment standards vs. soil conditions form the basis of decision making about system installations, and treatment zones are generated from GIS-based maps on soils and resources.

Bob Siegrist at the Colorado School of Mines is beginning a study to incorporate decentralized wastewater treatment into EPRI’s WARMF (Watershed Analysis Risk Management Framework). WARMF is a complex decision-support tool that incorporates models for point and nonpoint-source loadings of pollutants in rivers and lakes. Siegrist will add models for septic system inputs, including microbes to drinking water wells, nutrient loading to lakes, surfacing of effluent, and land use and sprawl. Decision options will include: upgrade of existing systems, increased design requirements on new systems, changed density of development, provision of public sewers, and required routine monitoring and reporting.

Additional models include: a three-dimension ground water flow and nitrate fate and transport model by USGS in LaPine, Oregon; a site-by-site risk analysis by Jerry Stonebridge and Bill Stuth in Burnett, Washington; and a decision tree risk analysis to determine whether systems could constitute high, medium, or low risk to drinking, irrigation, or industrial use of surface waters or ground waters, which has been developed by Dick Otis.

Karen Refsgaard, at the Norwegian Agricultural Economics Research Institute, has developed a linear programming model that can find the least-cost onsite method for achieving whatever treatment goals the community chooses to set. It can also be used to find out how much extra cost a given increase in treatment standards would impose.

Research project: Assessment and development of low-cost, community-level models. These models are in various stages of development. Each has a cost in terms of data collection requirements and analytical time. Ultimately, communities need models which provide useful information at reasonable cost. A comparative analysis of the models would be a significant contribution.

3.3.2 Other developments

Research project: Fecal typing techniques. Fecal typing techniques have been developed by Charles Hagedorn (1998; 1999) and others to identify accurately the sources of water-related risks to public health and the environment. Continued work in this area should be supported.

3.4 The role of economics and other factors in decision-making processes

As touched on in the introduction, domestic wastewater is treated to reduce risks to health and the environment and to improve natural resource management. However, decisions about wastewater treatment reflect additional factors, such as engineering reliability, financial costs, and political acceptability. Nelson and Shephard (1999) define accountability in wastewater treatment decisions in three major dimensions: protection of public health and the environment; community needs and preferences; and practicality or feasibility. We have seen also that where wastewater is treated can profoundly affect the local hydrological cycle, and it can be a tool in "smart growth" planning. Ridderstolpe (1999) gives an overview of other parameters for decision makers to consider, including reliability and how well the system fits into existing spaces and institutions. Finally, improvements to the environment are often described in terms of changes leading to sustainability. Sustainability by itself has many more aspects than are listed here; some lists are found in Otterpohl et al. (1997) and Lundin et al. (1999).

In Section 2.3.1, concerns of homeowners about issues such as inconvenience and intrusiveness were outlined. These also come to the fore at the macro or community level of decision making. Voters will typically question the creation of new public bureaucracies, particularly those that involve oversight over individual onsite systems. They will resist the imposition of new fees, unless it has been shown that public health or a valued community resource are threatened. Both regulators and municipal officials will prefer known technologies, traditional management approaches, and minimal public opposition (Nelson et al. 2000).

3.4.1 Facilities planning, benefit-cost analysis, risk assessment, and other decision-making frameworks

Facilities planning, benefit-cost analysis, and risk assessment, if conducted properly, should include the same considerations of public health, environmental protection, financial costs and public acceptability. However, in practice, these analytic structures tend to emphasize different assumptions or features of the decision-making process.

As described in Section 3.1, facilities planning has typically focused on central sewer technology options and costs, with lesser attention to a comprehensive identification of all pollution sources or to small system technologies. Existing water quality standards embedded in regulatory requirements would be assumed, without regard to relative risk.

Conventional benefit-cost analysis, infrequently used at the community level but commonly used to assess state or federal policy options, weighs benefits against costs to rank alternatives. The emphasis is inevitably on measurable monetary costs. Costs not normally measured in dollars, such as emissions and discharges, are translated into economic terms by the use of a host of techniques (e.g., Smith 1996). The legitimacy of using these techniques is hotly contested, with many people arguing that these methods cannot measure many things that are valued in political decisions (e.g., Sagoff 1998).

A macro or community level risk assessment would be organized around the four areas of engineering risks, risks to public health, risks to the environment, and socioeconomic risks, as described in the introductory paper to the National Decentralized Water Resources Capacity Development Project Research Needs Conference by Dan Jones (2000). Use of this framework for analysis would lead to a much more careful assessment of fate and transport of pollutants, along with exposure and effects on human or natural organisms. Existing water quality standards might or might not emerge as appropriate targets for a balanced program of risk reduction in any given community.

Finally, multiattribute (or multicriteria or multiobjective) decision making is another way to formalize decisions that involve tradeoffs among different parameters, without converting all values to money. Balkema et al. (2000) are working on a computer model to optimize selection of wastewater treatment options, given criteria for sustainability, cost, and other factors. Eilersen et al. (1999) use a different approach, polling future residents of a Danish eco-village on what they consider sustainability to entail, and then rating three wastewater treatment options according to these criteria.

An important aspect of garnering political support for any technological change is consulting the stakeholders (Sclove 1995). This can lead to development of alternatives that are better suited to the community's needs, as well as increase acceptance of the chosen course of action. Involvement in the decision brings greater endorsement of it. This is especially important for decentralized wastewater treatment, where the system's efficacy relies on the behavior of people in each household. Many regulatory decisions can have no effect on actual health and environmental conditions if there is not broad support for them.

Research project: Community-level risk assessment. A prototypical risk assessment methodology at the community or watershed level should be developed as a means to investigate whether water pollution control strategies are appropriately balancing risks. For example, risks of virus transmission can be severe, but have been given inadequate attention in wastewater facilities planning.

Research project: Usefulness of multiattribute decision making. It would be helpful to document the models used for multiattribute decision making currently used for wastewater treatment and assess their usefulness.

Research project: Importance of homeowner support to achieving risk reduction. Case studies could be compiled to identify factors which increase acceptance of changes in decentralized wastewater technologies, and to assess how important homeowner support is to ensure that changes result in reduced consequences for health and the environment.

4 SUMMARY AND PRIORITIZATION OF RESEARCH NEEDS

A list of the research projects identified appears in Appendix A. Here we list the projects we prioritize. We have used three criteria in developing these priorities: 1) basic information the field of decentralized treatment needs, 2) information needed to assess new directions the field is taking, and 3) information needed to overcome existing obstacles to decentralized treatment. Appendix B provides further detail about the prioritized projects.

Documenting the importance of performance standards is both basic information to the field of decentralized treatment and, we believe, would help overcome an obstacle to more widespread use of decentralized technologies. The lack of national standards causes needless difficulties both for the local regulators and for the manufacturers. We recommend an investigation of how other regulated fields have developed performance standards, and how these standards have affected the development of the field. This could be a strong motivation for those with competing approaches to performance standards to unite on common standards, and could provide a road map for how to accomplish that.

The effects of hydrological impacts from wastewater treatment choices is basic information that is needed to assess the benefits of decentralized treatment. There is currently little thought given to these benefits in planning. A thorough investigation of the magnitude of the environmental and economic benefits of local infiltration or reuse of water, in a number of different climates, would provide planners with a better basis for deciding where decentralized treatment should be used.

The lifespans, failure rates, and risks associated with decentralized and centralized solutions also contribute basic information needed to rationally evaluate alternatives. When, for instance, system replacement plans do not keep pace with deterioration, and treatment units fail or sewers leak, this considerably increases risk associated with the systems. An overview of these issues would give planners a better idea of the risks associated with each wastewater decision.

Economies and diseconomies of scale in treatment options need to be systematically compiled. It may be that centralized treatment systems are being built too large, from a financial risk point of view, based on misapprehensions of economies of scale. More criteria for deciding the optimal scale for treatment options will give basic information for the field.

Cost effectiveness of management systems, including performance-based codes, is important to assess now, when fundamental changes in the way decentralized treatment systems are run are being widely considered. It is important to know under what circumstances inspections, remote monitoring, and the enforcement of performance-based codes can be cost effective.

Compatibility of decentralized treatment with smart growth is important to establish to overcome an obstacle to decentralized treatment in many areas. Concerns about increased sprawl when advanced onsite treatment is allowed have fueled opposition to these technologies. Documentation of the actual effects of advanced decentralized treatment, and the ways in which decentralized treatment has been used to further smart growth goals, could help remove these objections.

Appendix A: List of Research Projects

Potential projects are listed in the order presented in the report. No prioritization is intended or implied by this ordering.

