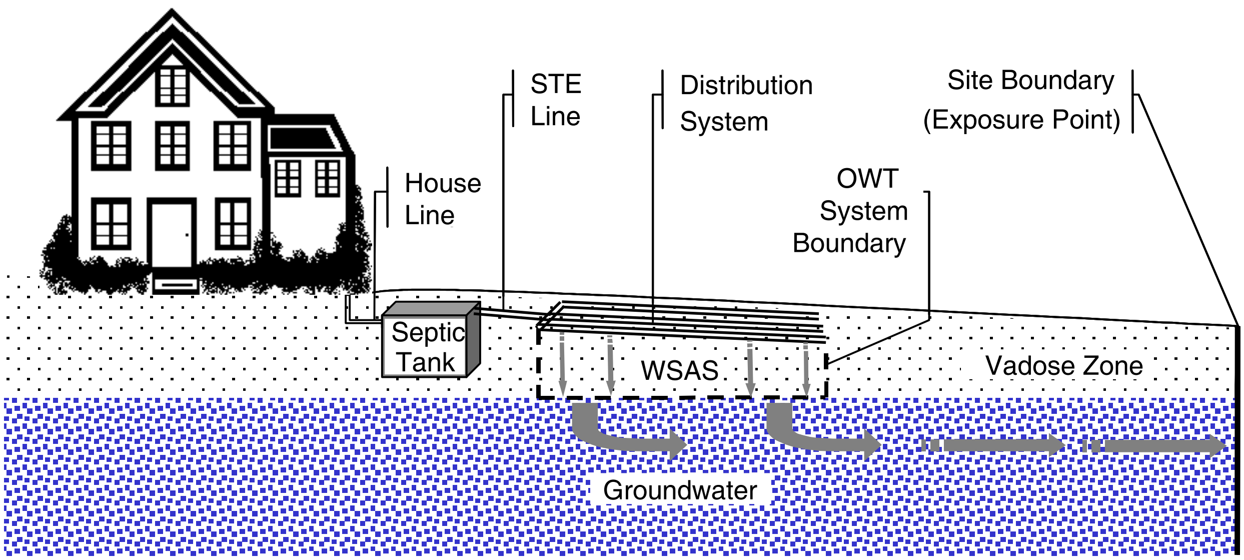
National Decentralized Water Resources
Capacity Development Project



Integrated Risk Assessment for Individual Onsite Wastewater Systems

Oak Ridge National Laboratory
Oak Ridge, Tennessee

December 2004

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Submitted by
Oak Ridge National Laboratory
Oak Ridge, Tennessee

NDWRCDP Project Number: WU-HT-01-18

National Decentralized Water Resources Capacity Development Project (NDWRCDP) Research Project

Final Report, December 2004

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DISCLAIMER

This work was supported by the National Decentralized Water Resources Capacity Development Project (NDWRCDP) with funding provided by the U.S. Environmental Protection Agency through a Cooperative Agreement (EPA No. CR827881-01-0) with Washington University in St. Louis. This report has been reviewed by a panel of experts selected by the NDWRCDP. The contents of this report do not necessarily reflect the views and policies of the NDWRCDP, Washington University, or the U.S. Environmental Protection Agency, nor does the mention of trade names or commercial products constitute endorsement or recommendation for use.

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The final report was edited and produced by ProWrite Inc., Reynoldsburg, OH.

This report is available online at www.ndwrcdp.org. This report is also available through the

National Small Flows Clearinghouse
P.O. Box 6064
Morgantown, WV 26506-6065
Tel: (800) 624-8301
WWCDMG39

This report should be cited in the following manner:

Jones, D. S., R. A. Efroymson, A. Q. Armstrong, M. D. Muhlheim, and S. A. Carnes. 2004. *Integrated Risk Assessment for Individual Onsite Wastewater Systems*. Project No. WU-HT-01-18. Prepared for the National Decentralized Water Resources Capacity Development Project, Washington University, St. Louis, MO, by Oak Ridge National Laboratory, Oak Ridge, TN.

ACKNOWLEDGMENTS

Appreciation is extended to the Technical Advisory Committee for assistance in the preparation of this report:

Charles Gerba, University of Arizona

Arthur Gold, University of Rhode Island

Richard Pinkham, Booz Allen Hamilton

Robert L. Siegrist, Colorado School of Mines

George Tchobanoglous, University of California at Davis

Appreciation is also expressed to the NDWRCDP for their support of this work:

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ABSTRACT

The primary objective of this project was to develop an approach to risk-based decision making for individual onsite wastewater treatment (OWT) systems. The framework of this approach integrates four different interdependent types of risk analyses:

- Engineering
- Public health
- Ecological
- Socioeconomic

For example, the economics of water recreation is linked to the presence of water that is clean enough for drinking and swimming. Similarly, ecological risks depend on the failure rates of OWT systems.

The three stages of risk assessment were used to structure the framework:

- Problem formulation (a planning process)
- Analysis of site-specific exposure and effects
- Risk characterization

A general problem formulation is recommended to define the scope and objectives of the integrated risk assessment for an individual OWT system. Three example systems were selected to represent categories of modern OWT systems:

- Traditional
- Contemporary
- Emerging

Outdated treatment/disposal systems are also included in this framework. The definition of the four OWT categories and the general types of components that are used to represent those systems is based largely on expected effluent quality for each treatment train. The primary stressors addressed by this framework are:

- Pathogens (such as bacteria, protozoa, and viruses)
- Total and specific forms of nutrients (nitrogen and phosphorus)
- Unaesthetic features

- Noxious odors
- Oxygen demand
- Stress induced by both ordinary function (nominal performance) and dysfunction (non-performance)

The engineering risk assessment component framework makes use of Failure Modes and Effects Analysis to address the issues specific to the design and performance of the OWT system of interest. The engineering component framework permits the explicit and transparent consideration of mitigative measures to reduce the risks of system dysfunction.

The public health risk assessment component framework is used to evaluate potential health risks from exposure to wastewater effluent or environmental media that have come in contact with wastewater effluent. The human health property evaluated as the result of exposure to chemicals is systemic toxicity (non-carcinogenic effects). The microbial assessment endpoint evaluated in this pubic health framework is risk of infection.

The ecological component framework is used to evaluate the potential adverse impacts on non-human biota and ecosystems. Two types of surface water ecosystems are distinguished based on differences in prevailing nutrient dynamics: freshwater systems (for example, ponds) and estuarine systems (for example, coastal lagoons).

The socioeconomic component framework is used to evaluate potential socioeconomic impacts and risks from exposure to wastewater effluent or environmental media that have come in contact with wastewater effluent, and efforts to manage those effluents with an OWT system. The socioeconomic component addresses many issues that are typically part of the risk-management process, such as

- Monetary costs of the design and installation or replacement of the OWT system
- Inequities in the distribution of costs/risks among members of the community
- Intrusiveness of regulatory requirements and management
- Aesthetic impacts

In the integrated risk characterization, risks were divided into two general categories: independent risks and conditional risks. Conditional risks are those for which the estimation of risk is conditional on the estimates for one or more other risks. The characterization of conditional risks is based on a variation of the weight-of-evidence process. A rating is assigned to each of the conditional risks evaluated.

EXECUTIVE SUMMARY

The primary objective of this project was to develop an approach to risk-based decision making for individual onsite wastewater treatment (OWT) systems. This framework for individual OWT systems follows the format of *Integrated Risk Assessment/Risk Management as Applied to Decentralized Wastewater Treatment: A High-Level Framework* (Jones *et al.* 2001), which was largely based on the principles and practices of ecological risk assessment (US EPA 1998b and Suter 1993).

Risk Assessment Framework

The risk assessment framework is designed to be most useful to professionals who are trained in the principles of risk assessment. These may include biologists, engineers, or social scientists at regulatory agencies or consulting companies, the latter of which may be hired by regulatory agencies, developers, or industries that design OWT systems. The users who would make the most complete use of this framework would be multidisciplinary teams of risk assessment and OWT design experts. Users of this framework for individual OWT systems may also include technically trained stakeholders, such as county planning commissions with an interest in educating themselves about the factors contributing to biological, engineering, or social risk from individual onsite wastewater systems and methods for their assessment. Entrepreneurs might create user-friendly mathematical models that take advantage of the links among risks in the onsite wastewater system context.

Methods

The methods discussed include measurements and models for retrospective and prospective risk assessments. If the goal is to conduct assessments of proposed new individual wastewater systems (for example, permitting), then the user can focus on the discussions of modeling and ignore discussions of measurement methods for chemicals and pathogens. If the goal is to conduct a risk assessment for existing OWT systems, (for example, to determine the cause of observed illness), then measurement of nutrient or pathogen concentrations may be more important than modeling. If the user is an OWT system designer or the agent of a designer, the user must select generic environments in which the system will be used prior to conducting risk assessments. As an engineer, an OWT system designer may choose to focus on the Failure Modes and Effects Analysis of the engineering component framework, but he or she would need some knowledge of the three other types of risk in order to estimate severity of failure.

Risk Assessment Process

The general risk assessment process consists of three steps:

1. **Problem formulation**—A planning process for generating and evaluating hypotheses about the effects that might occur.
2. **Analysis**—Typically includes both the site-specific analysis or characterization of occurrence or exposure and the more general analysis or characterization of effects (exposure-response relationships). These analyses are interdependent and are typically performed concurrently.
3. **Risk characterization**—The process of combining the estimates of occurrence or exposure with the exposure-response relationships from the analysis of effects to estimate the magnitude and (if possible) probability of effects.

This framework for individual OWT systems is designed to integrate four different types of risk analyses:

- Engineering
- Public Health
- Ecological
- Socioeconomic

The risk analyses are integrated because many of these types of risks are dependent on each other. For example, the economics of water recreation is linked to the presence of water that is clean enough for drinking and swimming. Similarly, ecological risks depend on the failure rates of OWT systems. Therefore, an expert in one risk assessment component discipline would rely on information outside of that discipline to complete a risk assessment for OWT systems. The integration of risk assessment components is accomplished by embedding a component framework for each of these four types of analyses in an overall (that is, integrated) framework (Figure 1).

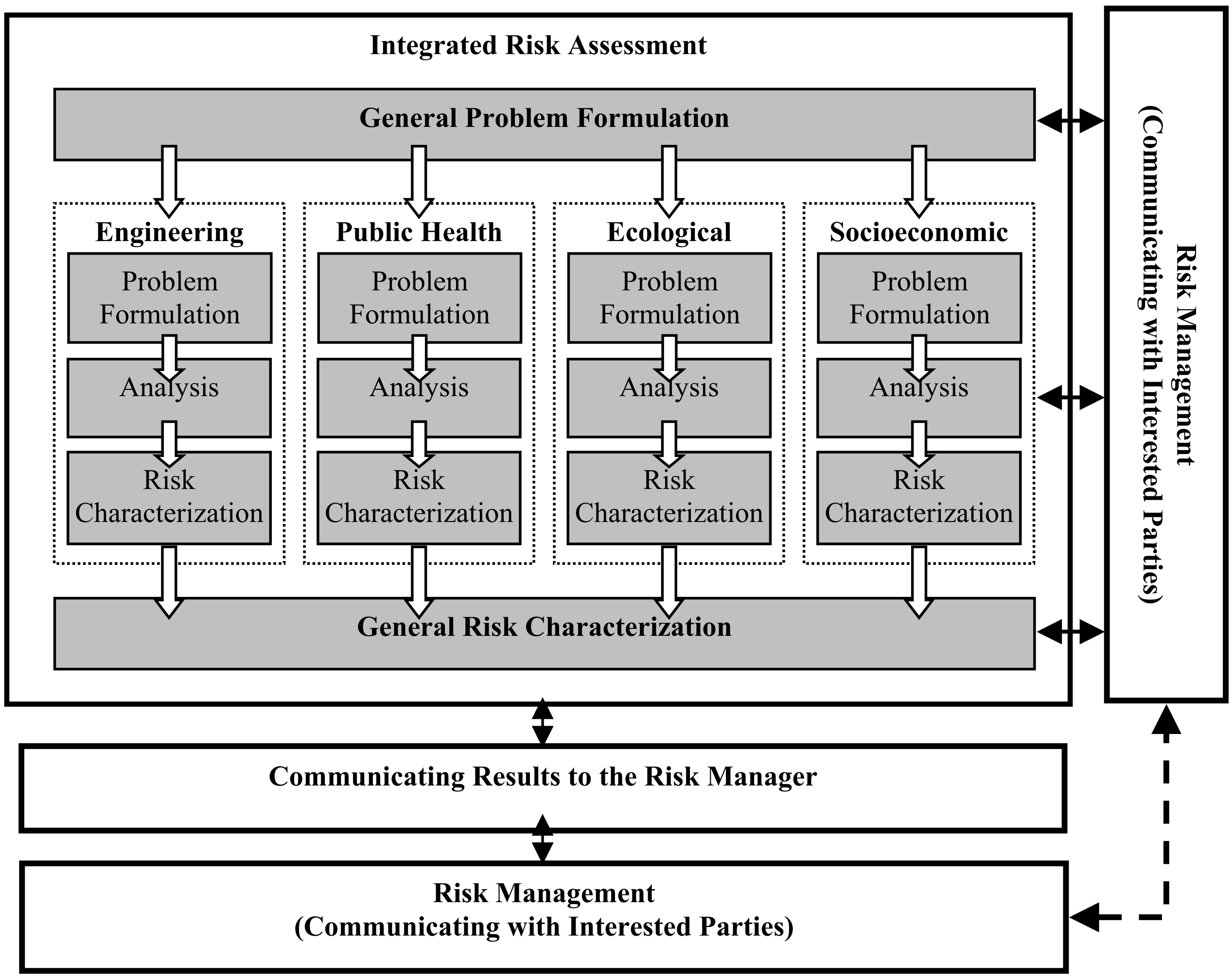

Figure 1
Integrated Risk Assessment Framework for Onsite Wastewater Treatment Systems

Integrated risk assessments can be used to address some of the issues that are typically left to the risk management process. For example, the socioeconomic component of this framework systematically addresses issues that are often addressed *ad hoc* in the risk management process. Examples include:

- Inequities in the distribution of costs/risks among members of the community
- Intrusiveness of regulatory requirements and management practices (for example, property inspections by non-owners)
- Aesthetic impacts (for example, noise, smell, and visual appearance)

Similarly, the engineering component framework permits the explicit and transparent consideration of mitigative measures to reduce the risks of system dysfunction.

This framework was not developed to support macro-scale (for example, watershed-scale) risk assessments for multiple OWT systems. An addendum to this framework is needed for use in addressing the cumulative and emergent effects of multiple OWT systems and offsite treatment systems. Also, the framework was not developed to address benefits of different treatment systems; only the socioeconomic component framework addresses benefits directly. The framework does not support fully probabilistic analyses; only the human health risk assessment component framework (and to a lesser extent, the ecological risk assessment framework) expresses risk in terms of probabilities. Similarly, the framework was not developed to support comparative assessments of alternative wastewater systems. However, it could be used in that context if conservative estimates of exposure and effects, which could bias the analysis toward particular alternatives, are not used. Methods for balancing different types of risks, such as low human health and ecological risk for one alternative system, and high socioeconomic risk for another system, must be developed and applied in the risk management process.

General Problem Formulation

The general problem formulation is a planning step that defines the scope and objectives of the integrated risk assessment for an individual OWT system. This planning step must involve all components of the risk assessment. Three primary purposes for which assessments for individual OWT systems may be conducted are

1. Planning for a new installation on a previously undeveloped site
2. Evaluation of the potential or observed effects of an existing OWT system
3. Evaluation of potential retrofits for a currently failing OWT system

The purposes and goals of the assessment should be aligned with those of decision makers (risk managers). A typical risk management goal is to balance: the risks of endangering public health and reducing local property values due to complete failure of an OWT system (for example, surface breakthrough) against the risk of increased installation and operating costs to the home owner and the risk of eliminating the opportunity for the community to develop the site in question.

The general problem formulation includes the following steps:

1. Description of the spatial and temporal bounds of the assessment
2. Definition of the OWT system to be evaluated
3. Identification and description of the source and the potential stressors and receptors
4. Selection of assessment endpoints (values that are to be protected) with assurance that they can be addressed within the appropriate component assessments
5. Development of a conceptual model for the system to be evaluated
6. Selection of appropriate measures of effects and exposure

These are also steps in the problem formulations for the component frameworks.

The spatial and temporal bounds of the assessment determine what types of stressors and receptors are appropriate. The High-Level Framework (Jones *et al.* 2001) considered two spatial scales—the micro-scale and the macro-scale. The micro-scale referred to an individual residential lot with an onsite drinking water well and an onsite wastewater treatment system. The macro-scale referred to a watershed that contains many individual decentralized systems, as well as other point and nonpoint sources of pollution. The current framework addresses the micro-scale assessment of OWT systems, including areas of potential offsite impact. Macro-scale issues are mentioned in this framework, especially in the context of ecological and socioeconomic risks, many of which tend to be observed at larger scales than the micro-scale.

For OWT systems, the temporal scale of assessment may be based on the time elapsed since the system or component was installed. A risk assessment is denoted as "retrospective" if the goal is to evaluate the potential causes of the current conditions. Prospective assessments are used to estimate potential future risks. A set of default reference points of time elapsed since installation has been selected to illustrate the framework.

Outdated treatment/disposal systems are included in the framework, in addition to three example systems selected to represent categories of modern onsite wastewater treatment systems (Table 1):

- Traditional
- Contemporary
- Emerging systems

The definition of the four OWT categories and the general types of components that are used to represent those systems is based largely on expected effluent quality for each treatment train.

Table 1
Example Treatment System Categories and Components Included in the Framework

System Categories	Major System Components			
	Receiving Tank/ Pre-Treatment	**Tank-Based Advanced Treatment**	**Soil Infiltration/ Vadose Zone Percolation**	**Discharge Point**
Outdated	Cesspool [a]	None	Incidental	Surface soil or water
Traditional	Septic tank [b]	None	Wastewater soil absorption system (WSAS), gravity-fed	Subsurface to Vadose and saturated zones
Contemporary	Septic tank [b, c]	Aerobic Treatment Unit (ATU)	WSAS, reduced sizing	Subsurface to Vadose and saturated zones
Emerging	Septic tank [b]	Porous Media Biofilter (PMB) and Disinfection	Drip irrigation	Shallow subsurface

Source: Adapted from Siegrist *et al.* 2000; Table 1, p. 6

[a] Straight-pipe systems may have cesspools or non-functioning septic tanks, but they are assumed in this example to be failing standard treatment requirements.
[b] Septic tanks are assumed to be designed water-tight. Leaks are addressed in the engineering framework as a potential failure mode.
[c] Some ATUs have a built-in trash trap and do not recommend the use of a septic tank in advance of the ATU.

Potential stressors include any physical, chemical, or biological entity that can induce an adverse response in a receptor. The primary stressors for this framework are:

- Pathogens (for example, bacteria, protozoa, and viruses)
- Total and specific forms of nutrients (nitrogen and phosphorus)
- Unaesthetic features
- Noxious odors
- Oxygen demand

Both stress induced by ordinary function (nominal performance) and that induced by dysfunction (non-performance) are considered. Potential receptors include human and non-human organisms and systems (for example, ecosystems, communities, and social and economic systems).

Assessment endpoints are an explicit expression of the value that is to be protected through the use of one or more component frameworks. Endpoints typically consist of

1. An entity
2. A property of the entity that can be measured or estimated
3. A level of effect on the property that constitutes an unacceptable risk

An appropriate level of effect cannot be specified in this assessment framework, because users of the framework must define these with input from stakeholders and regulators. Assessment endpoints are based on their susceptibilities to the stressors of concern and relevance to public policy and management goals. A set of default assessment endpoints is discussed in this framework. Conceptual models are used to describe and to depict visually the expected relationships among the stressors, exposure pathways, and receptors (assessment endpoint entities) in the problem formulation. Only those relationships considered in the risk assessment are typically included in the model.

A generic conceptual model for the transport of wastewater and its constituents (for example, organic material, nutrients, and pathogens) has been developed, as well as specific conceptual models for each component framework. The source is assumed to be the year-round residence of a single family of four with an average daily loading rate of approximately 280 gallons. However, the framework is flexible enough to accommodate other assumptions (for example, seasonal occupation, multi-family homes, or restaurants) with some modification. The conceptual model includes assumptions about:

- Backup of the treatment system
- Surface breakthrough from structural failure
- Contamination of the land surface
- Transport into drinking water wells and groundwater
- Exposure of offsite people
- Exposure of aquatic biota

Engineering Component Framework

This component framework uses the risk analysis methodology called Failure Modes and Effects Analysis (FMEA) to address the issues specific to the design and performance of the OWT system of interest. As with all component frameworks, the assessment format includes problem formulation, analysis, and risk characterization. The problem formulation consists of planning the FMEA.

FMEA is an inductive analysis in which a detailed systematic component-by-component assessment is made of all possible failure modes and their resulting effects on a system. A failure mode is the manner in which the system or component has failed. For example, failure modes for mixing liquids would include no mixing, too vigorous mixing, insufficient mixing, or mixing of the wrong thing. Possible single modes of failure or malfunctions of each component in a system are identified and analyzed to determine the effects on surrounding components and the system. The causes of a failure mode are the

- Physical or chemical processes
- Design defects
- Quality defects
- Part misapplication
- Other methods that are the reasons for failure

FMEA is designed for evaluating system failures (dysfunction), but the methodology can also be used to evaluate OWT system performance under normal operating conditions, in which case a fraction of the assessment endpoint (and an associated event duration) is specified as a level of treatment that warrants attention.

Problem Formulation

The problem formulation for a FMEA entails the organization of as much information as possible about the system concept, design, and operational requirements. The FMEA may be performed with limited design information by answering the following questions:

- How can each part conceivably fail?
- What mechanisms might produce these modes of failure?
- What could the effects be if these failures did occur?
- Is the failure in the safe or unsafe direction?
- How is the failure detected?
- What inherent provisions are provided in the design to compensate for the failure?

The task of identifying subsystem failure modes can take either of two approaches:

1. Functional approach—Listing each subsystem, its functions, and the failure modes leading to the loss of each function
2. Hardware approach—Listing each part and its probable failure modes

The hardware approach is used most often when detailed part design information is available. With either approach, the potential failure modes are identified through answers to the following questions:

- In what way can this subsystem fail to perform its intended function?
- What can go wrong although the subsystem is manufactured/assembled to specifications?
- If the subsystem function was tested, how would its failure mode be recognized?
- How will the environment contribute to or cause a failure?
- In the application of the subsystem, how will it interact with other subsystems?

General types of failure modes for the functional approach include:

- Failure to operate at the prescribed time
- Failure to stop operating at the prescribed time
- Intermittent operation
- Wearing out of components
- Degraded output

General types of failure modes for the hardware approach for the traditional wastewater disposal system include:

- Plugged system
- Too little flow
- Too much flow
- No settling
- No anaerobic activity
- Leak or rupture
- Flooding

Causes of a failure mode can be divided into two categories: (1) design deficiency, or (2) process variation that can be described in terms of something that can be corrected or can be controlled.

Analysis

In the analysis stage of the assessment, the probability and magnitude of the failures are estimated. Occurrence (OCC) is the likelihood that a specific cause of failure will occur. Each cause of failure listed in the FMEA requires an estimate of its possible failure rates and/or its mean time between failure probabilities.

OCC can be based upon historical data, including the service history, warranty data, and maintenance experience with similar or surrogate parts. OCC probabilities can be based on the frequency of the initiating event (for example, seismic events or floods), the independent failure rate of components (for example, valves or piping), or historical experience/engineering judgment (for example, saturation of leach fields).

The second part of the analysis is the estimation of the severity of a failure. Severity (SEV) of the impact for each failure mode is assessed and classified according to rankings outlined in a severity table. Health and safety of the homeowners and offsite personnel are often the primary criteria in determining the SEV ratings for OWT systems, although effects on environmental entities and property are also considered.

Both the magnitude of effluent characteristics (that is, biological oxygen demand, total suspended solids, total nitrogen, phosphorus, and fecal coliforms), and duration of the failure event or routine performance level are considered. Duration is defined as the time elapsed between the start and end of the event, that is, between the point in time at which the OWT system component is no longer achieving the specified level of treatment and the point in time at which it is once again achieving the specified level of treatment.

The severity scales used in the FMEA for individual OWT systems vary by assessment endpoint. The severity scale is designed to yield a score of:

- Between 7–10 for events that warrant action
- Exactly 6 for events that may warrant attention
- Between 1–5 for events that are considered negligible

Example severity scales developed specifically for OWT issues are presented for each of the engineering assessment endpoints. The same severity scale can be used to estimate both the unmitigated and mitigated risks of a particular failure mode. Mitigation measures are assumed to reduce severity to a marginal level. Selection of appropriate severity scales is best accomplished with stakeholder input.

Risk Characterization

Two risk characterization procedures are recommended in the engineering component framework. The first risk characterization effort defines the “unmitigated” risks. If unacceptable risks are estimated in this step, then additional detection and process controls are considered and the risks are re-evaluated and categorized as “mitigated” risks. In the latter procedure, the user evaluates the ability of the proposed mitigative measures (that is, control processes or detection and correction attributes or mechanisms) to avoid a failure event or to detect a failure event and to correct the problem.

Public Health Component Framework

The public health risk assessment component is used to evaluate potential health risks from exposure to wastewater effluent or environmental media that have come in contact with wastewater effluent. The goal of this framework is to provide quantitative risk estimates for constituents of concern that originate in wastewater effluent of OWT systems.

Problem Formulation

The public health risk component focuses on primary constituents of concern (stressors) to humans in wastewater effluent: nitrogen-containing compounds (nitrate and nitrite) and microbial pathogens. The human health property evaluated as the result of exposure to chemicals originating in wastewater is systemic toxicity (noncarcinogenic effects). Chemicals of most concern with respect to adverse impact to public health in wastewater effluent are nitrogen-containing compounds—nitrate and nitrite. Cyanosis among infants who drink well water is a commonly encountered clinical manifestation of nitrate toxicity.

The two human health properties commonly evaluated as a result of exposure to microorganisms originating in wastewater are infection and illness. Because the risk of illness can vary greatly with the type and strain of microorganism, as well as host age and other host factors, the microbial assessment endpoint evaluated in this pubic health framework is risk of infection. Microbial pathogens of concern include viruses, bacteria, and protozoa. In this framework the indicator of bacterial pathogens is fecal coliforms, and the indicator of viruses is rotavirus.

Analysis

Assessment entities include potentially exposed populations and can be evaluated based on several subcategories such as age (for example, children, adults, geriatrics). In addition, sensitive subpopulations may be evaluated based on gender, ethnicity, baseline health status (immunocompromised, hereditary diseases, and other factors), or any other site-specific health characteristic of the potentially exposed population that warrants consideration. The level of effect guideline for chemicals in this public health risk assessment is defined as exceedence of the reference doses (RfDs) for systemic toxicity. The attribute evaluated for microbial exposures is risk of infection. Likewise, the microbial risk of infection guideline is defined as greater than 1×10^{-4}.

The conceptual site model should describe:

- Location of the OWT system
- Location of residences and wells
- Topography
- Groundwater
- Surface waters

- Soils
- Potentially exposed populations

Exposure pathways and points occur onsite (within the residential lot) and offsite. The generic conceptual model provided the basis for the development of the conceptual model for the public health frameworks for pathogens and nitrates.

The spatial extent of the risk assessment is defined by the boundaries of the site on which the OWT system is located and the migration pathways to potentially exposed persons (for example, offsite publics or tourists exposed at the site boundary). Likewise, the assessor should specify the temporal scale of analysis based on the lifetime of the OWT system or components of interest, the travel time of wastewater constituents of concern to a potential exposure point, or local or state regulations.

Current exposure concentrations for some constituents of concern can be directly measured, such as the concentration of nitrate in soil, surface water, or groundwater. Likewise, current concentrations of some microbial pathogens can be measured in soil, surface water, or groundwater. Fate and transport models are often utilized to estimate exposure concentrations of

- Viruses at all times (which are difficult to measure)
- Chemicals at past or future points in time
- Bacteria and protozoa at past and future points in time

For adverse impact to human health to occur, a potentially exposed population is required at the exposure point. Potential receptors include residents and visitors onsite and residents and tourists at the site boundary. Routes of exposure for potential receptors include ingestion, dermal contact, and inhalation. For each exposure route, an exposure model is constructed and applied to estimate a daily intake for the wastewater constituent of concern. Daily intakes can be estimated for acute and/or chronic exposures of individuals with differences in body weight, ingestion rates, and exposure frequencies. The estimated doses are compared to health toxicity values or guideline values to determine if adverse health impacts are predicted and the level of the effect.

Risk Characterization

The risk characterization step of the risk assessment process evaluates the exposure and effects data developed in the analysis step while producing estimates of risk as well as explanations of results and uncertainties associated with the risk estimates. If the calculated chemical intake exceeds the chemical-specific reference dose (RfD), adverse health effects maybe expected (that is, the hazard quotient is greater than 1.0). The quantification of microbial risk from exposure to wastewater effluent or to environmental media that have contacted wastewater effluent is more challenging because of the

- Numerous routes of exposure,
- Differing numbers of organisms in various media,
- Differing amounts of media consumed per individual

- Potential propagation of infectious agents
- Potential latency period

Dose/response data are available for several pathogenic microbial species associated with wastewater, including several bacteria, one or two protozoans, and select viruses. To assess risks of infection from microbial exposures, the measured fecal coliforms in environmental samples are assumed to be *E. coli*. While most *E. coli* are not pathogenic, the presence of *E. coli* suggests the potential presence of pathogenic strains.

To estimate the risk of microbial infection from ingestion and/or contact with soil, groundwater or surface water, a beta-Poisson dose-response model is used. The beta-Poisson model is also used to estimate risk of rotavirus infection. Measured rotavirus concentrations are used to determine doses of microorganisms from the exposure models. Exposure to microbes that results in a risk of infection greater than 1×10^{-4} exceeds the example guideline for this public health risk framework and indicates that risk management preventative measures may be warranted.

Conservative assumptions are intended to provide a margin of safety due to the uncertainty in the estimates of risk to the public. Because precise information is not known about all exposure parameters such as the amount of groundwater ingested, exposure durations, or the amount of time recreationers spend in the water while swimming, best estimates and conservative assumptions are made during the risk assessment process.

If time and resources permit, a quantitative uncertainty analysis of the parameters and models used to estimate risk may provide a better understanding of technical issues associated with the risk estimates. Quantitative methods for uncertainty analysis are more accessible for the public health component framework than for the broader socioeconomic or ecological frameworks.

Ecological Component Framework

The ecological component framework is used to evaluate the potential adverse impacts on non-human biota and ecosystems. The three-stage risk assessment format used throughout the integrated framework (that is, problem formulation, analysis, and risk characterization) is also used in this component framework.

Problem Formulation

As stated in the general problem formulation, this framework is designed to addresses micro-scale OWT systems, that is, an individual residential lot with an OWT system. Although some ecological impacts are evident only at the macro-scale (for example, population-level impacts on wide-ranging, mobile species), others can potentially manifest at smaller geographic scales (for example, impacts on individual plants and sensitive receptors). This situation is particularly true for sites where the dilution of OWT effluent at the exposure point is low.

The micro-scale is pertinent to residential treatment systems located adjacent to small ponds, streams, or lagoons and some parts of shallow estuaries (for example, coves where tidal water exchange is very limited). With respect to amphibians, the micro-scale is also relevant for sites with small or temporary ditches onsite or at the site boundary. The spatial bounds for the ecological assessment may extend somewhat beyond the site boundary, but it is still a micro-level assessment because only one OWT system is being evaluated.

Although the exposure models and exposure-response models presented here could also be used for macro-level assessments, the localized scale of analysis is the primary justification for the selection of stressors and assessment endpoints emphasized in this component framework. Most potential effects at the local scale are direct effects (for example, increased plant biomass), rather than secondary effects (for example, effects from losses of forage or habitat or from increases in predation).

Analysis

The temporal bounds of analysis may be based on several factors, including:

- The lifetime of a treatment system
- The lifespan or sensitive life-stage of a particular receptor
- Periods of high release (for example, storm events)
- Regulatory requirements
- A decision by a risk manager (decision maker)

The nutrients nitrogen and phosphorus are the principal ecological stressors associated with residential OWT systems. However, the user must identify the stressors that are the focus of each particular risk assessment. Nutrient inputs to a surface water body have the greatest impact if background concentrations limit production or growth rates (for example, primary production) of one or more assessment endpoint entities. In general, nitrogen is a limiting nutrient in estuarine waters in temperate environments and phosphorus is a limiting nutrient in most fresh waters in temperate environments.

Two types of surface water ecosystems are distinguished based on differences in prevailing nutrient dynamics:

1. Freshwater systems (for example, ponds)
2. Estuarine systems (for example, coastal lagoons).

In the characterization of exposure, differences between lotic (flowing) and lentic (still) waters also are noted.

A generic conceptual model for the potential effects of an OWT system on a freshwater receiving environment is generated. Phosphorus exposure is the major determinant of phytoplankton production in most North American lakes. This nutrient may also be limiting in streams, but high water flows and flood events may overwhelm the effects of nutrients.

Various forms of nitrogen can be directly toxic to aquatic biota, especially reduced forms such as ammonia, though the primary exposures of aquatic organisms and amphibians to nitrogen from an OWT system following release and oxidation in soil are exposures to nitrate. Organic matter and reduced nitrogen forms, such as organic and ammonium, that are associated with wastewater and directly released to surface water bodies are additional stressors that can cause oxygen limitation.

A generic conceptual model for wastewater treatment unit effects in a shallow estuary or lagoon is generated and discussed. Nitrogen is the primary stressor, which can be directly toxic or can interact with biota to produce secondary stressors (limited light penetration, oxygen limitation, reduction in habitat, or reduction in forage vegetation or prey). Organic matter and reduced nitrogen forms, such as organic and ammonium, that are associated with wastewater and directly released to surface water bodies are additional stressors that can cause oxygen limitation. Algal production, macrophyte production, fish community abundance and production, benthic community abundance and production, and amphibian community abundance and production are examples of options for risk assessment endpoint properties for OWT systems.

The characterization of exposure is the phase of an ecological risk assessment in which the spatial and temporal distributions of the intensity of the contact of endpoint entities with stressors (for example, nutrients) are estimated. Exposure must be characterized in terms that are useful for estimating effects. That is, if the average annual input of phosphorus is known, it may need to be converted to the average annual concentration of phosphorus in the water body if the exposure-response relationship is based on this latter unit.

The exposure of ecological receptors to nutrients is characterized by measurement or modeling. Measurements of most forms of nutrients and dissolved oxygen are easy, and if sufficient measurements are taken to characterize spatial and temporal variability, measurement is clearly more accurate than modeling for a risk assessment of current nutrient releases. However, measurement cannot distinguish the incremental exposure associated with wastewater treatment releases from other sources of nutrients. Prospective risk assessments require modeling of concentrations of nutrients in surface water at the exposure point. Retrospective risk assessments require modeling if historical measurements are not available. Most exposure-response models for ecological receptors in surface water require concentrations of nutrients as measures of exposure. Ecotoxicologists tend to measure or to model nutrient concentrations rather than loading rates, although loading rates may be the starting point.

OWT system effluent that migrates to surface water through the soil probably will not retain enough organic carbon to create substantial carbonaceous biochemical oxygen demand (CBOD) in receiving waters, based on data from sand filters and the behavior of dissolved organic carbon in sand aquifers. However, untreated wastewater that reaches the soil surface and either flows to the receiving water or is transported in surface run-off could produce locally high CBOD and, therefore, localized areas of hypoxia.

Exposure-response relationships may be available or derived from field observations, laboratory or mesocosm tests with site-specific media, or relationships from published studies. These latter relationships may focus on exposure measures, ecological receptors, and locations that are somewhat different from those of concern in a particular assessment, but they may be the only relationships available for retrospective or prospective assessments for which field observations or surface water samples are not available.

Exposure-response models may be empirical models derived from measurements at one or more sites (for example, biological surveys), mechanistic models derived from first principles, or thresholds determined from the literature or from site-specific tests. In this risk assessment framework, only one mechanistic model was identified to characterize ecological effects. When field observations are used, it may not be possible to attribute causation, if multiple stressors are present or if multiple sources of one stressor are present.

Risk Characterization

In the risk characterization, the information in the characterization of exposure and the characterization of effects is combined to estimate risks. If sufficient data are available, the risk characterization should consist of a comparison of distributions of exposure concentrations and those of probable effects for each assessment endpoint. The evidence is often presented in a weight-of-evidence table, with qualitative or quantitative uncertainty. For each line of evidence, several factors are considered, including:

- Data quality
- Relationship of measures of effect to the assessment endpoint
- Relevance of measures of exposure at the site to those that feed into the exposure-response relationship

For example, the weight of evidence could utilize effects models based on total nitrogen loading, models based on nitrate concentration, and biological survey results. Lines of evidence may be weighted differentially, if the assessor has more confidence in one than in another. If the goal of the ecological risk assessment is to estimate the magnitude of effect, the estimates of magnitude that result from using different methods to characterize exposure or effects may be weighted, and the result may be a weighted average estimate of the magnitude of effects. In any ecological risk assessment, sources of variability and uncertainty in results must be described, and wherever possible, quantified.

Socioeconomic Component Framework

This framework is developed for use in evaluating potential socioeconomic impacts and risks from

- Exposure to wastewater effluent or environmental media that have come in contact with wastewater effluent
- Efforts to manage those effluents with an OWT system

The socioeconomic component addresses many issues that are typically part of the risk management process, such as monetary costs of the design and installation or replacement of the OWT system, maintenance costs, and opportunity costs.

Only the impact and risk assessment at the micro-scale (that is, a single residential OWT system) is addressed in this document. However, macro-level issues are relevant in a micro-level assessment to the extent that they create or influence stressors and other aspects of risk assessment that must be addressed in an assessment of an individual OWT system. For example, the presence, capabilities, services, and costs of a maintenance contractor or a responsible management entity are macro-level factors that affect the likelihood of system failures and determine the fees that are paid out by the individual system for maintenance or oversight.