Research project: Failure rates and long-term performance. Good estimates of failure rates and/or long-run performance of various technologies or designs are not available, nor are causes of failures well-documented, in particular for technologies developed in the last five years. Minimally maintained advanced pre-treatment units generally have twice the failure rate of minimally maintained conventional septic systems (Gunn 1998; Ingram et al. 1994; Water Resources Research Institute News 1992; Nelson et al. 2000). However, definitions of failure differ among various studies. Widely accepted definitions of different types of failure plus a systematic study of long-term performance with respect to these failure types would have great value to the onsite designer.

Research project: Benefits and costs of mandated inspection programs. How much would long-term professional inspection and maintenance programs reduce risks to public health and the environment, and at what cost both in dollars and in alienation of homeowners? Conventional designs are conservative and able to withstand a certain amount of abuse and neglect. Homeowners typically repair hydraulic problems leading to sewage back-up or surfacing. However, hydraulic failure is not always a sufficient indicator of system performance; a system may not adequately treat wastewater indefinitely. Analysis of systems that are near sensitive surface waters or ground water, or that have mechanical and electrical components prone to failure, will likely show that the benefits of professional oversight exceed the costs.

Research project: Reliability and effectiveness of remote monitoring technologies to reduce engineering risks of treatment failure. Remote monitoring and control has the potential to reduce both the risks of treatment failure and the socioeconomic costs of professional maintenance, but technologies and approaches need to be examined (Tchobanoglous 1999). Anecdotal evidence indicates that remote monitoring can reduce the maintenance and repair costs for some systems, but what needs to be monitored and what system types give net returns on the investment in monitoring technology? And can monitoring reduce the political resistance of homeowners to maintenance programs? For remote monitoring uses which are not currently cost effective, how much would the cost of remote monitoring have to decrease for them to become cost effective?

Research project: Documentation of non-monetary costs. Additional surveys of homeowner preferences would be useful. How much would homeowners be willing to pay to avoid limitations on their water use, garbage disposals, etc. and to avoid having regulators or other maintenance personnel on their property? Answers will be very site specific, so it is also useful to consider how to regularly incorporate such surveys into the planning process, and how the education and planning process can be structured to motivate people to accept the ways onsite systems diverge from so-called “flush and forget” sewers. A second research approach would be to examine housing markets through the method of hedonistic pricing. How much lower are housing prices, all else being equal, for homes with frequent inspections and limitations on quality of life compared to homes with no outside maintenance requirements? What contributes to making this difference higher in some places than others?

Research project: Effective education campaigns. Various studies suggest that when a community has a valued resource, such as a lake, drinking water supply, or estuary, homeowners are more willing to spend money and accept mandated long-term maintenance programs (Eddy 1992; Eddy 1993; Breisemeister 1996). Training centers and other programs have developed onsite system educational campaigns. A study of the effectiveness of educational campaigns in changing behavior would be useful. In addition, advertising campaigns for other public benefits have lessons for wastewater planners (Earle 2000).

Research project: Data and guidelines for valuing water reuse. Guidelines for estimating and including hard-to-value benefits when costing-out treatment system alternatives would be useful to system designers, and their application would improve the economic attractiveness of advanced decentralized wastewater treatment systems. Property value improvements could be estimated with real estate appraisal techniques and hedonistic valuation approaches. Reductions in fire insurance premiums have probably been applied for rainwater collection systems and could be similarly applied for wastewater treatment effluent storage.

Research project: Risk-based guidelines for appropriate reuse conditions. Risk assessments to define the risks of reuse of treated blackwater under various physical conditions, including soil parameters and physical configurations such as surface vs. subsurface irrigation are needed. This will enable development of appropriate regulatory responses, and comparison of the economic values of reuse with any risks involved.

Research project: Guidelines for and effects of code variances. These efforts should be examined and formalized in guidance documents, which also document the additional risks, if any, these exemptions entail.

Research project: Cost structures and leverage points for advances in technology. A systematic study of possible future cost structures would be useful. What are the various cost profiles of typical treatment systems, including: design, installation, tanks, pumps, pipes, transportation, and profit? How much would each of these factors be reduced with mass production and distribution? If costs could be reduced substantially, what steps should be taken to encourage such a market evolution to occur?

Research project: Development of a prototype for decision making. Is it feasible to develop a general risk-based decision-making framework for a standard lot and a variety of higher-risk situations? If so, it would be a valuable contribution to the field, because it would provide the structure for examining a wide variety of technology and management options within a common framework. Such a model could help address a number of questions. For instance, what are the costs of and benefits of risk reduction from the installation of specific technologies, such as watertight (vs. non-watertight) septic tanks, effluent filters, washing machine filters, pre-treatment units, shallow drainfields, etc.? How do the benefits vary with the particular site conditions of the lot and/or the other pre-treatment or soil-based systems being utilized in the treatment train? What effluent quality is a cost-effective goal for pre-treatment units and when is disinfection advisable? Under which circumstances would costly professional inspection and maintenance programs be required for homeowners, in terms of engineering, public health, or environmental risk reduction achieved? Under what circumstances are strict, costly requirements for nutrient reduction or disinfection justified?

Research project: Model code development. A risk-based decision-making assessment framework, if developed, could also be utilized to assess various regulatory (risk management) proposals. Specific issues could include separation distances, seasonal high ground water, additional treatment requirements in sensitive areas, etc. Are some requirements more costly to the homeowner, and to society at large, than can be justified by the expected risk reduction achieved? Alternatively, are there provisions which could be introduced into the code, in particular for high-risk lots, which would reduce risks at minimal costs? What provisions of the code would reduce risks at comparable costs for risk reduction from agricultural waste management, stormwater management, transportation safety, public health measures, etc.?

Research project: Performance-based codes. A benefit-cost study of a performance-based code would be a worthwhile exercise. What specific risk reduction benefits could be achieved by adhering to a performance-based analysis of a site in contrast to implementing requirements of a well-designed prescriptive code? Costs of site analysis and design would be substantially higher than for a prescriptive code. Monitoring expenses, either at the point of release into soils, or in particular after soils treatment, would be very high as well. Regulatory oversight costs would increase (Crosby et al. 1998; Smithson 1995; Gunn 1998; Nelson et al. 2000). Are the risks of failure higher when greater discretion is given to the designer? From a societal perspective, widespread implementation of a performance-based approach would require a diversion of skilled manpower into the onsite field and/or a substantial upgrading of skills of personnel already in the field. Would these costs be justified?

Research project: Technological and regulatory standards for market stimulus. Technology-driven markets expand when consistent standards evolve, either through consensus in the field or dominance by one large manufacturer. Standards in the onsite field have the potential to increase homeowner and regulatory acceptance of new technologies, and to create conditions for substantial innovation, mass marketing, and economies of scale (Hoover et al. 1998; Ruskin 1999; Jantrania 1999). The U.S. EPA (1999) has recommended ground water performance standards after soils treatment be the goal, but this approach does not help to achieve consistency in requirements for pre-treatment units. Conversely, effluent standards may not allow for sufficient consideration of additional treatment in the soils. A project to develop a consensus on standards would be an important contribution to technological innovation and market expansion.

Research project: Cost components across scale. It would be a very useful contribution to the literature and to developing practice in the profession to have a thorough compilation of the cost components of centralized and decentralized (onsite and cluster) systems, illustrating cost ranges across system scale and across important situational parameters, and with a review of which components are most ripe for cost savings given likely and reasonably expected improvements in technology or institutions.

Research project: Infrastructure lifespans and associated risks. Documentation of the actual lifespan of different wastewater infrastructure solutions and the effects and risks associated with their deterioration would be an important contribution. Facilities planning generally considers a twenty-year lifespan for projects. It would be useful to document the actual lifespans of both decentralized and centralized systems, so that proper planning for their replacement could be made.

Research project: Comparative analysis of risks of system failures. A comparative analysis of the risks associated with routine failures would be useful. When a large sewer system and treatment plant flood or otherwise breakdown, huge volumes of wastewater may be released to nearby surface waters. How do these risks compare to flooding/saturation of onsite leaching fields and to continuous breakdown of a small percentage of onsite systems scattered throughout the community?

Research project: Manageability of failures. A properly managed wastewater utility would seek to prevent and/or repair breakdowns as soon as possible. What are the comparative costs of maintenance, the relative risks of breakdown, the costs and timing of major system repair, the likelihood that major repairs would be financed, and the relative risks from inaction for different wastewater treatment systems—onsite, cluster, and centralized? Also, while budgeting and rate-making techniques to provide adequate funds for maintenance, repair, and replacement are relatively straightforward, many wastewater management entities do not use them. Why not, and how could implementation of adequate financial structures be increased?

Research project: Impacts of lot sizes. In areas zoned for development into residences with decentralized wastewater treatment, it would be useful to compare the fiscal impacts of various lot sizes, the demand for lots of various sizes when a choice is given, along with the costs of the treatment systems needed for each alternative.