Problem Formulation

Many of the impacts and risks that wastewater treatment systems pose to the socioeconomic environment grow out of concerns related to the engineering, public health, and ecological dimensions of the system being studied. People value their money, their health, and their ecosystems, meaning that impacts or risks to any of these phenomena result directly or indirectly to impacts or risks to the socioeconomic environment. The impacts or risks of a wastewater treatment system are understood to occur in the context of an existing socioeconomic environment. Thus, the assessor needs to understand and be able to characterize that environment in order to be able to assess the impacts or risks associated with the given wastewater treatment system, compared with the pre-existing condition of that environment.

For the assessment of impacts and risks of wastewater treatment systems at the micro-scale, the existing socioeconomic environment would include but not be limited to the following characteristics:

- Economic status of each receptor and the receptor's neighbors (receptors are people, groups, or social or political constructs that are potentially exposed to one or more stressors)
- Presence or absence of vulnerable populations among each receptor and the receptor's neighbors (that is, vulnerable in terms of susceptibility to health or economic stresses)
- Development status of the receptor's property (for example, as a permanent, temporary, or seasonal residential property) and neighboring properties, including the current value of the properties and the aesthetic qualities of existing land uses

- Existing wastewater treatment capacity/capability of the receptor's and neighbors' environment
- Existing wastewater treatment capacity/capability of the source OWT system, including hydraulic capacity and capability of accepting household chemicals, high CBOD loads, and other components
- Presence, capabilities, services, and costs of a maintenance contractor or a responsible management entity (will affect the likelihood of system failures and will determine what, if any, fees are paid out by the individual system for maintenance and/or oversight)
- Capabilities and willingness of receptors (including those of absentee property owners) to maintain existing or new wastewater treatment system
- Sensitivity of receptors to intervention(s) taken by outside agents (for example, to inspect or maintain an onsite wastewater treatment system or otherwise take action on the property)
- Temporal and climatic variability of the receptor's environment (for example, seasonal, diurnal, or meteorological variations)
- Potential for catastrophic natural events (such as, flood, earthquake, landslide, or hurricane)

Stressors are any physical, chemical, or biological entity that can induce an adverse response in a receptor, either directly or indirectly. In many cases the effect can be positive or beneficial as well as adverse; therefore benefits are included in this framework. Benefits can include increases in property value, increases in development potential, and improved health status (and reductions in health care costs associated with that improved health status). In the context of the socioeconomic impact and risk assessment for wastewater treatment systems, stressors are more likely to indirectly affect the socioeconomic environment via the physical environment.

Socioeconomic stressors can be both tangible (such as real monetary costs or changes in property value) and intangible (such as "psychic costs" of allowing others on one's property for periodic system inspection). For many of the stressors (and attributes), the interrelationships among concepts and variables are complicated. For instance, one can argue that property value is simply a metric for a bundle of characteristics, some of which are physical and tangible (such as size of lot, size of house, number of bedrooms, *and* the kind of wastewater treatment system) but others that are more perceptual (such as perception of sanitation, healthiness of a home, aesthetics of landscaping, and sense of well being).

The assessor needs to identify those aspects of the wastewater treatment system that could adversely or beneficially affect the socioeconomic environment. Moreover, the assessor must identify those aspects of the system that could affect the environment if the system operates or works successfully and if the system fails (whether due to a design flaw, improper maintenance, or capacity overload due to climatic or behavioral changes).

Time and monetary costs are the principal socioeconomic stressors associated with the micro-scale of the OWT system. The time and monetary costs borne by the different receptors and the distribution of those costs by the receptors are the principal measurements that will need to be characterized in the assessment. Additional, intangible stressors can often be addressed by measuring related time and monetary costs as surrogates.

Receptors can be selected as assessment endpoint entities. At the micro-scale of this assessment framework, the receptors include:

- Individuals (property owners, occupants—permanent, temporary, or seasonally transient)
- Vulnerable subgroups or populations
- Adjacent populations (including any vulnerable populations)

The resources of those individuals or groups of individuals, and the relevant characteristics of those receptors (such as socioeconomic status, happiness, wealth, and health) are the important attributes for which endpoints are needed in the assessment.

Levels of effect may be specified for assessment endpoints that constitute adverse effects (but not usually for beneficial effects). The user of this framework should be aware that changes in values for the endpoints are susceptible to variable interpretation by different interested parties or stakeholders and, as pointed out previously, that both the “real” and “perceived” changes in value for some if not all of those endpoints may be of interest to the decision maker.

In contrast to the engineering, public health, and ecological dimensions of the assessment framework, where acceptability endpoints are better understood and more generally agreed to, assessment endpoints for the socioeconomic dimension of the problem are sometimes more difficult to identify and often more difficult to quantify (in general because they are less amenable to common understanding or agreement). This dilemma is particularly problematic when addressing the “intangible” values that are important to the assessment (such as the psychic costs or stigma of alternative wastewater treatment systems).

Analysis

In contrast with the other components of this framework, the characterization of exposure in the socioeconomic impact and risk assessment is not based on discipline-specific modeling or estimation but rather derives from either the presence (or absence) of the OWT system of interest (and results in the effects on the endpoints) or from the findings of the exposure assessment of the other risk assessment domains. In the latter instance, the socioeconomic risk assessor would characterize socioeconomic exposure in terms of the spatial and temporal distributions of the intensity of the engineering, human health, and ecological exposures and effects.

The characterization of effects is the determination of the nature of adverse and beneficial effects of the stressors on the receptors and the receptor’s (entity’s) properties and attributes. These effects may be available or derived from field observations and research, secondary data sources (for example, census data), or from published studies.

Risk Characterization

In the impact and risk characterization portion of the assessment, the information in the characterization of effects is used to estimate impacts and risks. The engineering subcomponent should provide information regarding the design, installation and maintenance and repair costs of the OWT system of interest. The assessor can stipulate, by assumption, the cost of money or assume variable costs of money (for opportunity cost). Likewise, the engineering subcomponent should supply information regarding the time cost of the OWT system (in terms of hours per month to maintain the system) and whether that time is a cost to the property owner or to a firm contracted for OWT system maintenance. Also, for socioeconomic risk assessment of operational failure, the engineering subcomponent would provide estimates of rates of failure, which then translate to frequency and severity of socioeconomic stressors such as cost to repair a system, aesthetic (for example, olfactory) “insult” from failure, loss of property value if repair is not possible, and other stressors.

In many socioeconomic impact and risk assessments, only one line of evidence may be available for the impact and risk characterization of a socioeconomic endpoint property and stressor. However, if multiple measures are available, all of these may be used to obtain distinct estimates of impacts and risks to the assessment endpoints. The evidence may be presented in a weight-of-evidence table, with consideration of qualitative or quantitative uncertainty. For each line of evidence, several factors are considered, including data quality and the relationship of measures of effect to the assessment endpoint.

General Risk Characterization

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

- Summarizing the risks and uncertainties characterized in each component assessment
- Characterizing the integrated risks and uncertainties for assessment endpoints that are potentially affected by one or more other assessment endpoints
- Summarizing the integrated risks and uncertainties characterized in this section

This approach highlights the fact that risks can be divided into two general categories for the purposes of this framework: independent risks and conditional risks. The estimation of independent risks is carried out without reference to the estimation of all other risks to a particular assessment endpoint. For example, the risk of treatment failure (dysfunction) due to seasonal flooding of the WSAS can be estimated in the engineering risk assessment even if the other three component assessments are not performed. Conditional risks are those for which the estimation of risk is conditional on the estimates for one or more other risks. The risk of infection by a virus is, in part, conditional on the risk of having a treatment failure due to seasonal flooding of the WSAS, which then results in exposure of the residents to viruses.

Independent Risks

Independent risk results need to be presented in adequate detail for decision making, including at a minimum, a rating for both the estimated risks and the uncertainties associated with those estimates. Some component assessments include a rating system in their design, whereas others may just report the results without explicitly classifying the risks as being more or less than the assessment endpoint level. The former method is preferred over the latter. For example, the engineering framework in this assessment uses a Roman numeral ranking system ranging from I to IV and the ecological framework uses a weight-of-evidence process to assign a plus/minus rating for each assessment endpoint. These ratings can be used without modification for the general risk characterization. However, a rating should be assigned in the general risk characterization section if the component assessment does not do so directly. For example, the risk of infection from the public health component framework is reported as an estimated rate of infection for example, 1:10,000) rather than as a rating.

A simple acceptable/unacceptable rating system is a useful and intuitive tool for this purpose. The decision rules for that system are:

- **U**—Indicates an unacceptable exceedance of the selected level of effect for the assessment endpoint
- **A**—Indicates an acceptable rating
- **I**—Indicates that insufficient evidence was available to conclude whether the selected level of effect for the assessment endpoint was exceeded (that is, the acceptability of the risk is indeterminate)

To support the risk management process, the general risk characterization needs to summarize the previously detailed uncertainties in a simple and consistent manner.

A simple and effective rating method also entails classifying the level of confidence associated with a risk rating as low, moderate, or high. These clearly defined rankings must be applied consistently across all assessment endpoints. Combining the risk and confidence ratings in one table for each assessment endpoint is a useful practice for risk communication purposes.

Conditional Risks

Conditional risks are best evaluated in the general risk characterization. Therefore, the user must provide additional details in this section regarding the characterization of risks, which are dependent on the estimates for one or more other risks.

Two potential methods for integrating the risks from multiple component assessments were discussed in Jones *et al.* (2001): mathematically propagating risks estimated in each component and logically weighing the evidence of risks presented in each component.

Mathematical propagation is only possible when quantitative estimates of risk are calculated for all of the assessment endpoints included in the conditional risk calculation. However, only the public health framework is designed to result in probability estimates (for example, a 1:10,000 risk of infection) as a component of the current integrated framework. Therefore mathematical propagation of risks cannot be used in this framework.

The characterization of conditional risks in this integrated framework is based on a variation of the weight-of-evidence process. In this general risk characterization section, the component assessment for each assessment endpoint is treated as a line of evidence for the conditional risks associated with two or more assessment endpoints. The user must logically evaluate the likely interactions between each assessment endpoint to see how these interactions support or refute the hypothesis that an OWT system poses a risk to a particular assessment endpoint. This evaluation is accomplished by weighing the evidence for conditional risks. Evidence that supports or refutes the hypothesis that interactions among two assessment endpoints lead to increased risks to one of those assessment endpoints should be discussed in detail.

A rating should be assigned in the general risk characterization section for each of the conditional risks being evaluated. This rating system should be compatible with the rating systems used to summarize the independent risks. A variation of the acceptable/unacceptable rating system discussed above is used to rate the conditional risks.

Uncertainties associated with estimating conditional risks should be described. Sufficient detail should be provided to help decision makers understand the origin, magnitude, and tractability of these uncertainties. Tractability refers to the level of effort that would be required to substantially reduce these uncertainties (that is, increase confidence). The ratings of low, moderate, and high confidence are recommended for the characterization of conditional risks. The risk and confidence ratings for conditional risks can be presented in a summary table similar to that used for independent risks.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1 INTRODUCTION

Risk is implicitly included in current permitting regulations for onsite 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 onsite systems in typical soil conditions. These 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 used typically when the prescriptive permitting guidelines for standard septic systems are violated. Given that baseline risks have not been estimated, it is difficult to establish risk-based siting rules for alterative treatment technologies.

Performance-based permits are often issued when prescriptive guidelines do not apply. Performance-based permits require potentially expensive monitoring and maintenance and may be viewed as a risk by homeowners and regulators alike. A primary concern for homeowners is the potential costs of repair if their system fails the tests. A primary concern for regulators is the potential impacts to the public and environment if the system fails. The likelihood and magnitude of risks of these two types of failures are generally not well known. Therefore, it is difficult to rectify the perceived risks with the actual risks.

To enhance or improve performance-based permitting for alternative treatment systems will require standardized methods for explicit risk estimation under a variety of conditions. With such methods a set of siting guidelines that account for the site-specific variables (such as depth to groundwater, soil type, and temperature) that drive risks to a suite of receptors could be developed and tested. (See Appendix A, *Glossary* for definitions of key technical terms.)

Efforts are underway within the field of onsite wastewater treatment to develop risk-based approaches to decision making. These efforts include modeling 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. Coincident limitations generally include a lack of explicit, risk-based endpoints (that is, 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 (engineering, ecological, public health, and socioeconomic).

Background

This integrated risk assessment framework for individual onsite wastewater systems is the second phase of an effort to develop a comprehensive approach to risk-based decision making for onsite wastewater treatment (OWT) systems. The first phase began with a series of regional forums and culminated in a national research needs workshop. The objective of that effort was to identify critical research gaps in the field of decentralized wastewater treatment. Regional forums were held in the southeast (Tampa, Florida), northeast (Kingston, Rhode Island), and northwest (Seattle, Washington). Five white papers were developed for the national workshop based on the results of the regional forums. The selected topics were:

- Integrated risk assessment/risk management
- Onsite wastewater soil absorption systems (WSAS)
- Pathogen fate and transport
- Nutrient contamination
- Economic costs and benefits

These white papers are available in the *National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment* (EPRI 2001) and were used as the basis for the current project. The white paper on integrated risk assessment/risk management included a high-level risk assessment framework for OWT systems (Jones *et al.* 2001). The high-level framework provided a blueprint for integrating four different discipline-specific assessments (that is, engineering, public health, ecological, and socioeconomic assessments). The current framework expands on that blueprint by providing more detailed guidance on applying risk assessment principles to OWT systems while narrowing the spatial and temporal scope of the framework to that of an individual OWT system.

Guidance will still be needed for larger spatial scales (such as watersheds). This shift from individual systems to larger geographic scales can best be accomplished by rolling the current framework for individual sites into a framework that can be used to address the additional issues associated with watershed-level assessments. Therefore, the current framework was constructed in a manner consistent with the anticipated structure and function of higher-level frameworks. For example, some ecological and socioeconomic issues are primarily relevant at larger geographic scales (such as effects on populations of mobile, non-human organisms; accrual of costs and benefits to certain human populations and not others; and equity and fairness of charges for system installation or system management).

Where appropriate, key macro-level issues are identified and briefly discussed with respect to their impact on assessments at the micro-level.

Objectives

The primary objective of this project was to develop an approach to risk-based decision making for individual onsite wastewater treatment systems. Additional objectives of this project are to provide an approach that can be used to:

- Integrate multiple types of risks (that is, engineering, ecological, public health, and socioeconomic)
- Test and parameterize OWT study sites and demonstration projects (sites with adequate data)
- Facilitate a streamlined approach for eventual application in the field by the typical wastewater professional (such as county health department officials) based on the results of detailed testing
- Support the eventual development of explicitly risk-based permitting guidelines for alternative decentralized treatment systems
- Guide research and development efforts toward solutions to the problems that drive risks and uncertainty in those risks

The risk assessment framework is designed to be most useful to professionals who are trained in the principles of risk assessment. These may include biologists, engineers, or social scientists at regulatory agencies or consulting companies, the latter of which may be hired by regulatory agencies, developers or industries that design OWT systems. The users who would make the most complete use of this framework would be multidisciplinary teams of risk-assessment and OWT design experts. Users of this framework for individual OWT systems may also include technically trained stakeholders, such as county planning commissions, with an interest in educating themselves about the factors contributing to biological, engineering, or social risk from individual onsite wastewater systems and methods for their assessment. Entrepreneurs might create user-friendly mathematical models that take advantage of the links among risks in the onsite wastewater system context.

The methods discussed include measurements and models for retrospective and prospective risk assessments. If the goal is to conduct assessments of proposed new individual wastewater systems (such as permitting), then the user can focus primarily on the discussions of modeling, with the understanding that measurements of chemicals and pathogens might be included in permit requirements. If the goal is to conduct a risk assessment for existing OWT systems (such as determining the cause of observed toxicity in a lake) then measurement of nutrient or pathogen concentrations may be more important than modeling. If the user is an OWT system designer or the agent of a designer, the user must select generic environments in which the system will be used prior to conducting risk assessments. As an engineer, an OWT system designer may choose to focus on the Failure Modes and Effects Analysis of the engineering component framework, but he or she would need some knowledge of the three other types of risk in order to estimate severity of failure.

This framework was not developed to support macro-scale (for example, watershed-scale) risk assessments for multiple OWT systems. An addendum to this framework is needed for use in addressing the cumulative and emergent effects of multiple OWT systems and offsite treatment systems. Also, the framework was not developed to address benefits of different treatment systems; only the socioeconomic component framework addresses benefits directly. The framework does not support fully probabilistic analyses; only the human health risk assessment component framework (and to a lesser extent, the ecological risk assessment framework) expresses risk in terms of probabilities. Similarly, the framework was not developed to support comparative assessments of alternative wastewater systems. However, it could be used in that context if conservative estimates of exposure and effects, which could bias the analysis toward particular alternatives, are not used. Methods for balancing different types of risks, such as low human health and ecological risk for one alternative system, and high socioeconomic risk for another system, must be developed and applied in the risk management process.

Framework Organization

This framework for individual OWT systems follows the format of the high-level framework (Jones *et al.* 2001), which was largely based on the principles and practices of ecological risk assessment (US EPA 1998b and Suter 1993). The general risk assessment process consists of three basic steps:

- **Problem formulation**—A process for generating and evaluating preliminary theories about what effects might occur. Problem formulation 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.
- **Analysis**—Typically includes analysis of occurrence or exposure and analysis of effects. Analysis of occurrence or 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**—The process of combining the estimates of occurrence or 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.

These steps occur in roughly the order presented, but all steps are interrelated and the overall process is iterative. Thus, all aspects of the problem formulation might not be determined prior to the beginning of the analysis and risk characterization.

This framework for individual OWT systems is designed to integrate four different types of risk analyses:

- Engineering
- Public health
- Ecological
- Socioeconomic

Integration is accomplished by embedding a component framework for each of these four types of analyses in an overall (integrated) framework. Thus, the analysis step of the integrated risk assessment framework consists of four separate, but coordinated, component frameworks. Coordination and integration of the component frameworks is accomplished via a general problem formulation step and a general risk characterization step, as shown in Figure 1-1.