Research project: Relationships of decentralized systems to "smart growth." It would be useful to document where advanced onsite treatment systems have been used to promote "smart growth" objectives, or at least have been compatible with them. One of the key factors is the strength of the local or state planning institutions. Where growth is regulated by genuine planning, and the septic code is not used as a surrogate for the difficult decisions planning requires, then changes in the code to permit more advanced wastewater treatment need not affect growth patterns. In areas identified as success stories for smart growth, what institutions have fostered this success? Has the septic code affected growth patterns in these areas in any way? If so, how? Is decentralized advanced treatment permitted in these areas? If not, what effect would changes in the code to permit advanced treatment have on growth? Answers to these questions could help overcome opposition to improvements in decentralized waste treatment among opponents of sprawl.

Research Project: Equity implications of wastewater treatment choices. How wastewater system choices affect growth and the sharing and fairness of system costs is a topic that comes up in many communities. To our knowledge, there has been little systematic work to clarify the potential issues, the conditions in which certain types of conflicts arise, and the pros and cons from an equity perspective of different financing and institutional arrangements. A survey of the range of issues and available approaches to resolving fairness questions, appropriately illustrated with case studies of real conflicts and adopted solutions, would valuably inform program development efforts by wastewater and community infrastructure financing entities and allow professionals and communities to avoid commonly occurring equity issues and related political pitfalls in the interrelated tasks of choosing physical configurations and methods of paying for them.

Research project: Avoided treatment costs through decentralized systems. It would be useful to clarify circumstances in which decentralized systems, as an alternative to centralized treatment and high-rate land application, could help a utility avoid increased treatment costs by reducing surface water discharges. Such work should also address how this alternative would affect risks to the environment or public health.

Research project: Hydrologic and economic effects of sewerage systems. Research to scope the nature and extent of this issue from hydrologic and economic perspectives, to clearly articulate how decentralized wastewater treatment systems can reduce hydrologic impacts, and to develop economic valuation methodologies and examples could spark important advances in environmental management. Values could be extrapolated from water supply studies that place costs on reductions or replacement of water supplies. A substantial body of literature on non-market valuation of the assimilative, recreational, and habitat values of instream flows could also be applied.

Research project: Implementation of LCA methodologies. It would be valuable to use LCA methodologies to elucidate the environmental impact of various decentralized treatment alternatives used in the U.S.

Research project: Quality assessment of source-separated wastewater for agricultural application. In Sweden and Norway, blackwater and urine separation systems are already in use. Farmers' organizations in those countries are also very aware of chemical constituents in centralized wastewater treatment biosolids, and wish to avoid applying these potentially hazardous biosolids to the soil. It would be useful to evaluate the efficacy of source separation and testing schemes. For instance, working from standards that the Scandinavian farmers' organizations set, the contents of a broad selection of blackwater and urine tanks should be analyzed to see whether they meet these standards. At the same time, low-cost testing methods and testing frequency protocols should be identified and evaluated. Together, these initiatives will give a basis for determining the risks, costs, and benefits of testing source-separated wastewater products for agricultural use, and a rationale for developing comparisons of source separation and centralized biosolids processing in this country.

Research project: Opportunities and barriers in public spending and regulation. Documentation of what role public spending and regulation have had in promoting or hindering the use of decentralized treatment is important background information for assessing its costs and competitiveness with centralized technologies.

Research project: Prototype cost analyses of blackwater separation for sewer life extension. A possible research project would be to examine areas where sewer pipes need renovation and do cost estimates on alternatives using separate collection of blackwater and maintaining the flow of graywater in the present pipes. Savings may be possible here.

Research project: Responsiveness of system types to changing performance demands. Further elaboration and illustration with case studies of the relative advantages and disadvantages of centralized and decentralized systems when performance demands change would be useful. Such research should also address how management systems and regulatory institutions can best develop to take advantage of distributed intelligence and control when small-scale systems are coupled with advanced information and communications technologies.

Research project: Data and methodologies for assessing economic risks and values associated with system scale. As technology and management systems for decentralized wastewater treatment improve, it would be useful to develop planning and economic analysis tools that can adequately account for any financial advantages incremental implementation of decentralized technologies may offer. Research to comprehensively compile and illustrate ways the scale of wastewater systems affects their costs and risks would be a valuable step toward development of such tools. Such research would consider forecasting risk, financial risk, risk of technological obsolescence, and risk of regulatory obsolescence. It would also incorporate as appropriate the other scale-related factors discussed earlier in this paper.

Research project: Assessment and development of low-cost, community-level models. These models are in various stages of development. Each has a cost in terms of data collection requirements and analytical time. Ultimately, communities need models which provide useful information at reasonable cost. A comparative analysis of the models would be a significant contribution.

Research project: Fecal typing techniques. Fecal typing techniques have been developed by Charles Hagedorn (1998; 1999) and others to identify accurately the sources of water-related risks to public health and the environment. Continued work in this area should be supported.

Research project: Community-level risk assessment. A prototypical risk assessment methodology at the community or watershed level should be developed as a means to investigate whether water pollution control strategies are appropriately balancing risks. For example, risks of virus transmission can be severe, but have been given inadequate attention in wastewater facilities planning.

Research project: Usefulness of multiattribute decision making. It would be helpful to document the models used for multiattribute decision making currently used for wastewater treatment and assess their usefulness.

Research project: Importance of homeowner support to achieving risk reduction. Case studies could be compiled to identify factors which increase acceptance of changes in decentralized wastewater technologies, and to assess how important homeowner support is to ensure that changes result in reduced consequences for health and the environment.

Appendix B: Recommendations for Priority Research Projects

Table B-1. Priority research topics on benefits and costs of decentralized wastewater treatment

Topic	Strategic focus	Tasks	Products	Uses
Development of performance standards	Can national performance standards for decentralized systems increase their acceptance?	What other fields have developed national performance standards. How have they done this? What results has this had for the fields?	Case studies of standards development in other fields and their results.	To guide and encourage the debate about developing standards for decentralized treatment systems.
Hydrological impacts of wastewater treatment	What is the value of releasing water near where it is used rather than discharging it from a centralized treatment facility?	Document the environmental and economic benefits of decentralized water release.	Data on hydrological impacts of release methods and economic benefits from local infiltration.	To help in watershed assessment of the impacts of wastewater treatment options.
The lifespans, failure rates, and risks of wastewater treatment technologies	What are the actual life spans and failure rates of onsite and centralized treatment systems?	Document life spans and failure rates for conventional and advanced onsite systems, sewer systems, and centralized treatment plants. Assess the risks associated with unanticipated failure.	Data on robustness of systems and risks associated with unplanned-for failures.	To assess whether systems, once built, can be counted on to perform in the way planned for.
Financial aspects of the scale of wastewater systems	How does the scale of wastewater services affect costs and benefits?	Fully articulate economies and diseconomies of scale in wastewater service provision. Document the extent to which incremental capacity increases with decentralized systems carry less financial risk and cost than extending centralized systems.	Scale-related cost data for centralized and decentralized methods of capacity expansion.	To develop least-cost plans for increasing treatment capacity.

Table B-1, continued. Priority research topics on benefits and costs of decentralized wastewater treatment

Topic	Strategic focus	Tasks	Products	Uses
Cost effectiveness of new directions in onsite management	When are management and remote monitoring systems cost effective?	Define the monetary and non-monetary costs that management and remote monitoring systems are intended to reduce. Balance these against the monetary and non-monetary costs of such systems.	Problem definition, quantification of monetary costs and benefits, and qualitative listing of non-monetary costs and benefits.	To lay the groundwork for understanding when it is useful to use management regimes and remote monitoring.
	What are the costs and benefits of performance-based codes?	Define the monetary and non-monetary benefits of using a performance-based code, and balance these against the monetary and non-monetary costs of enforcement.	Problem definition, quantification of monetary costs and benefits, and qualitative listing of non-monetary costs and benefits.	To lay the groundwork for understanding when and whether it is useful to use a performance-based code.
Compatability of decentralized treatment with smart growth	What effects do advanced onsite treatment technologies have on land-use patterns?	Use case studies to see whether and how changes in the septic code have affected land use patterns in areas with weak zoning and land use planning institutions.	Data on the nature and extent of the secondary impact from advanced onsite technologies on land use.	To inform the debate about permitting advanced onsite wastewater treatment in many states.
	How can decentralized treatment be used in the service of smart growth goals?	Determine the ways in which decentralized treatment can be compatible with smart growth, and what instituitonal factors are required in order for this to occur.	Technology assessment of decentralized treatment from a smart growth perspective.	To show where alliances may be possible between proponents of decentralized treatment and proponents of smart growth.

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Endnotes

[1] A generalized assessment of the magnitude of this benefit is useful here to illustrate a benefit stream that can be monetized to help offset the costs of decentralized wastewater treatment systems. For example:

> The value to users of water supplied by local water systems depends on the price of the water and the portion of the wastewater effluent that can be usefully reused, and thereby substituted for first-use water. Nationally, the median retail price of water is $2.10 per thousand gallons (Raftelis Financial Consulting1998). Indoor water use averages 69 gallons per capita per day in single-family homes (Mayer et al. 1999). Assuming 3 persons per household, no treatment loss, and use of 100% of the effluent for one-third of the year (landscape water use equals or exceeds inside use in season in many parts of the country) , the annual value of avoided water purchases per home is $52. The present value of this revenue stream across a 20 year "project lifetime" using a 5 percent discount rate is $648.
>
> Electricity costs for residential ground water wells are typically less. For a hypothetical home system pumping against 180 feet of head with a properly-sized, efficient pump at national average electricity prices, electricity would cost about $0.54 per thousand gallons (McCray 2000). Using the same household size and reuse assumptions as above, the value of reuse water would be about $14 annually, or $174 capitalized.
>
> These average values are not impressive, but the potential savings can be considerably higher in many situations; e.g., higher water or electricity prices, greater depth to ground water, larger households or cluster systems, greater irrigation requirements. For instance, 25 percent of U.S. community water systems serving less than 10,000 persons charge $4.00 or more per thousand gallons on residential water accounts (Shanaghan 2000). Assuming this water rate, 4 persons per household, no treatment loss, and use of 100% of the effluent for one-half of the year, annual water savings would amount to $201, and the capitalized value across 20 years at a 5% discount rate is $2,505.

It is important to note that price does not always indicate value. In many water systems water prices are subsidized from other revenue sources or may be based on incomplete cost accounting; for instance, by not including depreciation of assets. In many places, the avoided-cost value of reuse water is likely to increase with time, as utilities increase prices to cover currently unmet replacement and improvement needs. Allbee (2000) provides a succinct overview of the gap between current spending patterns and projected spending necessary to replace and upgrade existing water and wastewater systems.

[2] Lengths per service (connection to a house, business, or other account) increase due to geometries of connecting services in larger networks, and because as one moves from home to neighborhood to sub-regional to regional scales, increasing amounts of land in parks, schools, roads, parking lots, industrial and institutional campuses, golf courses, lakes, etc., are added, decreasing the density of land use. Also, larger networks include larger pipes which have higher material and installation costs per meter of pipe.

Larger systems may also have increased pumping station requirements. Because gravity sewer lines must be sloped, as distances between services and a plant increase, more pumping units must be in place to bring the sewer water back up to economic trenching depths for line continuation. This requirement depends significantly on local topography.

[3] One example of a rigorous examination of scale economies for wastewater systems is a study by Clark, Perkins, and Wood (1997; see also Clark 1997) of Adelaide, Australia, a medium-sized city in a mixed hilly and coastal plain setting. Adelaide is served by four conventional wastewater treatment plants that range in size from 50,000 to 190,000 services (accounts) per plant. The analysis shows that the sewer network in Adelaide experiences diseconomies of scale that offset economies of scale in Adelaide's treatment plants. These diseconomies with increasing service area size include decreasing density and thereby increasing length of pipe per service, increased costs per meter for purchase and installation or replacement of larger pipes, and increased pump station requirements. Density per service ranges from 750 square meters per service at the level of a typical urban house to 1250 square meters at the residential subdivision level to 1855 square meters at the metropolitan scale. Pipe lengths per service range from zero at the house scale (if onsite treatment were to be employed) to 11.6 meters at the residential subdivision scale to 15.1 meters at the metropolitan scale.

For the average densities in Adelaide, cost modeling showed that the optimum scale may be 2500 services/plant. Calculated costs vary by plus or minus 10% from the optimum over the range from 250 to 30,000 services. Some sensitivity tests of assumptions indicated a broader range for the potential optimum. The authors concluded that whole system analysis of wastewater treatment scale options indicates overall cost is relatively insensitive to scale, and therefore small scale systems are likely to be cost competitive with large plants on the basis of whole system analysis of the wastewater system, and more competitive if the wastewater system is designed to realize synergies with localized stormwater treatment, and local reuse of treated sanitary or storm sewer water.

These results supported what Australians refer to as "sewer mining," whereby smaller treatment/reuse plants tap the water resource in the sewer network at points where the water is most needed or could be beneficially applied, thereby allowing drinking water supplies to be diverted to higher-value uses. While sewer mining is a different concept than the non-sewered onsite and cluster system concept that is the focus of the National Decentralized Water Resources Capacity Development Project, the Adelaide study's results imply two of the conceptual issues this white paper raises:

- diseconomies in conventional sewered systems may be more prevalent than commonly thought, and so the costs of connecting to sewered systems versus choosing decentralized alternatives should be thoroughly examined;
- integration and valuation of additional benefits a decentralized wastewater treatment system may provide may "tip the scale" toward a decentralized system.

[4] *Coastal areas:* Lack of recharge due to ocean outfalls threatens saltwater intrusion to aquifers in many coastal locations. Some centralized systems, notably in California and Florida, reduce saltwater intrusion by recharging aquifers with treated effluent through ground injection or surface infiltration basins.

Long Island, New York: Several decades ago, installation of sewers and centralized treatment resulted in detectable lowering of ground water. Resulting reductions in stream baseflows led to development of a Flow Augmentation Needs Study (FANS) that assessed ways to reduce impacts to stream flows, fish, and wildlife (Herring, 2000).

Metropolitan Madison, Wisconsin: In Dane County, Wisconsin, the municipality of Verona withdraws water from the Sugar River watershed and sends a portion of its wastewater to the city of Madison's treatment facility, which disposes effluent in a different watershed. This arrangement was instituted in the early 1990s to replace Verona's local wastewater treatment plant, which was approaching capacity. The regional tie-in only required a pump station and a 2 mile force main to connect to Madison interceptors, at a cost of about $2.5 million, compared to $25-30 million for upgrading and expanding the local plant. However, local residents wanted the effluent returned to the Sugar Creek watershed due to concerns over potential instream flow reductions that regionalization could cause, including possible shoreline level reductions in a pond on Sugar Creek in the central park of a downstream community. A 10-mile long, $4.5 million line and pumping system was built to return up to 3.6 mgd to the local watershed. Annual pumping costs are about the same as they would have been had disposal been made via Madison's usual outfall, which also requires pumping. Fisheries specialists at the Wisconsin Department of Natural Resources have raised some concerns that the treated effluent might raise ambient temperatures in Sugar Creek, to the detriment of spawning trout, and monitoring of the effluent and stream is now underway to determine if this is the case. In the meantime, only 2.2 mgd of the 3.0 mgd wastewater flow generated from the Sugar Creek watershed is returned to the watershed. Also of note, within Madison's own Yahara River watershed, the wastewater system was built to avoid discharge to a series of four lakes on the river in the city. Given growth in the watershed upstream of the lakes, projections show that in the not-too-distant future there may be no flow through the lakes during years of low precipitation (Schellpfeffer 2000).

Metropolitan Boston, Massachusetts: Over the years, a highly regionalized wastewater collection system has been built to serve the greater Boston metropolitan area. The Massachusetts Water Resources Authority now provides sewer service to most people in the region, treats sewage at its Deer Island wastewater plant, and disposes the effluent via an ocean outfall. MWRA provides water service from a distant, western Massachusetts reservoir to a smaller portion of the region's population. Many towns have their own local water supplies, mostly from ground water. This configuration of the supply and collection systems results in much locally withdrawn water being sent out of local watersheds for treatment and disposal. Further, I&I to sewer lines is removing water from local watersheds. These mechanisms of interbasin water transfer significantly impact three major watersheds in the region, those of the Ipswich, the Charles, and the Neponset Rivers.

In the Ipswich River basin, local supply withdrawals coupled with the regional wastewater collection system have actually resulted in the river drying up in some summers, producing significant fish kills. Yields of local ground water wells may have been reduced as well. Flows in the Charles River are also reduced, with negative impacts on water quality. Sub-regional municipal wastewater collection and treatment systems, which bypass long reaches of the Charles, contribute to this, as well as the regional wastewater collection system (Zimmerman 2000).