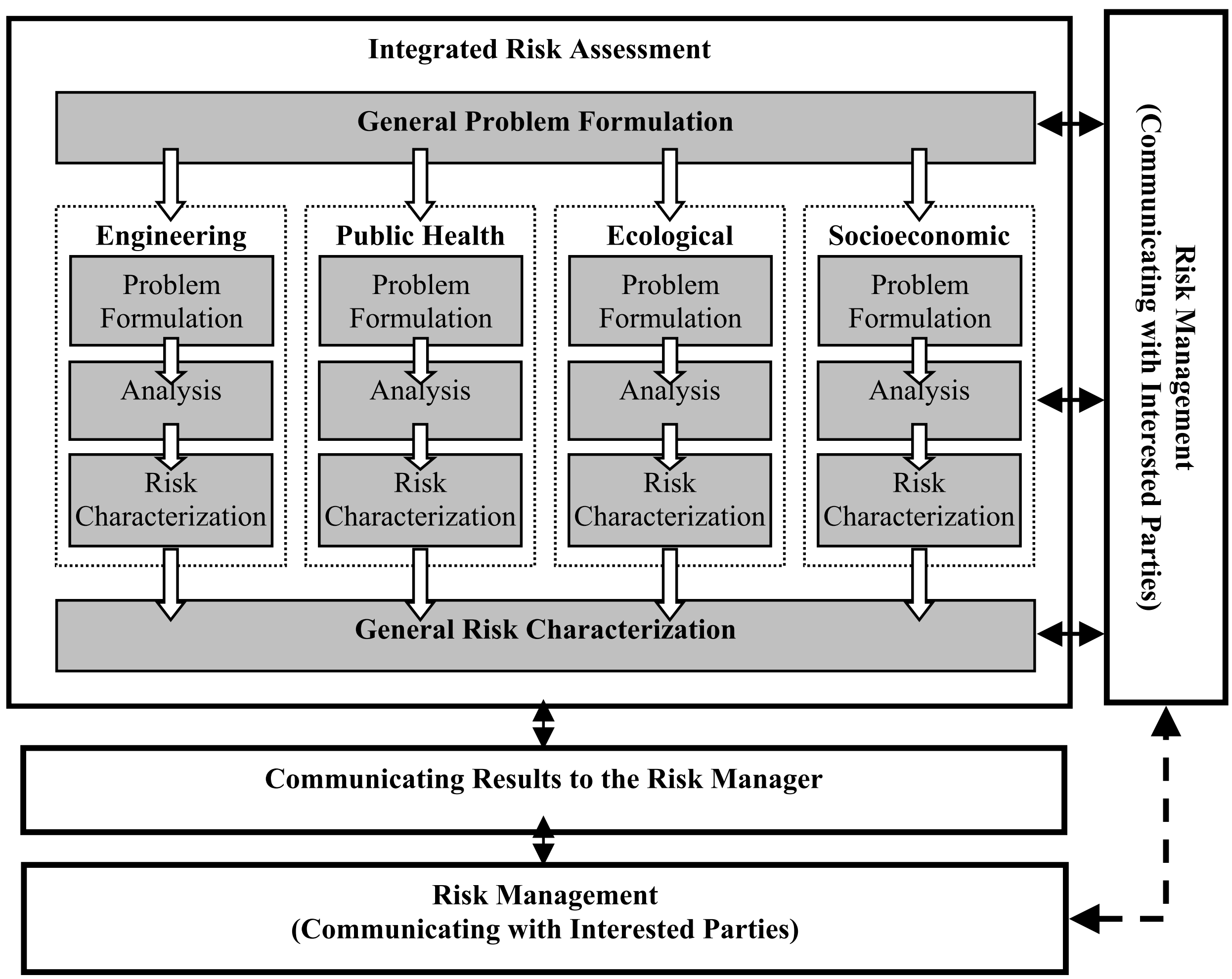

Figure 1-1
Integrated Risk Assessment Framework for Onsite Wastewater Treatment Systems.

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. Risk management is the stage at which the estimated risks for each assessment endpoint are weighed (subjectively compared) against the estimated risks to all other assessment endpoints and against other factors that were not specifically evaluated in the risk assessment (such as ethical and political considerations). The primary objective of risk management is to balance the risks and benefits to all assessment endpoints and parties of concern.

The sections of this integrated framework for individual OWT systems are organized following the integrated risk assessment and risk management diagram shown in Figure 1-1. The general problem formulation is in Chapter 2, *General Problem Formulation*. The engineering, public health, ecological, and socioeconomic component frameworks are presented in Chapters 3, 4, 5, and 6, respectively. The general risk characterization is presented in Chapter 7. Risk management issues are discussed briefly in the Executive Summary (a detailed discussion of principles and practices of risk management is beyond the scope of this risk assessment framework). Examples are used throughout the framework to illustrate the application of this risk assessment methodology to individual OWT systems.

2 GENERAL PROBLEM FORMULATION

General problem formulation is used to define the scope and objectives of the integrated risk assessment for an individual onsite wastewater treatment (OWT) system, and entails:

- Describing the spatial and temporal bounds of the assessment
- Defining the OWT system to be evaluated
- Identifying and describing the source and the potential stressors and receptors
- Selecting assessment endpoints and ensuring that they can be addressed within the appropriate component assessments
- Developing a conceptual model for the system to be evaluated
- Selecting appropriate measures of effects and exposure

These steps occur in approximately the order presented, though the problem formulation process is iterative. Each of these steps is discussed in the following sections.

Assessment and Management Goals

Prior to conducting the risk assessment, the user should specify the goals for the assessment. Assessments for individual OWT systems may be conducted for many purposes, but four primary purposes worth noting are:

- Planning for a new installation on a previously undeveloped site
- Evaluating the potential or observed effects of an existing OWT system
- Evaluating potential retrofits for a currently failing OWT system
- Setting regulatory policy and design parameters

The purposes and goals of the assessment should be well defined during meetings between the user (that is, the risk assessor) and the decision makers (that is, the risk managers) and clearly stated in the problem formulation. The subsequent components of this problem formulation section address key considerations for the selected goals. The content of this framework is generally broad enough to support the primary purposes mentioned above, as well as other less common purposes (such as a generic assessment for one or more specific OWT system configurations for purposes of aggregating the results at large geographic scales).

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 process, in turn, determines which assessment endpoints should be included and what should be addressed in each of the discipline-specific assessments.

Spatial Scale

The high-level framework (Jones *et al.* 2001) considered two spatial scales: the micro-scale and the macro-scale. The micro-scale referred to an individual residential lot with an onsite drinking water well and an onsite wastewater treatment system. The macro-scale referred to a watershed that contains many individual decentralized systems, as well as other point and nonpoint sources of pollution. The current framework addresses the micro-scale assessment of OWT systems. Macro-scale issues are considered in this framework for purposes of context (that is, with respect to the ways in which an individual OWT system is expected to influence the next larger scale of geographical organization and with respect to the ways in which macro-level issues drive assessments for individual OWT systems).

The spatial bounds of this framework are those of a typical residential lot. Lot size is generally a function of zoning regulations and standard setbacks, which may vary among geographic areas due to environmental conditions (such as climate and geology) and political factors (such as developmental pressures). Hence, lot size is a variable to be defined by the user of this framework.

Location of the site relative to potential receptors and receiving environments is an important aspect of the spatial bounds of the assessment. The distance from the wastewater discharge point (contact with groundwater under a wastewater soil absorption system (WSAS) or a near-surface release point) to the exposure point (such as a drinking water well or body of water) should be defined by the user. This distance is typically a function of zoning and setback requirements (minimum lot size). For framework development purposes, a drinking water well (onsite or on an adjoining site) and a body of water (such as a stream, pond, or estuary) are assumed to be located at the edge of the lot and down gradient from the wastewater treatment system. A set of default exposure points for the human and ecological receptors has been selected for use in the examples in this framework (see the Spatio-Temporal Variables section).

Temporal Scale

The user must also specify the temporal bounds of the assessment. For OWT systems the temporal scale should be based on the time elapsed since the system or component was installed.

Risk assessments can be divided into two general categories

- Retrospective
- Prospective

Retrospective Assessments

Retrospective assessments are used to evaluate the potential causes of the observed (current) conditions, and to estimate the likelihood that observed adverse impacts are due to the dysfunction of an existing OWT system. Retrospective assessments typically have measured data for the site being assessed. For example, the user might measure stressor levels at the exposure point (such as nitrate concentrations in a drinking water well) and then determine the likelihood and magnitude of effects based on these measurements. The temporal bounds of the retrospective assessment are determined by the age of the system or component at the time of the assessment, which should be specified in the assessment.

Prospective Assessments

Prospective assessments are used to estimate potential (future) risks (conditions). The user must estimate the probability and magnitude of potential adverse impacts on the OWT system or receptors. These potential adverse impacts, in some cases, may increase in magnitude and probability with increasing age of the component or system. Therefore, the user should clearly define the service-time over which the component or OWT system is being assessed.

Prospective assessments can be further divided into the following subcategories based on the initial conditions of the site being assessed, including:

- Sites that have never been used for wastewater treatment
- Sites that currently have a properly functioning OWT system
- Sites that currently have a failing OWT system for which a retrofit system (or components) are being installed

The three subcategories of prospective assessments differ primarily based on the current conditions and the types of data that are available. Prospective assessments for planned OWT systems at new sites (previously unused for wastewater treatment) typically have little, if any, site-specific data. For example, general geohydrological conditions may be known, but movement of wastewater constituents through the subsurface has not been measured at the site. Thus, all data used in the assessment are estimated.

Prospective assessments for existing OWT systems that are properly functioning may have measured performance data for the site and the OWT system being evaluated. However, the user must still extrapolate from the current conditions to estimate the potential future risks. For example, the user might measure the performance and condition of the system after five years of operation and then use these measurements to estimate the risks this system will pose to the receptors after 10 more years of operation. In this example, the temporal bounds of the assessment would be five and 15 years of operation.

Prospective assessments for existing OWT systems that are currently failing and being retrofit with a new system or components are likely to have substantial measured data for the site and OWT system being evaluated (the data collected to determine the type and magnitude of the system failure). For a prospective assessment the user is not trying to estimate the risks at the time of the assessment, because effects of the failing system have already been determined (an inspection or retrospective assessment has already been performed to justify the expenditure for a replacement system). In a prospective assessment the user needs to estimate the likelihood and magnitude of adverse impacts at a future point in time based on the state of the OWT system and the receiving environment after the retrofit. If the system is not completely replaced, some components may be older than others.

For example, the user might be assessing the future risks associated with replacement of a traditional WSAS distribution system (but not the septic tank) after five years of operation, due to improper installation and subsequent failure. The user might select 20 years of overall system operation as the point in time of interest, at which time the retrofitted WSAS distribution system would be in only its fifteenth year of operation. In this example, the age of each component would be specified in the general problem formulation. Then, each component assessment and the integrated assessment would specify if and how the variation in component age is assumed to affect the likelihood and magnitude of adverse impacts.

For purposes of framework development, a set of default temporal reference points has been selected (see the Spatio-Temporal Variables section). They are defined based on the time elapsed since the system was installed. Thus, the framework can be used to estimate future risks when evaluating new systems, existing systems, and retrofit systems; and current risks when evaluating existing systems.

Spatio-Temporal Variables

The spatial and temporal bounds of the framework are not the only aspect of time and space that need to be considered in the problem formulation. Consideration of how changes in time and space (such as distance) will be represented in the input and output variables is also necessary. That is, the data (either measured or estimated values) must have the correct units for each component assessment in which they are used. The units of measure also must be consistent (or readily converted to consistent units) among the component frameworks to ensure that the results can be integrated in the general assessment.

Ideally, changes in condition (such as flow rate, nitrate concentration, and other conditions) can be estimated as a continuous function related, where appropriate, to changes in time or space. Under these circumstances, the condition that will exist at any given point in time (or space) and the point in time (or space) when any particular condition will exist can be estimated. For example, nitrate concentrations in groundwater might be estimated to decrease linearly (with a specified slope and intercept) with distance from the WSAS. From this curve, the nitrate concentration at the boundary of a specific site can be estimated (for example, 5 mg/L at 80 ft from the release point).

Unfortunately, the available data will often not be of sufficient quality or quantity to support this type of quantitative model. In many cases the best that might be achieved are a series of estimated conditions at pre-selected points in time or space. That is, not all outputs can be estimated as continuous variables. For example, the likelihood of surface breakthrough might be estimated as being low after ten years of operation, but high after 20 years of operation. Continuous functions may not be possible because some inputs are nominal or ordinal variables, rather than interval (continuous) variables, or the data are too sparse to derive a continuous function.

Several points in time and space were selected as defaults for use in this framework. The component frameworks were designed to allow estimation of the condition (or risk) at one or more points in time and space. Continuous functions also can be used if they are available. The selected points in time are 1, 10, and 20 years from installation. The selected points in space are 10, 50, and 100 feet down gradient from the discharge point. These reference points in time and space were selected based on common practices for evaluating onsite treatment systems.

OWT System Categories

There are many different specific technologies that can be combined in a variety of ways to make up an onsite wastewater treatment system. Each permutation cannot be addressed separately in a risk assessment framework. Instead, the various treatment technologies must be organized into a relatively small number of groups or categories. Those categories should be based on key characteristics of the technologies with respect to their vulnerabilities and intended functions.

For example, a WSAS can take many different forms (such as beds, trenches, and mounds), be located on sites with different conditions (such as soil type, depth of unsaturated zone, and groundwater characteristics), and be operated in a variety of ways (such as gravity-fed, cyclic dosing, and pressurized dosing). From an engineering risk assessment perspective, a key difference between these various treatment systems is whether or not they include a pump or other electro-mechanical components. Treatment systems with such delivery components have additional vulnerabilities that must be characterized, as compared to simple gravity-fed systems.

Example Categories and Systems

Modern treatment systems are the focus of this framework. These include traditional septic systems and those systems that include more engineered unit operations. Three modern treatment systems are used as examples to facilitate development and communication of the framework. These example systems were selected as representatives of three general categories of modern onsite wastewater treatment systems Table 2-1:

- Traditional
- Contemporary
- Emerging

Note that these categories are for example purposes only and should not be construed as consensus-based labels for assessment or regulatory purposes. Outdated treatment/disposal systems also are included in this framework, because a risk assessment framework for watersheds would need to explicitly include the contributions from all existing types of onsite treatment and disposal systems.

Table 2-1
Example Treatment System Categories and Components Included in the Framework

System Categories	Major System Components			
	Receiving Tank/Pre-Treatment	Tank-Based Advanced Treatment	Soil Infiltration/ Vadose Zone Percolation	Discharge Point
Outdated	Cesspool [a]	None	Incidental	Surface soil or water
Traditional	Septic tank [b]	None	Wastewater soil absorption system (WSAS), gravity-fed	Subsurface to Vadose and saturated zones
Contemporary	Septic tank [b, c]	Aerobic Treatment Unit (ATU)	WSAS, reduced sizing	Subsurface to Vadose and saturated zones
Emerging	Septic tank [b]	Porous Media Biofilter (PMB) and Disinfection	Drip irrigation	Shallow subsurface

Source: Adapted from Siegrist *et al.* 2000; Table 1, p. 6

[a] Straight-pipe systems may have non-functioning septic tanks, but they are assumed in this example to be failing standard treatment requirements.
[b] It is assumed that these septic tanks are designed to be water-tight. Leaks are addressed in the engineering framework as a potential failure mode.
[c] Some ATUs have a built-in trash trap and do not recommend the use of a septic tank in advance of the ATU.

Definition of the four OWT categories and the general types of components that are used to represent those systems was based largely on expected effluent quality for each treatment train. Effluent quality characteristics and values to be used in the examples in this framework are shown in Table 2-2. Reference values and assumptions used to select the assumed effluent quality characteristics, the specific treatment components selected for the example treatment trains, and the key factors considered in the selection process are provided in Appendix B, *Supporting Information: General Problem Formulation*.

Table 2-2
Assumed Effluent Quality Characteristics at the Discharge Point of Each Example OWT System

Effluent Characteristic [a]	Level for Each Example OWT System Type [b]			
	1	**2**	**3**	**4**
Biochemical Oxygen Demand (BOD; mg/L)	350	25	6	10
Totals Suspended Solids (TSS; mg/L)	350	15	3	10
Total Nitrogen (TN; mg/L)	70	54	18	10
Total Phosphorus (TP; mg/L)	10	4.5	6.4	6
Fecal Coliforms (FC; CFU/100 ml)	10^7	10^2	10^3	10 [d]
Virus (V; PFU/ml) [c]	0 to 10^5	0 to 10^2	0 to 10^3	0 to 10 [d]

Note: These values are not intended to be considered as research proven values due to the high variability of soils and design parameters.

[a] Example OWT system categories are described in Table 2-1. Type 1 is a straight pipe disposal system, Type 2 is a traditional OWT system, Type 3 is a contemporary OWT system, and Type 4 is an emerging OWT system.

[b] Discharge point for the straight pipe (1) is end-of-pipe. Discharge point for the traditional OWT (2) and contemporary OWT (3) is the boundary of the defined wastewater soil absorption system. Discharge point for the emerging OWT (4) is a near-surface drip irrigation system. System 3 BOD is CBOD (Carbonaceous Biochemical Oxygen Demand) Fecal coliform is measured as Colony Forming Units (CFU); Virus is measured as Plaque Forming Units (PFU).

[c] Viruses are assumed to be episodically present in the wastewater at high concentrations.

[d] Irradiation with an ultraviolet light disinfection unit is assumed to inactivate 99.99% of the virus particles.

Note that it is important to recognize that the values of specific parameters used in the examples from Table 2-2 and throughout this document, while carefully chosen, are not necessarily the values that would occur or that should be used in risk assessments for a particular system in a particular environment. Users of this framework **MUST** identify and justify appropriate effluent quality characteristics for each and every site-specific assessment.

Outdated systems are defined here as those that are no longer considered adequate even at sites with favorable conditions for onsite treatment (such as adequate drainage and depth to groundwater). They provide little or no treatment prior to discharge to the receiving environment. Outdated systems are included in this framework because they will need to be included in the frameworks for larger spatial scales. However, they are addressed only qualitatively; specific assessment methodologies are not discussed.

Outdated systems may include true disposal (non-treatment) systems such as straight pipe discharges or ancient-style treatment systems that are generally assumed to provide inadequate treatment under current rules and regulations (for example, cesspools and seepage pits). The straight pipe is used as the example outdated system (Figure 2-1). The water released from these systems is assumed to have only slightly reduced amounts of solids and BOD. Exposure is assumed to occur at the site boundary for normally discharged effluent and onsite (in the residence or on the property) for backup and surface breakthrough of wastewater.

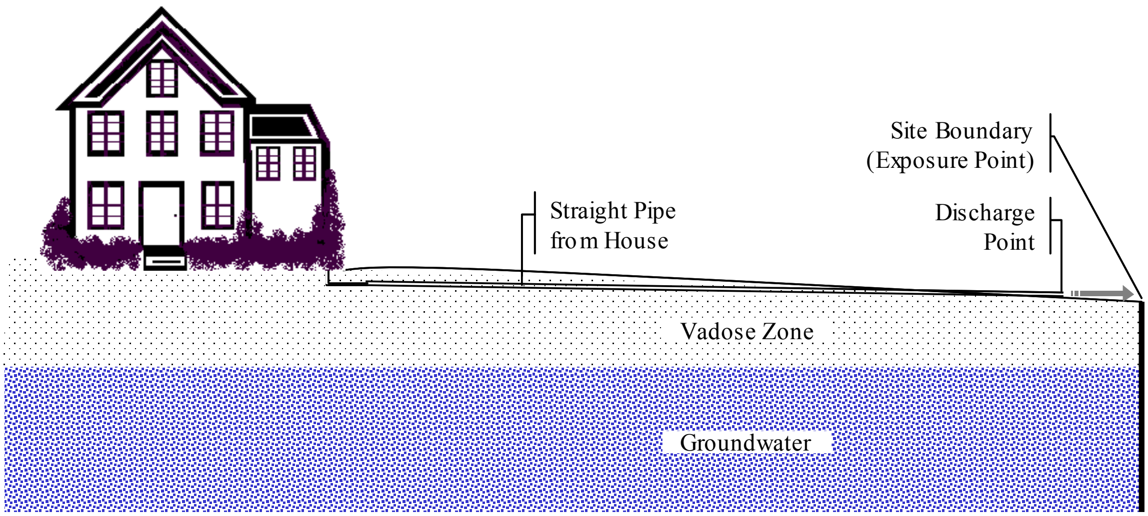

Figure 2-1
Schematic for an Outdated Treatment/Disposal System and the Associated Site Boundary

The cesspool in the example shown in Figure 2-1 is assumed to be providing only minimal treatment (that is, some settling of solids).

Traditional systems are defined herein as those that are typically installed at sites with favorable conditions for onsite treatment. The system selected as the example traditional, onsite treatment system (Figure 2-2) is comprised of a concrete septic tank and a gravity-fed, engineered drainfield that discharges via infiltration into a vadose zone and, after percolation through approximately three feet of soil, into the groundwater with subsequent discharge to the adjoining surface water. STE refers to septic tank effluent and WSAS refers to wastewater soil absorption system.

The septic tank is assumed to have been designed to be watertight (that is, there is only one intended discharge point); however, septic tanks commonly leak in the real world. This reality is addressed in the engineering component by specifying a high probability of subsurface containment failure. Exposure is assumed to occur at the site boundary for normally discharged effluent and onsite (in the residence or on the property) for backup and surface breakthrough of wastewater. The water released from these systems is assumed to have reduced TSS and CBOD. These traditional systems comprise the vast majority of onsite wastewater treatment systems used in the US.

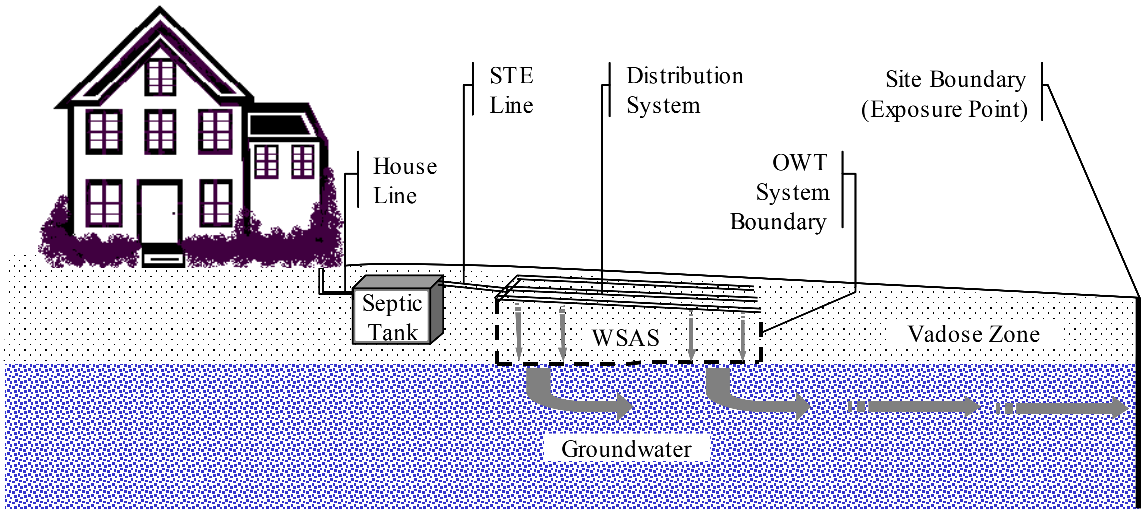

Figure 2-2
Schematic for a Traditional OWT System and the Associated Site Boundary

Contemporary systems are defined herein as traditional systems that have been modified to enhance performance under less-than-favorable site conditions (such as low permeability, high groundwater levels, and other less favorable conditions). They are designed to require less soil treatment than traditional systems. They may also achieve some nutrient removal. The system selected as the example contemporary system (Figure 2-3) also has a septic tank, for solids separation and digestion. The septic tank is coupled with an aerobic treatment unit (ATU). (Note that some ATUs have a built-in trash trap and the use of a septic tank in advance of the ATU is not recommended.)

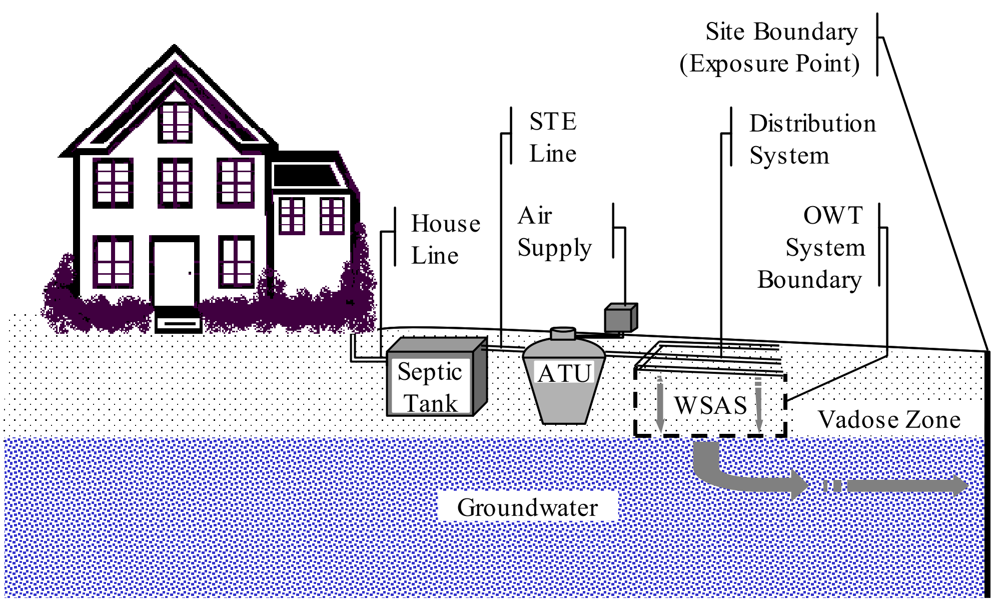

Figure 2-3
Schematic for a Contemporary OWT System and the Associated Site Boundary

As with the traditional system, discharge is via infiltration into a vadose zone and, after percolation through approximately 18 inches of soil, to the groundwater. Exposure is assumed to occur at the site boundary for normally discharged effluent and onsite for backup and surface breakthrough of wastewater. The water released from these systems is assumed to have reduced amounts of CBOD, TSS, and nitrogen.

Emerging treatment systems are defined herein as those that include additional stages or technologies to enhance treatment beyond what can be achieved at a particular site with traditional or contemporary treatment systems. They are designed to require little or no soil treatment prior to discharge to surface water and potable groundwater. Therefore, they generally include some type of disinfection process. The system selected as the example emerging treatment system (Figure 2-4) also has a septic tank as the first component. However, subsequent treatment is provided by a porous media biofilter (PMB) (such as a sand filter) and an ultraviolet (UV) irradiation unit for disinfection. Discharge is assumed to be via a drip irrigation system (not shown in Figure 2-4) to the shallow subsurface. Exposure is assumed to occur at the site boundary for normally discharged effluent and onsite for backup and surface breakthrough of wastewater. The water released from these systems is assumed to have reduced amounts of CBOD, TSS, nitrogen, and viable pathogens.

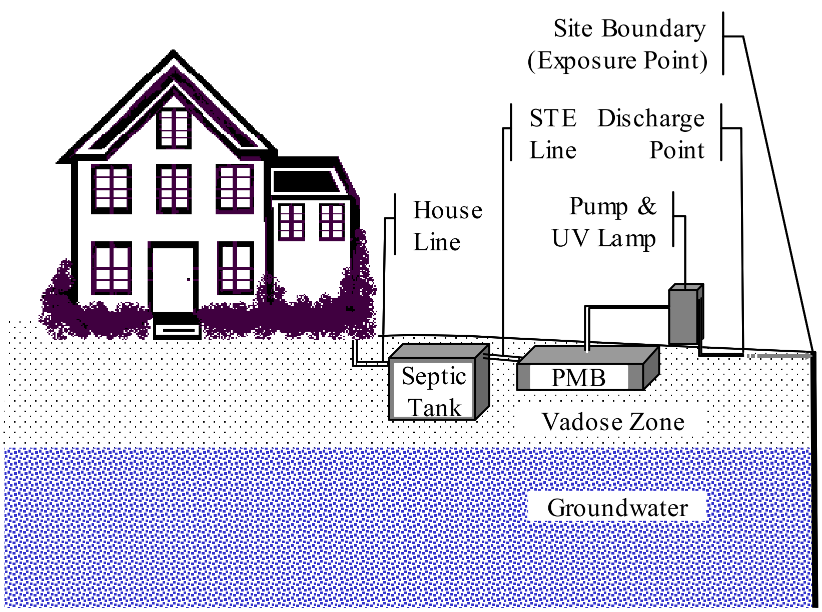

Figure 2-4
Schematic for an Emerging OWT System and the Associated Site Boundary

Retrofit systems are not treated as a separate category in this framework. Instead, the component frameworks are designed to accommodate non-pristine initial conditions and variations in component age.

Operation and Maintenance

For each type of system (and component) there are one or more variables that can be used to determine the state of the system with respect to current performance and the probability of dysfunction. Two of these parameters, operation and maintenance, might be confused with system categories (for example, properly maintained systems versus improperly maintained systems).

All modern wastewater treatment systems require some level of maintenance to ensure proper functioning of the system. Therefore, proper maintenance is a variable to be considered when assessing the likelihood and magnitude of system dysfunction, rather than as a basis for categorizing system types. Traditional septic systems require relatively little maintenance. Wastewater treatment systems that include more advanced treatment technologies may require substantial maintenance. A standard septic tank with a gravity-fed drain field is typically managed (operated and maintained) by the owner. Systems that are more technologically advanced are generally maintained by a service provider with special skills and training. Whether a treatment system is owner-maintained or professionally maintained can affect cost and the likelihood that the system will be maintained properly. Therefore, the level and type of maintenance performed on the system being assessed should be specified in the general problem formulation of the assessment.

Improper operation (misuse) is a separate, though related, factor to be considered when assessing the likelihood and magnitude of system dysfunction. A treatment system can be misused even though it is being maintained properly by the owner or a subcontractor. Misuse might include discharging substances that will damage the treatment system or loading the system beyond the designed capacity. Thus, even proper maintenance may not be able to prevent system dysfunction if the degree of misuse is severe enough. Improper operation is included in the engineering component as a potential cause of system failure (dysfunction). The general types of misuse considered in the framework can also be briefly summarized in the general problem formulation.

Stressors and Receptors

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

Primary stressors and receptors are discussed here, in the general problem formulation. These are the stressors and receptors to which the general framework applies. Each of the component frameworks will address at least one of the primary stressors and receptors. However, the component frameworks may have any number of secondary (intermediate) stressors and receptors that are used within that subcomponent, but not in the general framework. For example, some household chemicals can damage the microbial community in a tank-based component of a treatment system. Those chemicals and microbes might be included in the engineering subcomponent framework as secondary stressors and receptors, because they act on the treatment process rather than on the receptor directly.

Primary Receptors

People are the primary receptors of greatest concern at the micro-scale, because most non-human organisms and systems are best addressed at larger spatial scales (the macro-scale). However, it is important to include ecological and societal receptors (or receptor groups) at the micro-scale so that they can be accounted for properly when assessments for individual sites are combined to yield cumulative impacts at a larger spatial scale. The primary human and ecological receptors for this framework include:

- Human population groups
  - Property owners
  - Occupants, permanent and non-permanent
  - Visitors to an onsite property
  - Vulnerable subgroups
  - Adjacently located subgroups
- Ecological receptors
  - Terrestrial biota
  - Aquatic biota

Primary Stressors

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. Nutrient loading in the form of nitrogen and phosphorus is the primary stressor of concern for ecological receptors. The primary stressors for this framework, include:

- Pathogens (for example, bacteria, protozoa, and viruses)
- Nutrients (nitrogen and phosphorus)
- Unaesthetic features

- Noxious odors
- Nitrate-nitrogen
- Oxygen demand

Note that for all stressors there are two dimensions to consider: stress induced by ordinary function (nominal performance) and stress induced by dysfunction (non-performance).

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, and typically consist of:

- An entity
- A property of that entity that can be measured or estimated
- A level of effect on that property that constitutes an unacceptable risk

However, an appropriate level of effect often cannot be specified in an assessment framework, because accepted default values are not generally available. Therefore, this framework includes some specific metrics, but the user is expected to specify the levels of effect (values for those metrics) that define an acceptable risk for each site-specific assessment. Selecting the levels of effect will generally require input from stakeholders and regulators.

There are two main criteria for selecting assessment endpoints for this framework:

- Susceptibility to the stressors of concern
- Relevance to public policy and management goals

Susceptibility to the stressors of concern is a function of exposure and sensitivity. Exposure is typically defined as co-occurrence or contact of the receptor with the stressor. The likely sources, which are transport and fate of the stressors when selecting assessment endpoints, must be considered. Sensitivity is a function of the mode of action of the stressor and the characteristics of the receptor.

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.

Two additional criteria for assessment endpoints in this framework are that they:

- Can be addressed within the appropriate component assessments
- Are consistently addressed within all of those component assessments

Default Assessment Endpoints

A set of default assessment endpoints was selected for purposes of framework development. These are the endpoints that are most likely to be of concern for users of this framework. The user can add other assessment endpoints to a site-specific assessment by following the guidance provided in Jones *et al.* (2001) and the examples provided in this framework. The default assessment endpoints are listed in Table 2-3.

Table 2-3
Default Assessment Endpoints and the Associated Framework Components

Framework Component	Assessment Endpoints		
	Entity	**Property**	**Specific Metric[a]**
Engineering	Effluent at discharge point	CBOD (mg/L)	350, 25, 6, and 10
		TSS (mg/L)	350, 15, 3, and 10
		TN (mg/L)	70, 54, 18, and 10
		TP (mg/L)	10, 4, 5, 6.4, and 6
		FC (colony forming units per 100 ml)	10^7, 10^2, 10^3, and 10
		V (plaque forming units per ml)	10^5, 10^2, 10^3, and 10
	Indoor Surfaces	Untreated wastewater	Present
	Outdoor Surfaces	Untreated wastewater	Present
Public Health	Onsite resident or visitor, offsite resident or visitor (downgradient), and offsite recreationer (downgradient)	Systematic toxicity from chemical exposure	Hazard Quotient > 1
		Infection by pathogens	Risk of Infection > 1×10^{-4}
Ecological	Estuarine: seagrass population	Decrease in production	Macrophyte biomass density
	Estuarine: benthic invertebrate community	Decrease in abundance or production	Benthic invertebrate biomass density
	Estuarine: fish community	Decrease in abundance or production	Fish biomass density
	Fresh water: phytoplankton community	Increase in production	Algal biomass density

Table 2-3
Default Assessment Endpoints and the Associated Framework Components (Cont.)

Framework Component	Assessment Endpoints		
	Entity	**Property**	**Specific Metric**[a]
Ecological (Cont.)	Fresh water: macrophyte community	Change in production	Macrophyte biomass density
	Fresh water: fish community	Decrease in abundance or production	Fish biomass density
	Fresh water: benthic invertebrate community	Decrease in abundance or production	Benthic invertebrate biomass density
	Fresh water: amphibian populations	Decrease in abundance or production	Amphibian biomass density
Socioeconomic	Property owner, resident, adjacent property owner/ resident, and vulnerable populations	Economic status	Design and installation cost for the OWT system ($) OWT system inspection, maintenance, and repair costs ($) Opportunity cost (time value of money for design, installation and maintenance costs) ($) Change in property value associated with the OWT system ($)
		Convenience	Difficulty and time spent on system maintenance (hours/month) Limitations on water use (alarms if storage limits are exceeded and limited use of garbage disposal and washing machines) (yes/no)

Table 2-3
Default Assessment Endpoints and the Associated Framework Components (Cont.)

Framework Component	Assessment Endpoints		
	Entity	**Property**	**Specific Metric[a]**
Socioeconomic (Cont.)	Property owner, resident, adjacent property owner/ resident, and vulnerable populations	Aesthetic quality	Intrusiveness of OWT system in the visual landscape (mounds, risers) (ordinal rating) Indirect impacts on visual landscape (if pond or wetland is created with re-use of graywater) (ordinal rating) Changes in noise levels (due to pumps and blowers) (ordinal rating)
		Privacy	Change in presence of odors (ordinal rating) Change in ability of property owner to determine land use (ordinal rating) Intrusions by outsiders to maintain/monitor the OWT system (occurrences/month)
		Equity	Willingness to bear cost for other's benefit (pay higher cost for OWT system than neighbors have paid for wastewater disposal) (ordinal rating)

[a] Specific metrics are listed instead of levels of effect because an acceptable level of effect cannot always be specified in an assessment framework (widely accepted default values are not available). That is, appropriate levels of effects for these assessment endpoints may be widely variable among sites, dependent on stakeholder and regulator input, and controversial.

Conceptual Model

The conceptual model is used to describe and to depict visually the expected relationships among the stressors, exposure pathways, and receptors (assessment endpoint entities). Only those relationships considered in the assessment are included in the model. The assumptions used to develop the conceptual model are described in the supporting text. Relationships that cannot or will not be addressed are identified, and a rationale for the exclusion of prominent relationships is included.

This model differs from the example micro-level model provided in Jones *et al.* (2001) in that only one source is identified—the residence that the treatment system serves (the ultimate source). All of the proximate sources (such as groundwater and surface water) are included as components of the exposure pathways. The boundaries of the treatment system are more explicit in this conceptual model.

A generic conceptual model for the transport of wastewater and its constituents (such as organic material, nutrients, and pathogens) is presented in Figure 2-5. Solid-lined arrows indicate potential flow paths for wastewater and its constituents under normal operating conditions, dashed-lined arrows indicate flow paths that result **only** from system dysfunction, and rounded boxes indicate potential receptors (assessment endpoint entities).

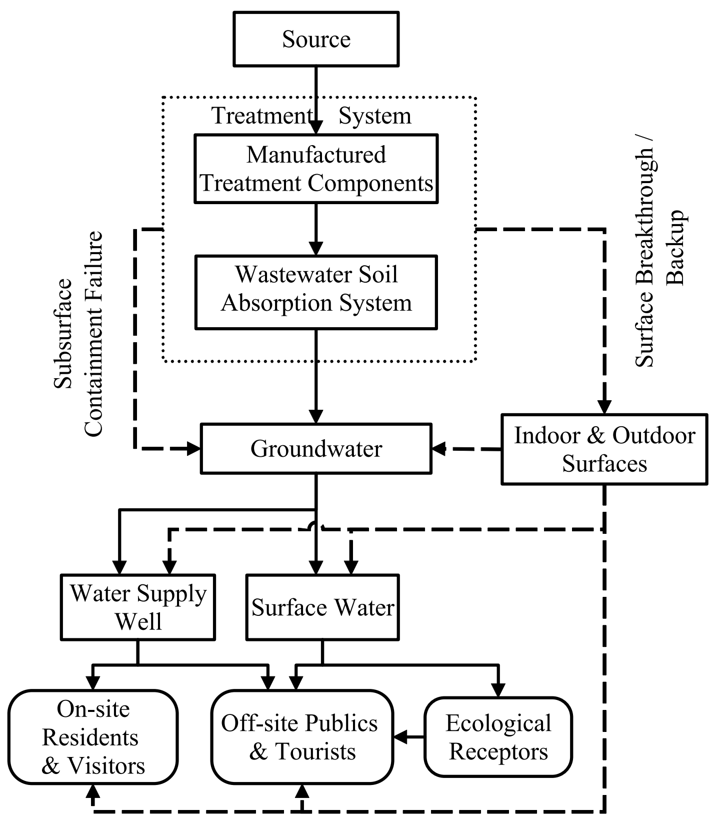

Figure 2-5
Generic Conceptual Model of the Exposure Pathways for Wastewater and Its Constituents From an Individual Onsite Treatment System

This generic conceptual model is sufficient for the general problem formulation. However, each component framework (engineering, public health, ecological, and socioeconomic) also includes conceptual models specific to the stressors and receptors addressed in that component. Those models include details not depicted in Figure 2-5. For example, the conceptual model in the ecological risk assessment component expands upon the Ecological Receptors box in Figure 2-5 to provide explicit assumptions regarding the relationships among various ecological receptors.