For the Neponset River watershed, some rough figures are available to indicate the extent of the problem. 300,000 people live in the watershed. About two-thirds get some portion of their water from sources in the watershed, mainly municipal ground water wells. One-half to two-thirds are served by the MWRA regional sewer system. Reductions in Neponset River flows are causing rising water temperatures and reduced dissolved oxygen, weed growth from increased sunlight penetration to the river bottom and mobilization of nutrients, and drying out of wetlands. The Neponset River Watershed Association has recently calculated that roughly 9 million gallons of water per year are transferred out of the basin by the regional sewer system. This amount is approximately equivalent to 20 percent of the Neponset River's annual flow. The basin's hydrology is not well enough understood for a determination of how much of the transferred water would otherwise have reached the river, but the 20 percent figure is cause for concern. The MWRA took another approach to assessing the magnitude of the regional sewer system impact. It calculated that sewer system transfers of water out of the basin were equivalent to only 10 percent of the total annual rainfall on the basin. This seems a small figure, but when one considers that 40-50 percent of annual rainfall is transferred back to the atmosphere via evapotranspiration, the relative importance of the 10 percent figure increases. Several mechanisms produce the interbasin transfer, according to the watershed association's study. Transfers of water via local withdrawal and supply system distribution to other watersheds are a small portion. So too are transfers of locally withdrawn water as sanitary sewage via the regional collection system. The bulk of the transfer is due to collection system I&I. The municipal systems feeding into the MWRA system are quite old and many are in poor repair. Six of the top 10 towns for high I&I rates in MWRA's entire service area are in the Neponset basin. I&I rates of 75 percent occur in some municipalities. Watershed-wide, 60 percent of the entire flow out of the basin in the MWRA system is I&I water. Of this, infiltration is the largest source, accounting for approximately 50 percent, while inflow accounts for 10-12 percent (Cook 2000).

[5] On *Block Island*, off the coast of Rhode Island, improvements to decentralized systems are seen as important not only to protecting the quality of ground water, but also as a way to recharge the island's sole source aquifer and prevent saltwater intrusion, along with water conservation efforts (Joubert et al. 1999).

Recent policy initiatives in *Massachusetts* are beginning to address interbasin water transfers via sewer systems. The state's Interbasin Transfer Act, initiated due to large scale surface water supply developments proposed in the 1970s, has recently been applied to proposed development of a municipal ground water well in Canton, a town in the Neponset watershed which is on the MWRA's regional sewer system. The act calls for implementation of "all practical alternatives" before transfers are allowed. Localized wastewater treatment, including decentralized systems, could support this standard. Also, the Massachusetts Department of Environmental Protection has recently developed new guidelines for sewer facility planning in its “Comprehensive Wastewater Management Planning Process.” The guidelines identify a range of options for evaluation when sewer extensions are planned, including leaving some areas on onsite systems or going to cluster systems (Cook 2000, Lyberger 2000).

[6] For example, the Cedar Grove Cheese factory in Plain, Wisconsin built a greenhouse-based sewage treatment plant for its effluent in 1998. Cedar Grove chose the 25 m3/day (6500 gal/day) facility as the most economical alternative available, but it also realized unexpected economic benefits. The greenhouse ecosystems are quickly and noticeably impaired when a spill of milk or whey gets in with the normal effluent. The crises in the wastewater treatment plant made the management of Cedar Grove much more aware of the number and quantity of spills of these valuable liquids. Each spill was quickly followed up by inquiries to find out what had caused it and what could be done to prevent similar accidents in the future. The savings in spills avoided are not possible to quantify precisely, but a clear trend of reduced numbers and quantities of spills is visible (Wills 2000; Miller 2000).

[7] A nearby dormitory for students at the Agricultural University of Norway was recently built using vacuum toilets to collect the blackwater in a holding tank, from which it could be hauled for treatment in a specially developed reactor and land application. The vacuum pump that this system uses is capable of handling hundreds of toilets, and transporting the waste hundreds of meters. Since the vacuum pipe is both much smaller than a sanitary sewer pipe and does not have to be buried as deep, there has been discussion of renovating the sanitation system by building a vacuum-based cluster collection for the blackwater, with the graywater still going to the existing sewer pipes.

It would be possible to add this blackwater to the sanitary sewer system at some point where the pipes are in better repair. Because the vacuum toilets produce a very concentrated blackwater—they use about one liter of water per flush—the water might have to be diluted at the point of addition to avoid backing up in the sewers. On the other hand, its concentrated nature suits it well to being trucked to a reactor on a farm and applied as fertilizer, after stabilization.

As Ås periodically experiences summer water shortages that lead to restrictions on watering, the residents may even choose to invest in onsite graywater treatment that allows reuse on the lawn and in the garden.

[8] *Unit size.* Can capacity by bought in small increments, or only in large, “lumpy” investments? Decentralized systems are highly modular. They can be added house-by-house as growth occurs. Or, as with centralized plants, cluster systems can be planned to accommodate growth with additional modules, for instance, incremental addition of sand filter beds—but the module capacity can be much smaller than with conventional centralized treatment systems.

Unit lead time. How long does it take to plan and build each new unit of capacity? In general, lead time is correlated with size. In the case of large centralized plants, planning, permitting, and construction can take years, in part because each plant is unique in important engineering parameters. For decentralized systems, uniqueness—proper accounting for soil type, for instance—is still a feature of many of the technologies, but the engineering usually, the permitting typically, and the construction generally can be completed in short order. This will become even more the case with improvements and cost reductions in decentralized technologies that are less reliant on soil conditions.

Unit flexibility. Does the size or type of unit lock one into a particular technology for future units? Because decentralized systems perform independently of each other, new technologies can be chosen at any point in the future for growth occurring from that point forward. For centralized plants, if new standards requiring new technology are put in place before the plant is near capacity, the unused capacity may become a stranded asset, or costly retrofits may be needed to make the yet-to-be-used capacity serviceable.

Peer Reviews

The preceding White Paper, *Economies of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits*, by C. Eitner and R. Pinkham was solicited for peer review. Reviewers comments are provided in this section.

Christopher D. English, P.E.
USDA Rural Development, Minnesota
Andover, MN

Section 1 – Abstract

- Minor editorial: (p1, s1) Add “protection” after “environment”; (p1,s1) Replace “risks” with “benefits/risks.”

Section 2 – Introduction

- (p2, general) Presumption that direct benefits of wastewater treatment are primarily “non-monetary” is only half correct. Direct financial benefits occur for the community through increased commercial, industrial, and residential development. Individual property owners also see an increase in property value.
- (p3, general) Concept of who pays costs and who benefits is extremely complicated with wastewater since so much public money is available. It is difficult to determine who benefits from subsidizing commercial entities if they employ residents in the community and add to the tax base.
- (p5, general) In some cases, the value of a property goes from zero to something if proper wastewater treatment can be obtained. This is the case when local government has prohibition on transfer of title or a piece of undeveloped lake property can’t be developed without alternative onsite technology.
- (p6, general) Other benefits include the ability for local government to control growth, zone and preserve a “rural” environment.
- (p7, general) A community’s decision to choose centralized wastewater over decentralized has a lot to do with consulting engineers and their motives. Centralized wastewater systems provide for a greater return on investment to a firm based on the economy of scale. For example, it is cheaper to design a stabilization pond than 10 decentralized pretreatment systems. These higher costs carry through to construction monitoring, testing, and surveying. Another factor influencing a city’s decision on this is education. Finally, what I call the “step child syndrome” plays a huge role in a community’s self perception of worth. This ties directly into paragraph 10.

Section 3 – Benefits and Costs at the Micro-Scale

- (3.1.2) In general, your planning period should not be less that your financing period.
- (3.2, p3) I completely disagree with this statement. The concept of high costs is derived from the fact that most onsite system owners spend nothing on maintenance. If one where to perform routine maintenance and account for saving through increased system life, proper maintenance at a professional level is affordable. Economies of scale are available through REAs or WQCs. In addition, remote monitoring can reduce O&M costs substantially. For example, prior to automation, a person had to visit every home and property to read an electrical meter. This cost, in terms of overall service, was very small due to the large customer base.
- (3.2, p5) Again, I disagree. For example, a homeowner on a lake with a failing system once told me the "the economy of scale to fix the problem is to high." In effect, when you compare "do nothing" to "do anything," there is no economy of scale, it is infinite. One must first determine the baseline cost for properly maintaining a system in that particular area (i.e. lake, tight soils, high groundwater) and then compare that to professional maintenance.
- (3.2, p6) The issue of remote monitoring must be better defined. What is to be monitored—everything? If so, then the costs will be very high. Can statistical models be developed to meet regulatory needs which allow for a limited number of data points and, therefore, equipment?
- (3.2, p6) Note: The Stearns County Project will look at this very issue.
- (3.3.1, p1) The issue of intrusion can be addressed through education. I like to compare it to the electric meter reader. Most people accept this minimum intrusion on their privacy and I expect the same will be true for wastewater as long as there is a perceived value.
- (3.3.2, general) One must differentiate between undeveloped land and land which has already been developed. The cost for providing wastewater is different under these two scenarios. More economic benefits are available to the developer if high density development is allowed due to centralized wastewater. This is a profit motive. If the properties are developed, the costs go to the homeowner and are viewed as a loss and not a profit.
- (3.3.3, general) Add disinfection, signage, and fencing to list of costs.
- (3.3.3, general) Combining storm water retention with wastewater disposal and using dry hydrants for fire protection. Additional offsetting costs available.
- (3.5, general) Due to the existing prescriptive nature of onsite codes, density of development is driven by soil conditions and not via risk assessment. For example, very good soils allows for higher density development and close spaced domestic water wells. However, better soils with higher conductivity allow for more rapid transport of pathogens and contaminants.
- (3.5, general) The monetary and non-monetary costs of advanced treatment can also be offset with the benefits of smart growth and sustainable development. Impacts such as conversion of valuable farm land or other resources must be accounted. Again, costs to individuals are high but must be viewed from a larger societal perspective. The question as to who pays is still up for debate.

- (3.5, general) Economies of scale for public services directly correlate to development density and tax base. Costs for public services such as garbage collection, schools, mass transit, policing, etc., are higher in high density communities which have a proportionately lower tax base vs. unit land area.
- (3.5, general) I realize that this is an analysis of "micro scale" impacts, however, I once again argue that an economy of scale exists for responsible management. It is intuitively evident that there are costs for maintaining an individual wastewater system and that these costs can be lowered if several homeowners work together. For example, most septic hauling companies will charge a lower rate for contact users vs. individual homeowners who call every ten years or so.
- (3.5, general) Factors which may forestall technology advancements include both market forces and the current lack of regulatory enforcement. As long as there are no direct costs to the individual homeowner for non-compliance, then nothing will change. Other market forces include manufacturers and haulers who don't what to lose business.
- (3.5, general) One other factor forestalling decentralized treatment is the political pressures for cities to annex adjoining developed areas. Utilities and access to them are often viewed as tools to entice orderly annexation. Cities must find ways to recuperate their costs for construction of large centralized facilities in the face of outward migration and loss of tax revenues.
- (3.6, p7) A performance based code will almost certainly require some type of professional supervised O&M. This will be due to the varied abilities of individual homeowners and the community's interest to protect itself. I argue that the most common failure mode after bad design is poor or no maintenance. Once performance based systems are allowed, the design issue will be addressed and the predominant failure mode will be O&M.
- (3.6, p8) This again is the argument between prescriptive designs which rely on soil permeability and not on risk assessment of natural resources such as the groundwater. This again argues for professional design and management of performance based systems.

Section 4 – Benefits and Costs at the Macro Scale

- (4, general) I can't agree enough!!! Bravo! However, this also shows the need to develop funding mechanisms for planning which are almost nonexistent.
- (4.2, general) Replacement costs are an extremely important component of the economic evaluation and is typically ignored in the small city facility plan due to requirements of funding agencies.
- (4.2, p2) Per capita treatment costs decrease with increases in population. A larger facility for the same population will not increase the economy of scale. I know this is the point you wish to make but it is not clear.
- (4.2, p3) Recommend mentioning the economy of scale for collection as it relates to development density and distances between developed areas.
- (4.3, p4) Again, it is not clear that an increase in facility size relates to an increase in population. Also, development density will have an impact on O&M costs.

- (4.2.2, general) I recommend a fourth research area—Rate Analysis and Affordability. Mandatory rate increase and replacement funds should also be explored.
- (4, general) Many of my above comments are being addressed quite well in this section. Recommend including this information earlier in paper.
- (4.2.3.2, general) Current funding mechanisms promote over-design and subsidization of new development. This is another argument for quality community planning. I argue that decentralized systems allow for more flexibility in preventing unwanted subsidization and controlling growth patterns.
- (4.2.3.3.3, general) Dilution may not be the solution!
- (4.2.3.3.3, general) Other research may include the benefits of decentralized drinking water treatment and distribution where the resource is extracted, used, treated, and recycled within a very small geographic area.
- (4.2.3.4, general) Solids handling, processing, stabilization, and disposal costs are a major part of the budget for any centralized wastewater facility. These costs must be compared with decentralized facilities and reuse of bio-solids. In other words, there may be more benefit than just the $5.00 per capita per year. One cause of this cost is the availability of space (tankage) at a centralized facility for storage and stabilization of solids. Therefore, processes must be accelerated to reduce detention time. Decentralized systems provide for a larger storage capacity spread out over a larger geographic area which, in turn, allows for a slower, more passive stabilization of solids. In addition, costs for transporting the solids is cheaper since you only do it once.
- (4.2.4, general) Anecdotal evidence exists in the EPA Report to Congress and other sources on the impacts of public funding on centralized versus decentralized treatment.
- (4.2.5, general) I believe that management of all onsite systems is viable when incorporated into an existing utility structure and risk based costs are evaluated. Other value added capabilities of REAs (RMEs) are low interest financing as well as availability of other services (electricity, phone, cell, etc.).
- (4.2.5.2.2, general) Utility districts can "leap frog" geographic areas to avoid political and financial risks by using decentralized technologies. This allows for revenue production while planning and negotiations occur in areas of higher sensitivity.
- (4.2.5.2.3, general) Funding agencies with large portfolios would like to avoid having too much "inventory" on hand which is degrading in value over time. Typical scenario—a large plant is built with future expansion in mind. When that expansion occurs, reevaluation of permits allows for regulators to change or modify discharge standards causing additional costs in modification of the facility.
- (4.3, general) Engineers must be convinced that their exposure to risk is equal or compensated somehow if they are to propose decentralized systems. Current perception is that decentralized systems are too "risky."
- (4.3, general) It seems that some of these models and decision tools are meant to supplement the lack of understanding consultants have when it comes to distributed and alternative technologies and risk assessment. What can be done to improve the education of practicing engineers and students? Research into impacts education will have is recommended. Even

after education, decision making tools are available. However, one must overcome the "comfort level" issue.

Section 5 – Summary and Prioritization of Research Notes

- (5, general) Again, I recommend research into users rates and public subsidization of utilities.

John Herring
Coastal Program
New York State

As a first principle, I have assumed that the research agenda should be heavily focused on issues which are likely to have impacts on management, and in general the more direct the impact, the higher priority the research.

Significant progress in reducing the adverse environmental and public health impacts of onsite or decentralized wastewater treatment is largely dependant on addressing the issue of maintenance. The conventional system of a septic tank and leach field is, under suitable circumstances, very low maintenance. However, it is frequently considered to be “no maintenance” by the owners and operators, at least for residential systems. While there are certainly examples of successful maintenance programs, they represent only a very small fraction (probably significantly less than 10%) of the existing systems nationwide.

While it is clear that under ideal circumstances, a variety of alternatives such as aerated systems, clusters, etc. can be more effective than the traditional single system, many responsible governmental agencies have thus far declined to certify such systems for use in any but the most extreme situations. Virtually all such alternatives require a higher level of maintenance than the traditional system. Given the paucity of institutional arrangements to assure such maintenance, regulatory agencies typically decline to allow systems which are a) less likely than traditional systems to be adequately maintained and b) more likely to fail if not adequately maintained.

If the maintenance issue is not addressed, then, it is difficult to conceive of technical advances which will have any actual impact on the problem. As an example, consider efforts to allow remote monitoring. Assume a 100% effective and inexpensive remote monitoring system is developed. Under what circumstances would it be used? For existing systems, not under any current obligations to manage, such a system wold pose an initial cost, however low. At best, the cost would result in no additional costs or benefits because no problems would be detected. The only alternative is that the homeowner would be notified of a problem. In the absence of other funds for remediation, the homeowner would either have to a) ignore the warning, obviating any benefit of the monitoring system, or b) repair the system, adding to the costs of the monitoring system. At best, if the monitoring system detected a problem before it was detectable by other, less capital intensive means (odors, surface discharge, etc.), it might allow repairs to be done at a lower cost than would have otherwise been the case. However, routine maintenance of the type explained by such organizations as Cooperative Extension and several National Estuary Programs would serve the same function, at a very low cost. However, evidence of the effectiveness of such education programs is not overwhelming, to say the least.

Similar arguments obtain for various other research topics relating to new technologies.

If the primary barrier to improved management of onsite systems is maintenance, research needs can be organized around several facets of that issue. For example, a study currently underway in New York State examines the various approaches to management in place (special districts, intermunicipal agreements, watershed rules and regulations, education programs, etc.). There is currently an attempt being made to categorize these examples, a step preliminary to analyzing why the various alternatives are not adopted on a more widespread basis. This type of social science research, focusing on landowner perceptions and attitudes, will help in determining the characteristics of a management system which is likely to be accepted. It may well be, for example, that cluster systems are sufficiently different from traditional single lot systems in public perception that the creation and maintenance of some sort of managing entity will be perceived as acceptable, while deeply held values regarding private property preclude acceptance of such management for conventional systems.