The assumptions made for this framework are presented in this section as follows and depicted in Figure 2-5. The source is assumed to be the year-round residence of a single family of four with an average daily loading rate of approximately 280 gallons (approximately 70 gallons per capita per day, based on water usage studies cited in US EPA 2002). However, the framework is flexible enough to accommodate other assumptions (such as seasonal occupation, multi-family homes, or restaurants) with some modification. The wastewater treatment system includes not only the manufactured system components, but also the soil environment in which additional treatment is expected.

Backup of the treatment system is assumed to result in contamination of indoor surfaces and exposure to residents and visitors inside the residence. Surface breakthrough of untreated or inadequately treated wastewater can result from structural failure of any of the manufactured or constructed system components. Contamination of the land surface is the assumed exposure pathway for people using the site. Overland flow of surfacing wastewater also can enter drinking water wells through cracked casings, seep into groundwater through permeable surfaces, or cross property boundaries resulting in direct exposure to offsite publics and tourists. Transmission of pathogens via insect and animal vectors (such as flies and domestic pets) was not considered explicitly, for the purpose of simplifying the framework. The stressors for people exposed to contaminated indoor and outdoor surfaces include pathogens, household chemical constituents, and unpleasant odor and appearance.

Structural failure of engineered components can also result in subsurface containment failure with subsequent contamination of the groundwater with inadequately treated wastewater. The solid-line arrow indicates that wastewater treated by the soil absorption system is expected to flow into the local groundwater. However, groundwater contamination can result from inadequate soil-treatment of the wastewater. The possible causes of this dysfunction (inadequate separation from groundwater) are discussed in the engineering component assessment.

Contaminated groundwater may intersect a water-supply well on, or adjacent to, the site being assessed. Potential receptors are population subgroups, including residents and visitors both on and off the site. Other populations, such as the local community, transient workers, and tourists, may be exposed if the well is used for public water supply or commercial uses. The primary stressors of concern are pathogens, nitrate-nitrogen, and, to a lesser extent, household chemicals.

Surface water contamination is assumed to occur primarily via discharge of contaminated groundwater. The solid-line arrow indicates that wastewater discharged from the soil absorption system is ultimately expected to flow into surface water via the groundwater. Runoff of untreated wastewater on the soil surface may also result in contamination of surface water.

Potential human receptors include the population subgroups described previously. Direct exposure pathways include consumption of water via a public or private water supply intake and incidental water consumption while swimming, fishing, and boating. The stressors of concern include pathogens, nitrate-nitrogen, and, possibly, household chemicals. Indirect exposure may also occur via consumption of contaminated shellfish, for which the stressor of concern is pathogens. Surface water contamination is the primary exposure pathway for ecological receptors.

Generic receptors include aquatic plants and animals. Stressors of concern include nutrients (nitrogen and phosphorus), oxygen demand, and household chemical constituents. Specific pathways and receptors are detailed in the ecological risk assessment component.

Note that the relationships among socioeconomic stressors (such as costs and social issues) and receptors are not depicted in this conceptual model. Those relationships are addressed in the socioeconomic component assessment. This generic model also does not include details of the differences between each of the example treatment systems.

Measures

Measures are attributes that can be estimated or measured directly. The selected measures must establish a link between the stressor and the assessment endpoint. US EPA (1998b) identified three types of measures:

- **Measures of effects**—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**—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**—Measures of environmental attributes that influence the distribution of a stressor (for example, soil temperature and depth to groundwater) or receptor attributes that influence exposure and response (for example, age and behaviors)

These measures may be either direct measurements of the assessment endpoint or surrogate measures for the assessment endpoint, should direct measurement be impossible or impractical.

Measures that are expected to be most appropriate for the default assessment endpoints are discussed in the respective component frameworks. As with the assessment endpoints, the user can add other measures to a site-specific assessment. Example measures from each component framework are listed in Table 2-4.

Table 2-4
Example Measures for each Component Framework and the Associated Assessment Endpoints

Framework Component	Assessment Endpoint	Measures of		
		Effects or Consequence	**Exposure or Occurrence**	**System or Receptor Characteristics**
Engineering	Virus concentration in effluent from contemporary system at discharge point	Estimated magnitude and duration of virus loads at WSAS-groundwater interface.	Frequency of WSAS saturation by groundwater	Separation between WSAS and groundwater
Public Health	Risk of infection of offsite residents or visitors (downgradient) by pathogens	Observed incidence of infection	Predicted or observed virus concentrations in downgradient water supply well.	Groundwater travel time to exposure point Flow and volume of groundwater Immune status of receptors
Ecological	Decreased production of seagrass population	Measured biomass density of macrophytes	Concentration of nitrogen Light penetration	Flushing rate of estuary or lagoon
Socioeconomic	Economic status of property owner	Change in property value	Design and installation cost	Ability of owner to pay Property value

3 ENGINEERING COMPONENT FRAMEWORK

This component framework is used to address the issues specific to the design and performance of the OWT system being evaluated. Generally it follows the three-stage risk assessment format used throughout this framework (problem formulation, analysis, and risk characterization). However, the engineering component framework uses the risk-analysis methodology called Failure Modes and Effects Analysis (FMEA). The process flowchart used for this component framework is shown in Figure 3-1. Each aspect of the framework is described in detail in this chapter.

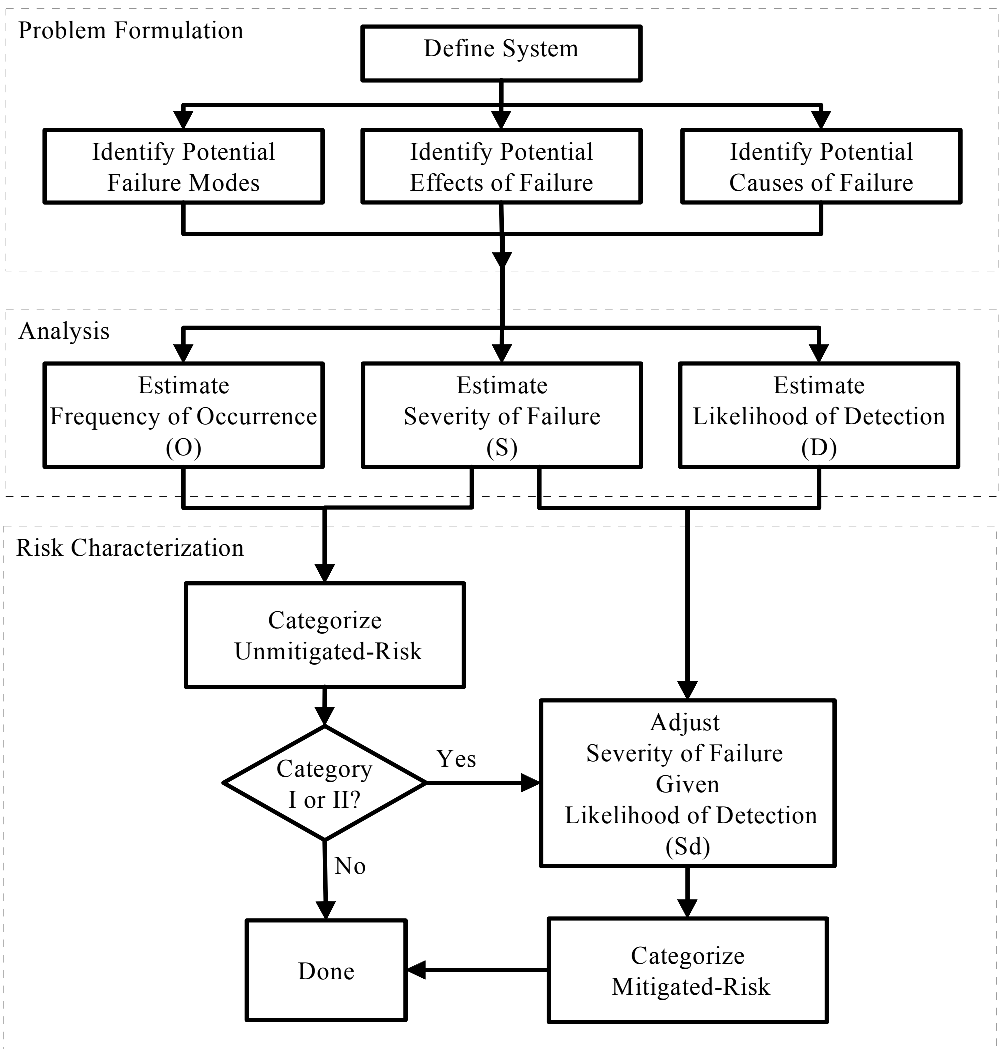

Figure 3-1
FMEA Process Flowchart

Problem Formulation

The scope and objectives for the overall (integrated) risk-assessment framework were defined in Chapter 2 in the General Problem Formulation section. However, a problem formulation is also required for each component framework. Whereas the general problem formulation provides a summary of the issues to be addressed throughout the framework, the engineering problem formulation provides a detailed discussion of the scope and objectives of this component framework.

FMEA

FMEA is an inductive analysis in which a detailed systematic component-by-component assessment is made of all possible failure modes and their resulting effects on a system. Possible single modes of failure or malfunctions of each component in a system are identified and analyzed to determine the effect on surrounding components and the system (Henley and Kumamoto 1981).

One of the challenges in the building of wastewater treatment systems is that the processes have to be designed to meet a wide variety of circumstances as the wastewater composition and loads might change on an hourly basis. Because each location has its own characteristics, designs suitable for one location may be unsuitable at another. As a result historical precedence, where it exists, will provide at best an indicator (that is, learning/statistics are not fully transferable). More complex designs (active systems) may create a greater risk potential because of the increased failure probability (Frodsham and Cardew 2000) of the active components versus that of passive components.

The process of conducting a FMEA can be examined in two levels of detail. The first level of analysis consists of identifying potential failures and the effects on a systems' performance by identifying the potential severity of the effect. The second level of analysis consists of additional steps for calculating the risk of each failure through measurements of the severity and probability of a failure effect.

A FMEA table is shown in Table 3-1. The approach used to complete the FMEA table, using an OWT system for an example, is presented in the following sections.

Table 3-1
Example FMEA Worksheet for an OWT System Process and Component [a]

PROCESS: Ingress to Septic Tank				COMPONENT: House Sewer Line					
Process Requirement	**Failure Mode**	**Causes(s) of Failure**	**OCC**	**Effects of Failure**	**SEV**	**Process Control/Detection and Correction**	**DET**	**Risk Ranking [b]**	
								Unmit.	**Mitig.**
Move wastewater from house to septic tank	Insufficient flow/plugged	Excessive use of disposal	5	Wastewater remains in house with potential exposure of residents to pathogens	9	User training/failure mode is obvious	4	II	III
		Fats or grease plug line	4		9	User training/failure mode is obvious	4	II	III
		Excessive toilet paper	6		9	User training/failure mode is obvious	4	II	III
		Too many bends in sewer line	3		9	Installer training and inspections/failure mode is obvious	3	II	IV
		Sewer line pitch too small	4		9	Installer training and inspections/failure mode is obvious	3	II	III
		Tree root intrusion	5		9	Landscape placement/failure mode is obvious	5	II	III
	Leak/Rupture	Improper material selection	4	Untreated wastewater leaks into soil and groundwater and potentially reaches a water supply well or surface water	7	Installer training and inspections/failure mode NOT obvious	8	III	N/A
		Poor construction/ installation	6		7	Installer training and inspections/failure mode NOT obvious	8	III	N/A
		Crushed by vehicles driving over shallow pipe	5		8	Owner training/failure mode NOT obvious	8	III	N/A

Note that the values used in Table 3-1 and throughout this framework are for example purposes only and that specific values would need to be determined or estimated for use in an actual risk assessment.

[a] Occurrence (OCC), severity (SEV), detection (DET), and risk ranking values are discussed in detail later in this framework.
[b] Unmit. = unmitigated, Mitig. = mitigated, N/A = not applicable (unmitigated risks are considered acceptable).

Define the System and Its Requirements

FMEA is an inductive process in which the effect of a single point failure on the overall performance of a system is examined through a “bottom-up approach.” FMEAs are used on single components or stages of a system. The aim of FMEAs is to identify all causes of failure (Military Standard 1980; Ford Motor Company 1996; and NASA John H. Glenn Research Center, Lewis Field 2002).

All types of failure categories are considered in FMEAs, including:

- Failure to meet regulatory or quality parameters
- Failure to achieve required volumetric throughput (insufficient processing capacity)
- Potential injury to homeowners
- Potential injury to others (offsite)
- Potential environmental damage

For each of the functions listed, a failure identification exercise is undertaken. A failure mode is the manner in which the function has failed. For example, failure modes for mixing liquids would include no mixing, too vigorous mixing, insufficient mixing, and mixing of the wrong thing.

The first step in performing a FMEA is to organize as much information as possible about the system concept, design, and operational requirements. By organizing the system model, a rationale, repeatable, and systematic means to analyze the system can be achieved.

FMEA is a systematic way of assuring that every conceivable potential failure mode of a design/process has been considered with the objective of minimizing the probability of failure. The purposes of a FMEA are to

- Assist in selecting design alternatives with high reliability and high safety potential during early design phase
- Ensure that all conceivable failure modes and their effects on operational success of the system have been considered
- List potential failures and identify the magnitude of their effects
- Develop early criteria for test planning and the design of the test and check-out systems
- Provide a basis for quantitative reliability and availability analyses
- Provide historical documentation for future reference to aid in analysis of field failures and consideration of design changes
- Provide input data for trade-off studies

- Provide a basis for establishing corrective action priorities
- Assist in the objective evaluation of design requirements related to redundancy, failure detection systems, fail-safe characteristics, and automatic and manual override

The FMEA may also be performed with limited design information by assuming the following basic questions:

- How can each part conceivably fail?
- What mechanisms might produce these modes of failure?
- What could the effects be if these failures did occur?
- Is the failure in the safe or unsafe direction?
- How is the failure detected?
- What inherent provisions are provided in the design to compensate for the failure?

Before undertaking a FMEA, it is essential to undertake certain preparatory steps. The scope of the analysis depends on the complexity of the system/component being studied and requires the following information:

- Definition of the system/component to be analyzed and its mission
- Description of the operation of the system
- Identification of failure categories
- Description of the environmental conditions

Individual function(s), conditions, and requirements of the subsystems being analyzed must be identified and evaluated for each failure mode. When the subsystem has many functions with different potential failure modes for each function, each function is listed separately. Complicated technologies or processes are broken down into functions, called functional nodes. The systems are defined in terms of their primary functional activity. For example a pumped system would be examined under the functional node of liquid movement or perhaps mixing (depending on the purpose of the system).

For the traditional wastewater treatment system shown in Figure 2-2, the subsystem requirements are to

- Move wastewater from the house to the septic tank
- Enable settling of solids from the wastewater (in the septic tank)
- Distribute wastewater effluent from the septic tank to the leaching system
- Filter wastewater effluent from the septic tank through soils prior to reaching the atmosphere or groundwater
- Retain floatables and grease in the septic tank

Identify Potential Failure Modes

Potential failure modes are defined as the manner in which systems/components could potentially fail to meet their intended function. Identifying the potential failure modes covers every way in which the part could fail and includes random and degradation failures. The question is not: "Will it fail?" but: "How could it fail?"

The failure mode is the manner that a failure is observed in a function, subsystem, or component. Failure modes of concern depend on the specific system, component, and operating environment. The past history of a component/system is used in addition to understanding the functional requirements to determine the failure modes.

Several common failure modes include:

- Complete loss of function
- Uncontrolled output
- Premature/late operation

The cause of a failure mode is the physical or chemical processes, design defects, quality defects, part misapplication, or other methods, that are the reasons for failure. Note that more than one failure cause is possible for a failure mode; it is important to identify all potential causes of failure modes, including human error.

Each potential failure mode for the particular subsystem function is listed with the assumption that the failure could occur, but may not necessarily occur. The task of identifying subsystem failure modes can take either of two approaches:

- **Functional**—Involves listing each subsystem, its functions, and the failure modes leading to the loss of each function.
- **Hardware**—Involves listing each part and its probable failure modes. The hardware approach is used most often when detailed part design information is available.

With either approach, the potential failure modes are identified by asking:

- In what way can this subsystem fail to perform its intended function?
- What can go wrong although the subsystem is manufactured/assembled to specifications?
- If the subsystem function were tested, how would its failure mode be recognized?
- How will the environment contribute to or cause a failure?
- In the application of the subsystem, how will it interact with other subsystems?

Analysts enter the potential failure mode(s) for each function in the "Failure Mode(s)" column in the FMEA table (see Table 3-1). Potential failure modes should be described in physical or technical terms, not as a symptom noticeable by the homeowner. Analysts should not enter trivial failure modes (that is, failure modes that will not, or cannot, occur).

General types of failure modes for the functional approach include

- Failure to operate at the prescribed time
- Failure to stop operating at the prescribed time
- Intermittent operation
- Wear out
- Degraded output

General types of failure modes for the hardware approach for the traditional wastewater disposal system include

- Plugged
- Too little flow
- Too much flow
- No settling
- No anaerobic activity
- Leak/rupture
- Flooding

Additional failure modes for more complex OWT systems could include

- Warped
- Corroded
- Loose
- Misdirected
- Fails to perform task (human error)
- Performs task improperly (human error)

Identify Potential Cause(s) of Failure

Associated with each failure mode is a list of every conceivable potential cause. There are often two types of causes of failure: first-level causes and root causes. A first-level cause is the immediate cause of the failure mode, which will directly make the failure mode occur. A root cause(s) may be below the first-level cause, and will ultimately lead to the first-level cause and the failure mode. For example, a first-level cause would be material cracked because of overstress, while the root or second-level cause is that the material is too thin because of inadequate design. The list of potential first-level causes should be as complete as possible.

All causes of a failure mode can also be divided into the following two categories of causes:

- Design deficiency
- Process variation that can be described in terms of something that can be corrected or can be controlled

Potential cause(s) of each failure mode can be identified by asking questions such as:

- What could cause the subsystem to fail in this manner?
- What circumstance(s) could cause the subsystem to fail to perform its function?
- What can cause the subsystem to fail to deliver its intended function?

For the traditional wastewater example, the cause of a building sewer line plugging could be excessive use of the disposal or fats or grease in the OWT system. The septic tank could crack because of poor construction materials or overburden pressures. Examples of causes of failure for the different failure modes for the components in the example traditional wastewater system are listed in Table 3-2.

Table 3-2
Example Causes of Failure for Different Failure Modes in the Traditional Wastewater Treatment System

Component	Failure Mode(s)	Example Cause(s) of Failure
Building sewer	Plugging	• Excessive use of disposal • Fats or grease that plug line • Root intrusion
	Leak/rupture	Cracked or broken pipe due to construction damage
Septic tank	Effluent quality consistently much lower than typical expected values	• Tank(s) sizing too small • Peak flows unusually high • Raw wastewater higher strength than typical • Loss of biological activity due to discharge of harmful chemicals to the septic tank • Residual solids not pumped from tank as required
	Periodic "burping" of solids into effluent	• Tank(s) sizing too small • Peak flows unusually high • Residual solids build up in the tank(s)
	Tank leak/rupture	• Crack in tank due to poor construction materials • Crack in tank due to overburden pressures or soil movement

Table 3-2
Example Causes of Failure for Different Failure Modes in the Traditional Wastewater Treatment System (Cont.)

Component	Failure Mode(s)	Example Cause(s) of Failure
Effluent distribution unit	Localized overloading to soil	• Pump or siphon failure • Improper piping design and installation • Pipe blockage due to wastewater solids or root intrusion
	No flow to soil system	Broken pipe due to construction damage or soil movement
	Piping leak/rupture	Cracked or separated piping due to tree root invasion or soil movement
Soil infiltration system	Septic tank effluent backing up into home or surfacing to the ground	Infiltration capacity less than daily loading due to: • Construction damage • Excessive soil clogging due to high BOD and suspended solids • Daily wastewater flow higher than design • Infiltration and inflow increases to loading
	Septic tank effluent contaminating groundwater	• Poor siting of the soil infiltration system and inadequate depth of unsaturated soil • Hydraulic overloading due to poor design, excessive flow, or distribution system malfunctions
	Periodic flooding	• Siting within a floodplain • Poor grading of soil infiltration area

Note that these potential causes and failure modes are for example purposes only and that the user must identify all causes and failure modes that are appropriate for an actual risk assessment.

Identify Potential Effect(s) of Failure

Analysts must briefly describe the consequences of failure. One failure mode could have more than one effect, and the same effect could apply to a number of different failure modes. Potential effects of failure are defined as the consequence(s) of the failure mode on the subsystem, described in terms of safety.

For the OWT, the effects of the failures are based on effluent characteristics at the endpoints of interest given in. For purposes of framework development, effects of effluents above the values given in are assumed to result in personnel injury or sickness. However, it should be clearly understood that the values used in Table 3-3 (and Table 2-2) are for example only and that specific values would need to be determined or estimated for the type(s) of OWT and site and soil conditions applicable in an actual risk assessment being conducted under “real world” conditions.

Table 3-3
Example Assessment Endpoints for the Engineering Component Framework

Assessment Endpoints		Example Levels for Each OWT System Type			
Endpoint Entity	Endpoint Property	Straight Pipe Disposal [a]	Traditional OWT [b]	Contemporary OWT [b]	Emerging OWT [c]
Effluent	CBOD (mg/L) at discharge point	350	25	6	10
Effluent	TSS (mg/L) at discharge point	350	15	3	10
Effluent	TN (mg/L) at discharge point	70	54	18	10
Effluent	TP (mg/L) at discharge point	10	4.5	6.4	6
Effluent	FC (cfu/100 ml) at discharge point	107	102	103	101
Indoor Surfaces	Untreated wastewater	Present	Present	Present	Present
Outdoor Surfaces	Untreated wastewater	Present	Present	Present	Present

[a] Discharge point is at the end-of-pipe (see Figure 2-1).

[b] Discharge point is the boundary of the defined wastewater soil absorption system (see Figure 2-2 and Figure 2-3).

[c] Discharge point is at the drip irrigation point (see Figure 2-4).

Quantitative Analysis

In this stage of the assessment, the user moves from qualitative assessments to quantitative (or semi-quantitative) estimation. The process changes from identifying potential modes, causes, and effects of failures to estimating the likelihood and magnitude of the failures.

Estimate the Probability of Occurrence

Occurrence (OCC) is the likelihood that a specific cause of failure will occur and is a rating that corresponds to the likelihood that a particular failure mode will occur within a specific time period. Each cause listed in the FMEA table requires an estimate of its possible failure rates and/or its mean time between failure probabilities. The occurrence of failure can be based upon historical data, including the service history, warranty data, and maintenance experience with similar or surrogate parts.

Questions to consider include:

- How will the potential failure be detected? (Some failures are obvious to the person using the subject of the FMEA; however, if this is not the case, the means by which the failures can be detected should be listed.)
- What statistical data are available from previous or similar process designs?
- Is the process a repeat of a previous design, or have there been some changes?
- Is the process design completely new?
- Is the environment in which the process is to operate changeable?
- Have mathematical or engineering studies been used to predict failure?

Some example occurrence evaluation criteria and ranking bins were developed for the wastewater treatment system options and are provided in Table 3-4. Note that these criteria and rankings are for example purposes only; the user is expected to develop appropriate criteria for each assessment.

Table 3-4
Example Frequency Scale for Time-Independent Events

Probability of Failure (annual basis)		Expected Rate of Occurrence	Rank
Very High: Failure is Almost Inevitable	>1.0	More than once per year	10
	1.0	Once per year	9
High: Repeated Failures	0.50	Once every two years	8
	0.33	Once every three years	7
Moderate: Occasional Failures	0.25	Once every four years	6
	0.20	Once every five years	5
	0.10	Once every 10 years	4
Rare: Relatively Few Failures	0.05	Once every 20 years	3
	0.03	Once every 30 years	2
Extremely Rare: Failure is Unlikely	0.02	Once every 50 years	1

Note: These rankings are for example purposes only. The user must assign rankings that are appropriate for the site and systems being evaluated in each assessment.

Occurrence probabilities can be based on the frequency of the initiating event (for example, seismic events or floods), the independent failure rate of components (for example, valves or piping), or historical experience/engineering judgment (for example, saturation of leach fields).

Storms and floods are typically classified based on the time interval between events of similar magnitude. For example, a 100-year flood is an event that occurs, on average, once every 100 years. Such events are assumed to have a probability of occurrence of 1×10^{-2} (that is, 0.01) in any given year, regardless of when the last event of similar magnitude actually occurred. Thus, the probability of a 100-year flood on an annual basis is the same in the first year of operation as it is in the twentieth year of operation.

The probability of a 100-year flood occurring at least once over a 20-year OWT system lifetime can be approximated by multiplying the annual probability times the number of years of service (that is, $0.01 \times 20 = 0.20$). Thus, if a 100-year flood is the minimum size flood expected to fail the system (or subsystem), then the example OCC rank from Table 3-4 would be Moderate (5). This formula for estimating the probability of a time-independent event is likely to be sufficiently accurate for most assessments of OWT systems. However, a more exact formula is provided in Appendix C, *Supporting Information: Engineering* for assessors who wish to use it.

Many of the failure (dysfunction) events associated with OWT systems are best described as being independent component failures. The failure probabilities for these events are based on the component failure rates, as described in Appendix C, *Supporting Information: Engineering*. The failure rate, $\lambda\ (t_x)$, is the probability of failure during a given period of time (for example, t_{10} can be the first ten years of operations). Because FMEA analyses typically concentrate on single failures (that is, the failure of a single component causes the system or subsystem to fail), $\lambda\ t$ is also the expected number of failures (ENF). Thus

$$ENF_i = \lambda_i t \qquad \textbf{Equation 3-1}$$

for both repairable and non-repairable systems (Fussell 1975).

Examples of independent failures include mechanical failure of pumps, structural failure of pipes and tanks, and hydraulic failure of wastewater soil absorption system (WSAS). A constant failure rate (that is, λ = constant) means that the probability of occurrence of these independent failure events increases as a function of time. Thus, if the failure rate for a septic tank is once in 10 years (0.10/year) and the projected service time for the OWT system is 20 years, then the expected number of failures is 2.0 over its lifetime ($0.10 \times 20 = 2.0$). From Table 3-4, the example OCC rank is 10; the probability of failure is considered to be "very high: failure is almost inevitable."

There will be instances where component failure rates and initiating event frequencies are unavailable or inappropriate. In these cases, if the user has good empirical data for operation of a particular component, curves representing the probability of failure over time can be generated. For example, the buildup of phosphorous in a leach field will not cause the system to fail until some saturation point is reached. In this case, the failure probability will be some exponential distribution near the time when the saturation point is expected (see Figure 3-2). Thus, if a leach field is expected to work for 20 years, its rank would be 3 (see Table 3-4).

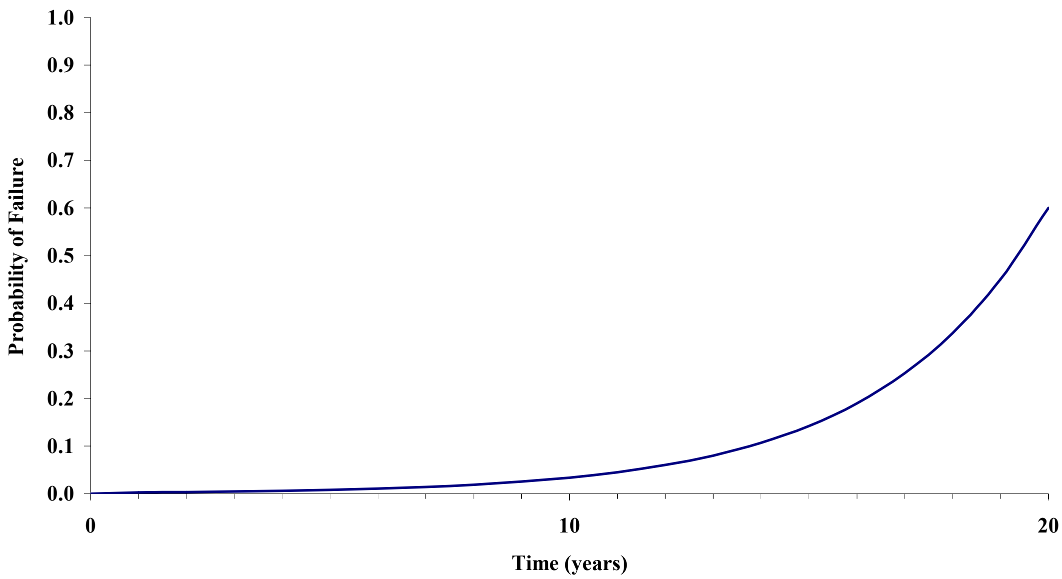

Figure 3-2
Hypothetical Probability Distribution for Failure Data for a Leach Field

Estimate Severity

The severity scales (SEV) used in the FMEA for individual OWT systems will vary by assessment endpoint, though some similarities exist among all endpoints. The severity scale is designed to yield a score:

- Between 7–10 for events that warrant action
- Of 6 for events that may warrant attention
- Between 1–5 for events that are considered negligible

The severity of the impact for each failure mode is assessed and classified according to the rankings outlined in a severity table that is appropriate for the sites and systems being evaluated. For example, Table 3-5 is a typical severity table that has been modified to reflect example wastewater treatment disposal options. To show that each particular failure mode has been considered, a no-effect category should be included. It is important to note that these criteria and rankings are for example purposes only; the user is expected to develop appropriate criteria for each assessment.

Table 3-5
Example Severity (SEV) Evaluation Criteria

Effect	Severity of Effect	Ranking
Extreme	A failure that is expected to cause serious damage to property, serious injury or death, and serious environmental damage.	10
Severe	A failure that might cause serious damage to property, serious injury or death, and serious environmental damage.	9
Very high	An event that would result in immediate failure resulting in potential environmental damage or significant damage to property. Possible personal injury to residents and general public.	8
High	A failure that would engender a high degree of owner/resident dissatisfaction and potential personal injury to residents and/or the general public, and/or potential damage to property.	7
Moderate	A failure that would cause discomfort or annoyance to owners or residents and/or result in reduced process capability.	6
Low	A minor failure that would cause only slight annoyance or minor deterioration in quality. A temporary effect that would not result in a safety risk to people.	5
Very low	Component/system operable, but possesses some noticeable defects (aesthetic and otherwise).	4
Minor	A very minor failure that would have no noticeable effect on the quality of the effluent or the safety of owners, residents, or the general public.	3
Very minor	Component/system operable, but is in noncompliance with manufacturer's recommended operational conditions.	2
No effect	A failure that has no effect on the failure modes.	1

Severity is a rating corresponding to the seriousness of the effect(s) of a potential equipment failure mode. Severity is comprised of two components: safety considerations to the homeowner or public and equipment downtime or performance. Health and safety of the homeowners and offsite personnel are often the primary criteria in determining the SEV ratings for OWT systems.

Example severity scales developed specifically for OWT issues are provided for each of the engineering assessment endpoints as follows in Table 3-6 through Table 3-10. These tables were developed to account for the differing concentration levels that could be present at different times (that is, the failure modes are not binary).

The values in these tables are examples only. Users of this framework are expected to modify these scales and assumptions to suit their particular application (US EPA 1994). Selection of appropriate severity scales is best accomplished through a Data Quality Objectives (DQO) process or some other similar process.

Table 3-6
Severity Scale for Carbonaceous Biochemical Oxygen Demand (CBOD), Total Suspended Solids (TSS), and Total Nitrogen (TN) Based on Event Magnitude and Duration

Relative Magnitude (discharge conc./ criterion conc.)	Event Duration (days)					
	<1	1	7	14	28	>28
≥ 5.0	5	9	9	10	10	10
≥ 2.0 < 5.0	5	7	8	9	10	10
≥ 1.0 < 2.0	5	6	7	8	9	10
≥ 0.8 < 1.0	3	3	4	4	5	5
≥ 0.5 < 0.8	2	2	2	2	3	3
< 0.5	1	1	1	1	1	1

Table 3-7
Severity Scale for Total Phosphorus (TP) Based on Event Magnitude and Duration

Relative Magnitude (discharge conc./ criterion conc.)	Event Duration (days)					
	<1	1	7	14	28	>28
≥ 2.0	5	9	9	10	10	10
≥ 1.5 < 2.0	5	7	8	9	10	10
≥ 1.0 < 1.5	5	6	7	8	9	10
≥ 0.8 < 1.0	3	3	4	4	5	5
≥ 0.5 < 0.8	2	2	2	2	3	3
< 0.5	1	1	1	1	1	1

Table 3-8
Severity Scale for Fecal Coliform (FC) Based on Event Magnitude and Duration

Relative Magnitude (discharge conc./ criterion conc.)	Event Duration (days)					
	<1	**1**	**7**	**14**	**28**	**>28**
≥ 100	5	9	9	10	10	10
≥ 10 < 100	5	7	8	9	10	10
≥ 1.0 < 10	5	6	7	8	9	10
≥ 0.1 < 1.0	3	3	4	4	5	5
< 0.1	1	1	1	1	1	1

Table 3-9
Severity Scale for Wastewater on Outdoor Surfaces, Based on Presence/Absence and Duration

Magnitude	Event Duration (days)					
	<1	**1**	**7**	**14**	**28**	**>28**
Present	6	6	9	9	10	10
Absent	1	1	1	1	1	1

Table 3-10
Severity Scale for Wastewater on Indoor Surfaces, Based on Presence/Absence and Duration

Magnitude	Event Duration (days)					
	<1	**1**	**7**	**14**	**28**	**>28**
Present	7	8	10	10	10	10
Absent	1	1	1	1	1	1

Severity of a failure event (or routine performance level) has two components: magnitude and duration.

Magnitude

Magnitude refers to the concentration of an effluent characteristic discharged during an event. For the five standard effluent characteristics (CBOD, TSS, TN, TP, and FC), magnitude is expressed as an effluent concentration (for example, mg/L). Wastewater backup into the residence and wastewater discharge to the soil surface are expressed simply as present or absent (Table 3-9 and Table 3-10), which assumes that any amount warrants action.

Concentration of an effluent characteristic can be expressed as an absolute concentration or as a relative magnitude (that is, expected concentration in the discharge relative to the concentration criterion for that effluent characteristic). The relative magnitude values used in the severity scale for a given effluent characteristic depend on the range of concentrations that are likely to occur in the effluent. The highest concentrations of some constituents (for example, CBOD or FC) may be one or more orders of magnitude higher than the selected criterion (assessment endpoint) concentration. For other constituents (such as TP) the maximum discharge concentrations may be only 20 percent higher than the criterion concentration.

Three different relative magnitude ranges are used in the example severity scales presented in Table 3-6 through Table 3-8. CBOD, TSS, and TN are ranked using the same relative magnitude scale (Table 3-6), which has a maximum value greater than 5.0 (that is, the discharge concentration is more than five times higher than the selected criterion concentration). This scale reflects the assumption that, for purposes of the engineering assessment, discharge concentrations of these constituents that are 5, 10, or 25 times the criterion concentration are equally severe for a given event duration. That is, any discharge greater than five times the criterion concentration will probably require immediate corrective actions. The relative magnitude scale for TP (Table 3-7) tops out at greater than 2.0, which reflects the fact that the example effluent concentrations of TP are assumed to have a fairly small range of values (Table 3-7). In contrast, the relative magnitude scale for FC (Table 3-8) ranges from less than 0.1 to greater than or equal to 100, because FC is measured on a log scale.

Note that the example relative magnitude scales are divided into fairly coarse increments. These increments reflect the prediction uncertainties that are assumed to exist for failure events. Users of this framework can make these scales finer or coarser, if they feel this is justifiable for their application. Explicit presentation and discussion of the severity scales will help ensure that the user’s assumptions are evident to decision makers.

The magnitude of an event may change over the course of the event, most commonly by increasing through time. Assigning a single magnitude to such an event adds some degree of error to the assessment process. One could try to reduce this error by dividing an event into multiple stages and assigning a severity level to each stage. However, this process would result in complicated and unwieldy risk assessments and would probably not substantially improve the accuracy of the assessment. That is, such refinements to the prediction of event magnitude are unlikely to be supported by the available science.

One alternative approach is to assign a magnitude that the user feels best represents the event. This approach would require the user to describe the rationale used to select this value and the uncertainties associated with these assumptions. This discussion of uncertainties would need to be carried through to the final risk characterization to ensure that these uncertainties are not overlooked by the decision-makers.

The simplest approach is to assign the maximum expected magnitude to the entire event. This approach will always be conservative, though the degree of conservatism may vary, and is therefore easier to carry through to the final risk characterization. Besides being relatively easy to use, another advantage of this simplifying approach is that it does not exaggerate the user's ability to predict the magnitude of the event as much as would specifying multiple stages and magnitudes of an event.

Although FMEA is designed for evaluating system failures (dysfunction), FMEA can also be used to evaluate OWT system performance under normal operating conditions. Assessment of normal operating performance can be accomplished by specifying a fraction of the assessment endpoint (and an associated event duration) as a level of treatment that warrants attention (but not corrective actions). For example, discharging 8 mg/L of TN (80% of the assessment endpoint concentration or a relative magnitude of 0.8) for seven or more days is the threshold level of performance at which the example emerging OWT system warrants attention (that is, risks appear to be acceptable, but are approaching levels of concern).