Given the centrality of the issue of maintenance, it is clear that an area needing significant research is that of institutional costs. The White Paper clearly distinguishes between micro- and macro-economic factors: that is, between costs and benefits from the perspective of the individual landowner and the society as a whole. However, it should be remembered that the various possible institutional arrangements aimed at improving maintenance may have quite different impacts on the responsible public authority. To take an extreme example, many of the "case studies" which are cited to demonstrate the efficacy of particular management authorities have involved substantial transfer payments in order to obtain landowner participation. Such programs are at best of marginal use in that if the objective is widespread adoption, there is no "outside" to provide the transfer payments. If all that is anticipated is the protection of a few specific areas, presumably of critical value, the question arises as to the overall importance of the program. Also, in such areas, the landowners are often far more able to afford pollution rededication than the average member of the public. Again, the issue of distribution of costs and benefits is relevant.

Thus far, I have focused on the need for addressing maintenance of existing systems, as the clearest and most immediately pressing issue. However, the maintenance issue also affects new systems. As noted above, in the absence of maintenance programs, regulatory agencies are less likely to approve alternative systems. Thus, lots which are not suitable for conventional systems may not be developable at all. The White Paper discusses the impacts of such a situation on property values, etc. I suggest that this discussion recognize the relationship to maintenance issues.

A related issue, which may or may not be properly considered as a research topic, is even more heavily weighted to the social sciences than the maintenance issue. That is the "meta-issue" of the extent to which we should be focusing on altering management of wastewater. In general, the adverse impacts of inadequate decentralized wastewater management are felt locally, either through environmental or public health impacts. Absent compelling evidence of impacts in these arenas, there is little justification for the allocation of scarce resources to improving decentralized wastewater management. Thus, high priority should be given to research aimed at establishing the actual threats to public and environmental health from this source. If, after adequate investigation, it appears the threats are significantly smaller than other analogous problems such as pollution from agricultural activity, the issue should be considered a lower priority, with research and remediation funding adjusted accordingly.

Following are more specific comments, keyed to the May 9 review draft:

- P2 Micro-scale costs and benefits – notes “off-site public health ...” issues. Should include onsite health impacts. Might want to consider phrasing last sentence of section in terms of option value.
- P2 Macro scale costs and benefits – paragraph 1– should include public health as a benefit.
- P3 Under types of questions – Note the fact that in general, public funds are far more available for public works than those benefiting private citizens. As a practical matter, this can cause economic distortions in that sewers might be eligible for significant public financing, while decentralized and in particular private approaches are not.
- P3 Paragraph beginning with “Finally, it is important.” There is some research which does not agree with the assertion that “It is much easier to make changes... when homeowners and other stakeholders understand what problems there are...” L. Wagenet work showed a case in which with increased understanding, survey respondents were less likely to maintain their private systems. While it is likely the particular political situation involved (New York City watershed) affected attitudes, the efficacy of education of homeowners as a method of significantly improving management is in many respects still an open question.
- P4, 1.1.1 Note the effect of specific site conditions on cost. Situations such as steep slopes and the need to pump waste uphill to an area flat enough for a leach field can mean that costs for systems in the same town can vary from $3000 to $50,000, as is the case in the Finger Lakes region of New York State.
- P4, 1.1.2 Costs include routine pumping of septic tanks – Are there any figures which indicate that this is in fact routinely done in most instances? If it is not (and I believe it is the exception rather than the rule) including it in life cycle costs overstates the cost of a conventional system. This is particularly important given the assertion that “Maintenance and repair costs are a significant portion of the present value figures.”
- P5, 1.2 “Given the high costs of long term-professional inspection...” Does this imply a possible role for education programs aimed at homeowner inspection?
- P6 Research project-failure rates – Highlight the need for definitions of failure.
- P6 Research project – Benefits and costs – Note that to answer this, it is necessary to determine the health (public and environmental) impacts of decentralized systems.
- P6 Might it be useful to construct a research program aimed at addressing a) failure rates, b) factors affecting failure rates (slope, soils type, use of septic “cleaners”, etc.), and c) ways to predict failures?
- P 7, 1.3 Research project – effective education campaigns – Should note need to determine the effectiveness of education campaigns in changing behavior. Such programs are rarely evaluated by that criterion, but more usually by such criteria as number of individuals contacted, self-reported changes in attitudes, etc. These are only partial surrogates for the question of whether the programs actually induce behavioral changes affecting system management.
- P7, 1.3.2 Paragraph 1. Note that in some cases (e.g., much of Long Island, NY) zoning density was largely based on assumptions involving groundwater loading of pollutants (especially nitrogen) from septic systems. Thus, those seeking to limit density may actively discourage improved pollution removal efficiencies.

- P8, 1.3.2 End of 1st paragraph – Suggest replacing “increase” land value with “affect.”
- P9, 1.4 Note that in general, as options increase for homeowner, with consequent overall decrease in costs for homeowner, there is a concomitant increase in costs for the managers (regulators). At some point, the increase social costs can outweigh the benefits to the individual homeowner.
- P 12, 1.6 Research project – Model Code development – see comment directly above.
- P12, 1.6 Research project – performance based codes – good point.
- P13, 1.7 Paragraph 3 – “A proper comprehensive water quality protection plan...” perhaps true, but the costs of such plans, if applied to more than a few selected areas, would be astronomical. Suggest comparison with any state’s estimate of costs to implement TMDLs in multiple watersheds, for multiple pollutants.
- P14, 1.7 Paragraph 4 – “...without the proper investigation of sources.” Again, the cost issue must be considered. In essence, there are 2 basic approaches which might be used. The first approach would argue that reputable levels of scientific proof should be reached before action is taken. this is the approach often taken by the property rights community. However, it does not take into account the costs of reaching such a level of confidence, in particular in situations with multiple potential sources of pollution, temporal or spatial variability, etc.

  The alternative approach is exemplified by the Coastal Nonpoint Pollution Control program enacted under the Coastal Zone Act Reauthorization Amendments (CZARA) of 1990. That program essentially asserts that sufficient research has been done to allow the induction that certain activities inherently pollute and that certain management practices reduce pollutant loadings. Thus, the CZARA program simply takes as a given that the runoff prom, say, parking lots, contains a variety of pollutants, and that such management practices as street sweeping, sedimentation basins, and buffer strips will reduce pollutant loadings to watercourses. Before dismissing such an approach as an “oversimplification,” it might be instructive to estimate the costs of determining actual pollutant loadings to the thousands of water bodies affected by such national programs.
- P14, 1.7.1 Paragraph 2 – Should note role of groundwater recharge. In some areas, this is an important factor in use of onsite wastewater disposal. Long Island, New York, experienced detectable lowering of groundwater levels once central sewers were put in place a few decades ago. the effects were sufficient to justify the development of a Flow Augmentation Needs Study (FANS), aimed at ameliorating impacts to stream baseflow and in particular to the effects on fish and wildlife which resulted from the reduction in such baseflow.
- P15, 1.2.1 Operating and maintenance costs – “O & M is thought to be more expensive...” What level of O&M for decentralized systems does this assume—recommended or typical real world?
- P17 Cluster systems – Is the assumption here that this is for new construction or as replacements of failed onsite systems? If the former, the issue of regulatory acceptability needs to be addressed. Also, the issue of public acceptance should be noted.
- P19 Research project – impacts of lot sizes. It should be noted that a complicating factor, which may often override the issues discussed here, is consumer preference. Might it be appropriate to propose initial research on whether such information would make a

difference? In many instances, given a choice between moderately sized individual lots and small clustered lots with neighboring undeveloped areas, most consumers preferred the former.

- P21, 1.2.3.3.3 Mitigation of hydrologic problems. See notes regarding the FANS study, above.
- P22, 1.2.3.4 Paragraph 3 – note the need to address pathogen issues.
- P23, 1.2.3.5 Paragraph 2 – " They only need to be tested before collection." Note that given the existence of many individually small systems, testing costs can mount rapidly. Also, for residential septic systems, it is unlikely that there would be high levels of such pollutants as heavy metals. Perhaps testing could be minimized through a random testing approach, to discourage deliberate illegal disposal. However, any such testing program will require an institutional framework for maintenance.
- P25, 1.2.5 End of second paragraph. "...and may not be appropriate." Does this mean "would not be supported by residents?" If so, assumption that utility-like approaches are inappropriate in less intensively developed areas needs to be defended.
- P26 Blackwater separation to extend life – Note that such systems can also be used as a compromise to improve wastewater management and protect private wells in older communities in which septic tanks are routinely failing but the costs of developing a collection system for a conventional sewer are prohibitive. Narrowsburg, NY, for example, uses existing septic tanks as essentially primary settling tanks. the effluent is then collected by small diameter pipes and treated at a conventional WWTP. This effectively addressed the problems for an older community in which lot sizes precluded traditionally sized leach fields.
- P 27, 1.2.5.2 Paragraph 2 – Should note public policy argument. Society may be willing to subsidize certain risks as a means of achieving other policy goals.
- P 27, 1.2.5.2.1 Paragraph 2 – Example is somewhat misleading. While the incremental capitalization burden is 1% rather than 33%, the capacity added is also different. Make explicit the concept that if large changes in capacity are required, the two approaches may be comparable, but that if there is a need for only a small increment, or if increments are desired continually over a period of years, the lumpiness issue is important.
- P28, 1.2.5.2.3 Paragraph 1 – "...will use the latest, most competitive technology." Note that this assumes the latest technology will not turn out to be ineffective. More importantly, there seems to be a research need regarding the social components of the trend towards "smaller technologies along with distributed intelligence and control." Again, these require management expertise, with a consequent institutional framework which at present does not exist.
- P29 Second bullet – "It may only be necessary to retrofit a small number of units..." Note that there is still the problem of determining which systems need to be retrofitted. The cost of such determinations can be considerable.
- P 29 Same bullet – distinction between watersheds and sewersheds. Why is this important?
- P33, 1.4 Research project 2. Case studies are only one approach. This section should be expanded.

- P34, 4 Summary...

  Obviously, I believe the priority projects should be determined based on the factors discussed at the beginning of these comments. In particular, I believe that nothing should be considered a priority unless it bears on the issue of maintenance, at least in the near term. A further criterion should be likelihood of impact on real world management. While this would unfortunately mean that many interesting issues which might be important in some circumstances would drop out, the purpose of the exercise is to determine top priorities.

REVISED AGENDA
RESEARCH NEEDS CONFERENCE
ST. LOUIS, MO MAY 19,20, 21 – 2000

TIME	TOPIC	PRESENTER
	DAY ONE (FRIDAY)	
8:00 - 8:30	Introduction	Tom Yeager/Valerie Nelson
8:30 - 9:30	Integrated Risk Assessment/Risk Management as Applied to Decentralized Wastewater Treatment	Dan Jones, ORNL
9:30 - 10:30	Performance of Soil Absorption Systems	Bob Siegrist, CSM Jerry Tyler, Univ. of WI Peter Jenssen, Univ. of Norway
10:30 - 11:00	BREAK	
11:00 - 12:00	Fate and Transport of Pathogens	Dean Cliver, UC – Davis
12:00 - 1:00	LUNCH	
1:00 - 2:45	Performance of Soil Absorption Systems	Review Panel & Audience Aziz Amoozeqar, NC State James Converse, Univ. of WI
2:45 - 3:15	Break	
3:15 - 5:00	Fate and Transport of Pathogens	Review Panel & Audience Chuck Gerba, Univ. of AZ Marylyn Yates, UC – Riverside
	DAY TWO (SATURDAY)	
7:45 - 8:45	Fate and Transport of Nutrients	Art Gold, URI Tom Sims, Univ. of DE
8:45 - 9:45	Economics/Social Issues	Carl Eitner, Univ. of WI Richard Pinkham, RMI Valerie Nelson, CAWT
9:45 - 10:00	BREAK	
10:00 - 11:30	Fate and Transport of Nutrients	Review Panel & Audience Ray Reneau, Virginia Tech W.D. Robertson, Univ. of Waterloo
11:30 - 12:00	Lunch	
12:00 - 1:30	Economics/Social Issues	Review Panel & Audience Chris English, USDA MN John Herring, NYS DOS - Coastal
1:30 - 3:00	Prioritization of Research Needs	Audience

DESCRIPTION OF AGENDA

The agenda is set up in order to maximize audience participation in a short time frame. Issues to be addressed in Day 1 include:

- Concepts of Risk Assessment/Risk Management
- Performance of Soil Absorption Systems
- Fate and Transport of Pathogens

Issues to be addressed in Day 2 include:

- Fate and Transport of Nutrients
- Economics/Social Issues
- Prioritization of Research Needs

Initially on each day, the white paper authors and their co-authors will present their papers and answer questions from the audience relating to clarification of the white papers. After the papers have been presented the review panel will present their review comments and input will be solicited for the prioritization of the research topics presented for a specific subject.

At the end of the second day, the audience will be asked to assist in the overall prioritization of the research topics previously defined. The Project Steering Committee (PSC) will continue their prioritization process during their meeting on Sunday.

The audience will be given the opportunity to submit written comments regarding prioritization of the research needs topics until June 1st. These comments should be sent to Andrea Arenovski (a_arenovski@earthlink.net) who will collate them and transfer them to the PSC. The PSC will then set the final prioritized list of research projects.

Decentralized Wastewater Management - Research Needs Conference

Attendees

First Name	Last Name	Affiliation	City	State
Aziz	Amoozegar	N. Carolina State Univ. College of Agriculture & Life Sciences	Raleigh	NC
Damann	Anderson	Ayres Associates	Tampa	FL
Andrea	Arenovski	Nat'l Decentr. Water Resources Capacity Development Project	Oakland	CA
Keith	Carns	EPRI Muni Water/Wastewater	St. Louis	MO
Kevin	Chaffee	Earthtek Environmental Systems	Batesville	IN
Kim	Choate	Tennessee Valley Authority	Chattanooga	TN
Dean	Cliver	Univ. of California - Davis	Davis	CA
James	Converse	Univ. of Wisconsin - Madison	Madison	WI
Stephen	Dix	Infiltrator Systems, Inc.	Old Saybrook	CT
Bruce	Douglas	Stone Environmental	Montpelier	VT
Scott	Drake	East Kentucky Power	Winchester	KY
Ray	Ehrhard	EPRI Muni Water/Wastewater	St. Louis	MO
Christopher	English	USDA Rural Development	St. Paul	MN
Carl	Etnier	Univ of Wisconsin - Madison	Madison	WI
Chuck	Gerba	Univ. of Arizona	Tucson	AZ
Arthur	Gold	Univ. of Rhode Island	Kinston	RI
Mark	Gross	Univ. of Arkansas	Fayetteville	AR
Sara	Heger	Univ. of Minnesota	St. Paul	MN
John	Herring	NYS DOS - Coastal	Albany	NY
Petter	Jenssen	Univ. of Norway	Aas	NOR
Dan	Jones	Oak Ridge National Lab	Oak Ridge	TN
Jim	Kreissl	USEPA	Cincinnati	OH
Steve	Lindenberg	Nat'l Rural Electric Coop Association	Arlington	VA
Ted	Loudon	Michigan St. University	E. Lansing	MI
Patricia	Miller	Michigan St. University	E. Lansing	MI
Valerie	Nelson	Coalition for Alt. Wastewater Treatment	Gloucester	MA
Richard	Otis	Ayres Associates	Madison	WI
David	Pask	NSFC, NRCCE, WVU	Morgantown	WV
Richard	Pinkham	Rocky Mountain Institute	Arvada	CO
Ray	Reneau	Virginia Tech	Blacksburg	VA
Will	Robertson	Univ. of Waterloo	Waterloo	CAN
Kevin	Sherman	On Site Management Consultants	Tallahassee	FL
Robert	Siegrist	Colorado School of Mines	Golden	CO
Tom	Sims	Univ. of Delaware	Newark	DE
Trent	Stober	Midwest Environmental Consultants	Jefferson City	MO
Jerry	Stonebridge	Stonebridge Constr. Co.	Freeland	WA
Jean	Tribull	Water Environment Research Federation	Alexandria	VA
Jay	Turner	Washington Univ.	St. Louis	MO
Jerry	Tyler	Univ. of Wisconsin - Madison	Madison	WI
Sheila	Van Cuyk	Colorado School of Mines	Golden	CO
David	Wahman	Earthtek Environmental Systems	Batesville	IN
James	Watson	Tennessee Valley Authority	Chattanooga	TN
George	Westall	S.D.S. Company	St. Louis	MO
Leanne	Whitehead	Tennessee Valley Authority	Columbia	TN
Rodney	Williams	Univ. of Arkansas	Fayetteville	AR
Brian	Wrenn	Washington Univ.	St. Louis	MO
Marylynn	Yates	Univ. of California - Riverside	Riverside	CA
Tom	Yeager	Kennedy/Jenks	Palo Alto	CA

Target:

Municipal Water and Wastewater Treatment

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