Duration

Duration of an event is the second component of severity. Duration is defined as the time elapsed between the start and end of the event, and is not to be confused with the event frequency (number of times an event occurs during a specific period of performance, as discussed previously). The beginning of an event is defined as that point in time at which the OWT system component is no longer achieving the specified level of treatment. For a failure (dysfunction) event, the specified level of treatment is the assessment endpoint (or measure of that endpoint). Thus, for the example emerging OWT system, a failure event with respect to the TN assessment endpoint begins when the discharge concentration exceeds 10 mg/L. The event ends when the TN discharge concentration drops below 10 mg/L, regardless of the cause (for example, system repair or change in environmental conditions).

The example severity scales were based on the assumption that exceeding the effluent criterion generally warranted action of some kind. In the example severity scales presented in Table 3-6 through Table 3-10, any event that lasts less than one day is assumed to pose, at most, a moderate risk, rather than a high or severe risk. This ranking is used for failure (dysfunction) events that are promptly detected and corrected. Including this moderate ranking means that the same severity scale can be used to estimate both the unmitigated and mitigated risks of a particular failure mode. Mitigation measures are assumed to be aimed at detecting and correcting failure events that would otherwise pose a high or severe risk (that is, the unmitigated risks warrant action). Thus, the OWT system gets credit for risk mitigation measures by reducing the severity level of a mitigated failure event to, at most, a marginal risk ranking.

Risk Characterization

Risk characterization is where the values estimated in the quantitative analysis (occurrence frequencies and consequences/severity) are combined to estimate the risk of a particular event. As shown in Figure 3-1, risks are characterized in two general steps. The first risk characterization effort defines the unmitigated risks. If unacceptable risks are estimated in this step, then additional detection and process controls are considered and the risks are re-evaluated and categorized as mitigated risks. These two risk characterization efforts are discussed sequentially in this section.

Unmitigated Risks

Although FMEAs do not incorporate risk matrices, this analysis technique was applied to the wastewater treatment system options because the severity of an event changes with its magnitude and duration. A graphical example of a four-by-four occurrence frequency and consequence (severity) ranking matrix (US DOE 2000) is illustrated in Figure 3-3. The logic behind Figure 3-3 is documented in Table 3-4 and Table 3-5, which describe the frequency of occurrence (OCC) rankings and the consequence/severity (SEV) rankings used in this ranking matrix.

The ranking schemes are designed to separate the lower-risk events that are assessed adequately by the hazard evaluation (FMEA) from high-risk events that may warrant additional analysis if the scenarios involved are not simplistic. A limited number of moderate risk events between the two extremes may also be identified for re-assessment. Example descriptions of likelihood (OCC) and consequence (SEV) thresholds for binning are presented in Table 3-4 and Table 3-5. Risk rankings (unmitigated and mitigated risk characterizations) typically use broader bins (categories) than those used for the OCC and SEV rankings. That is, multiple SEV rankings are combined in each risk-ranking category. For example, SEV rankings of 6, 7, and 8 are all included in the High Consequence (Severity) category in Figure 3-3.

Consequence (Severity)				
Severe (9–10)	II	II	I	I
High (6–8)	IV	III	II	I
Moderate (4–5)	IV	IV	III	II
Slight (1–3)	IV	IV	IV	IV
	Rare (1–3)	**Occasional (4–6)**	**Repeated (7–8)**	**Inevitable (9–10)**
	Frequency of Occurrence			

Figure 3-3
A 4 x 4 Occurrence and Consequence Ranking Matrix for Hazard Evaluation

Figure 3-3 categories are:

- I Major
- II Serious
- III Marginal
- IV Negligible

Note that the ranking of frequency and consequence into such broad categories is more of a qualitative than a quantitative exercise. This effort does not constitute the need for, or expectation of, a probabilistic/quantitative risk assessment.

Mitigated Risks

The second step in the risk characterization process is the estimation of mitigated risks. This step is done only for the failure events for which unmitigated risks were categorized as either Major (I) or Serious (II) (see Figure 3-1). For each of those failure events, the user identifies one or more control processes or detection and correction features that could be used to reduce the frequency of occurrence or the consequence (severity) of the event. Then the user evaluates the ability of the proposed mitigative measures (control processes or detection and correction attributes/mechanisms) to avoid a failure event or to detect a failure event and correct the problem. Finally, the risks for each failure event are re-evaluated based on the assumed effectiveness of the proposed mitigative measures and risk rankings are re-categorized as mitigated risks. For risks that are still categorized as Major (I) or Serious (II), the proposed system or component is generally considered to be unacceptable for the site (that is, environmental setting) being assessed (see Figure 3-1). However, risks that are re-categorized as Marginal (III) or Negligible (IV) would generally be considered to be acceptable (see Figure 3-1). The process of characterizing mitigated risks is discussed in more detail in the following section.

The only difference between the processes of characterizing unmitigated risks and mitigated risks is the consideration of mitigative measures. Mitigative measures typically consist of control processes or detection and correction attributes/mechanisms. Control processes are generally those mitigative measures that are taken to prevent a failure event from occurring (to reduce the frequency of OCC ranking). Example control processes for OWT systems include:

- Mandatory testing or certification for manufactured components
- Inspections/approval of OWT system designs for each site
- Inspection of installation process
- Training/education for installers, inspectors, designers, regulators, and residents

Detection and correction attributes are generally those mitigative measures that help reduce the magnitude or duration of a failure event once it has occurred (to reduce the SEV ranking by reducing the magnitude and/or the duration of the event). Example detection and correction attributes for OWT systems include:

- Visual, audible, physical, and odor clues inherent to the failure event (for example, the sight of untreated wastewater backing up into the residence and the odor of a soil surface breakthrough of untreated wastewater)
- Sensing devices with associated alarms
- Periodic inspections by residents, owners, or third-party inspectors

The effectiveness of each proposed mitigative measure should be evaluated based on a set of clearly specified criteria. Such criteria will help ensure consistency and transparency of the results. Assigning a Detection (DET) ranking is one way to document the assumptions used to evaluate the ability of the proposed mitigative measures to prevent or detect and correct a potential failure event. Example DET rankings and the associated criteria for OWT systems and components are shown in Table 3-11. Note that these criteria and rankings are for example purposes only; the user is expected to develop appropriate criteria for each assessment in consultation with the decision makers.

Table 3-11
Example DET Evaluation Criteria for Design of OWT Systems

Likelihood of Detection	Description	Ranking
Absolute Uncertainty	Design control will not and/or cannot detect a potential cause/ mechanism and subsequent failure mode; or there is no design control	10
Very Remote	Very remote chance the design control will detect potential cause/ mechanism and subsequent failure mode	9
Remote	Remote chance the design control will detect potential cause/mechanism and subsequent failure mode	8
Very Low	Very low chance the design control will detect potential cause/ mechanism and subsequent failure mode	7
Low	Low chance the design control will detect potential cause/mechanism and subsequent failure mode	6
Moderate	Moderate chance the design control will detect potential cause/ mechanism and subsequent failure mode	5
Moderately High	Moderately High chance the design control will detect potential cause/ mechanism and subsequent failure mode	4
High	High chance the design control will detect potential cause/mechanism and subsequent failure mode	3
Very High	Very high chance the design control will detect potential cause/ mechanism and subsequent failure mode	2
Almost Certain	Design control will detect potential cause/mechanism and subsequent failure mode	1

By definition, in the unmitigated event scenario no credit is taken for detection and correction capabilities. For example, in the unmitigated event scenario for an ultraviolet (UV) disinfection unit no credit is taken for automated detection and correction capabilities.

An analysis of the mitigated risks might first assume that the owner or operator inspects and maintains the unit according to a specified schedule. This schedule is not necessarily the manufacturer's recommended maintenance schedule. One could explicitly assume a more realistic schedule based on observations (that is, study results) of similar owners or operators or based on professional judgment. (Note that the assumed schedule should be explicitly included in the assessment to help ensure that the assessment is transparent to the reader and decision maker.) If the assumed maintenance schedule is less than the manufacturer's recommended maintenance schedule, then the assessment might be expected to conclude that the risks of failure exceed the assessment endpoint.

If the risks are unacceptable, a more realistic mitigative scenario could include the UV disinfection unit equipped with a monitoring device that alerts the owner or operator when the unit is dysfunctional. The user must then estimate how long it will take the owner or operator to correct the problem, including the response and repair time, and how much the corrective action will reduce the magnitude of the event. For this example, one might assume that the UV unit is repaired within one day and that the discharge pump was shutdown automatically at the start of the event.

4 PUBLIC HEALTH COMPONENT FRAMEWORK

The public health risk assessment component is used to evaluate potential health risks from exposure to wastewater effluent or environmental media that have come in contact with wastewater effluent. Human health risk paradigms developed by the National Academy of Sciences (NAS 1983), the United States. Environmental Protection Agency (US EPA 1989a) and Kolluru *et al.* (1996) provide the overall risk framework from which this risk assessment component was derived.

A general framework for conducting a public health risk assessment of onsite wastewater treatment (OWT) systems is described in detail along with example calculations. The goal of this framework is to provide quantitative risk estimates for constituents of concern that originate in wastewater effluents of OWT systems.

The steps of this public health risk assessment framework include:

- Problem Formulation
  - Developing a conceptual site model
  - Identifying wastewater chemical and microbial constituents of concern
  - Identifying potential exposure pathways and exposure points
- Analysis
  - Analyzing chemical and microbial data from environmental samples
  - Modeling fate and transport of constituents of concern if environmental sampling data is inadequate
  - Determining concentrations of constituents of concern at exposure points
  - Estimating exposure concentrations and intakes
- Risk Characterization
  - Evaluating assessment endpoints, calculating risks, and defining unacceptable risk levels
  - Characterizing risks from exposure to chemical and microbial constituents of concern that originated in wastewater

These steps are discussed in the following sections.

Problem Formulation

The approach for this public health risk component for evaluating OWT systems involves the following steps:

- Ascertaining the potential for wastewater effluent to be released to the environment
- Assessing migration of wastewater constituents of concern and potential exposure pathways
- Determining exposure concentrations and points of human exposure
- Evaluating assessment endpoints that determine the level of effect and potential for unacceptable risk

The public health risk component focuses on primary constituents of concern (stressors) to humans in wastewater effluent: nitrogen containing compounds (nitrate and nitrite) and microbial pathogens. The two categories of public health assessment endpoints evaluated are based on exposure to chemical and/or microbial constituents of concern.

The human health properties evaluated as the result of exposure to chemicals originating in wastewater are systemic toxicity and/or cancer as defined in US EPA's *Risk Assessment Guidance for Superfund* (US EPA 1989a). For a systemic toxicity (noncarcinogenic) endpoint, the dose for a chemical of concern to an exposed individual is estimated and compared to a human health toxicity value called a Reference Dose (RfD). If the dose of the chemical of concern is greater than the RfD, the potential for adverse health effects (noncarcinogenic) exists. Carcinogenicity is a second human health property evaluated from exposure to chemicals of concern. However, this public health risk framework focuses on nitrate and nitrite, which do not exhibit carcinogenic effects and are not classified as carcinogens. As such, only the noncarcinogenic endpoint will be evaluated and discussed. Additional risk assessment guidance is provided in Appendix D, *Supporting Information: Public Health* for evaluating carcinogenic risk endpoints.

The two human health properties commonly evaluated as a result of exposure to microbes originating in wastewater are infection and illness as defined in microbial risk assessment approaches described by the ILSI Pathogen Risk Assessment Working Group (1996) and Haas *et al.* 1999.

Levels of human health effects from microbial exposure are typically segregated into three categories:

- **No infection**—Meaning the individual was exposed and colonization by the microbe of interest occurred without infection
- **Subclinical disease**—Indicating exposure and infection have occurred, but the infection did not result in illness and medical treatment was not needed
- **Clinical disease**—Indicating that exposure, infection, and illness all have occurred. Illness can be classified as mild, moderate, severe, or death

Due to the wide variability in rates of human infectivity for the numerous microbes and the variability in susceptibility from one individual to another, it is difficult to quantify the dose/response relationship of microbes and humans. Determining correlations between the ingested dose and severity of consequence (duration and/or intensity of symptoms) in a given population is difficult. For some pathogens, the severity of the outcome may depend on the initial dose as well as the health of the exposed individual or population. Because risk of illness can vary greatly with the type and strain of microorganism, as well as host age and other host factors, the microbial assessment endpoint evaluated in this pubic health framework is risk of infection. Risk of infection from microbial pathogens for which dose response information is available will be evaluated, producing a resulting risk value such as 1/10,000 from exposure to a microbial pathogen.

For public health risk assessment purposes, exposed populations are evaluated based on age (children, adults, geriatrics). In addition, sensitive subpopulations maybe evaluated based on gender, ethnicity, baseline health status (immunocompromised, hereditary diseases, and other health factors), or any other site-specific health characteristic of the potentially exposed population that warrants special consideration.

Conceptual Site Model

The first step is to develop a conceptual site model for the public health risk assessment. Conceptual models include a graphic depiction of the model and the associated text that provides a more detailed discussion of the pathways, media, and receptors considered in the assessment. The conceptual site model should describe the:

- Location of residences
- Topography
- Groundwater
- Surface waters
- Potentially exposed populations

Most importantly, the model should describe the location of the OWT system to assist in identification of potential wastewater migration pathways, exposure pathways, and exposed populations. Information on the types of OWT systems and associated engineering risks are contained in Chapter 3 of this integrated risk assessment framework.

Several resources are available to assist in developing a conceptual site model such as:

- Local geographic and demographic information
- Topography
- Land-use permits

- Aerial maps
- Other information from state, local, and community planning authorities

Assimilation of this site-specific information in a conceptual model provides a foundation for starting the public health risk assessment.

The generic conceptual model (Figure 2-5) provides the basis for the development of the conceptual models for the component frameworks. For example, two possible conceptual models were developed for this public health component framework: one depicts the exposure pathways considered for pathogens (Figure 4-1) and the other depicts the exposure pathways considered for nitrates (Figure 4-2). The detailed discussion of the pathways, media, and receptors included in these conceptual models is provided in the subsequent sections of this component framework.

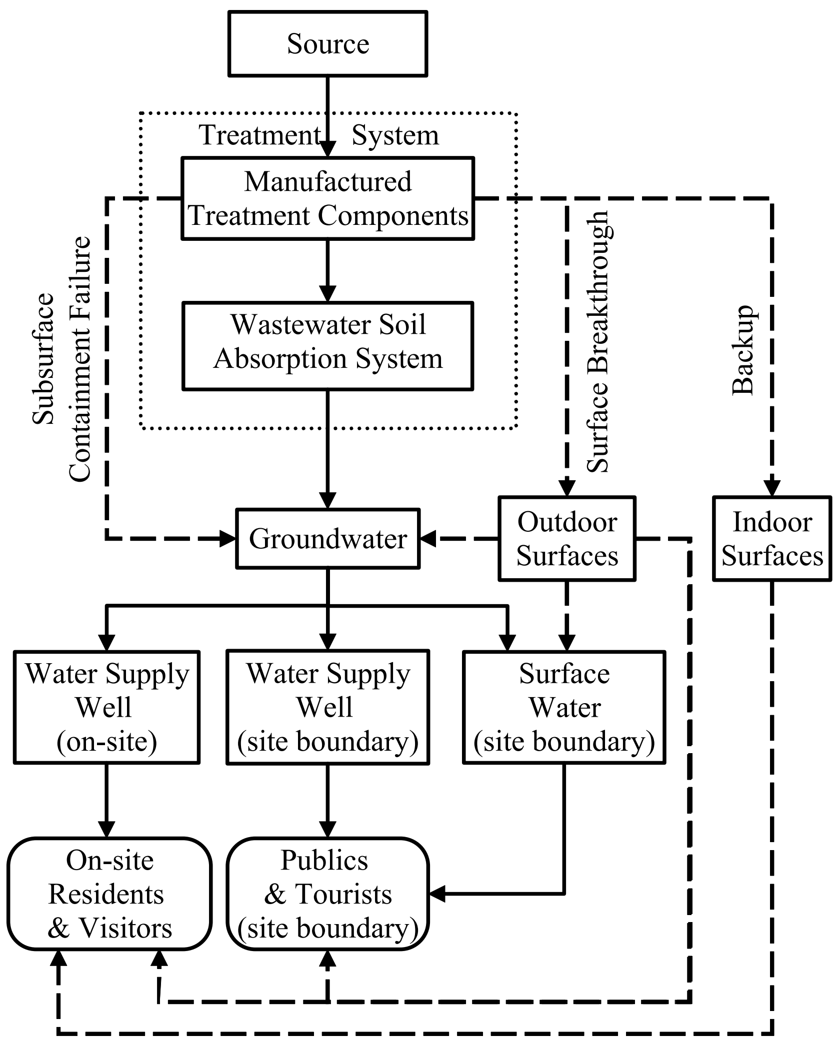

Figure 4-1
Conceptual Site Model for Assessment of Public Health Risks Associated with Pathogens From an OWT System

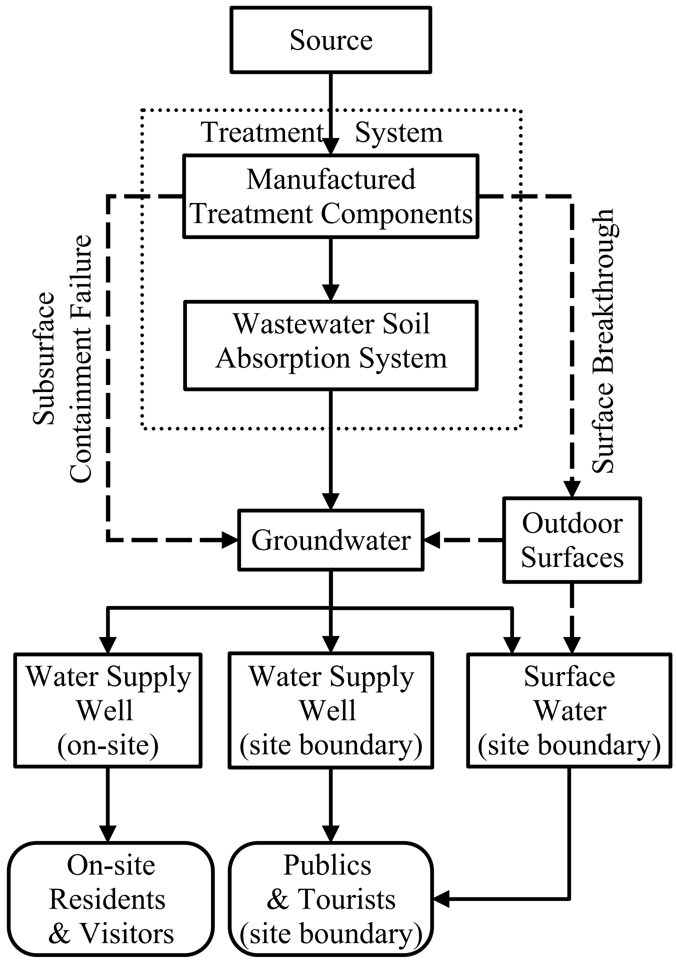

Figure 4-2
Conceptual Site Model for Assessment of Public Health Risks Associated with Nitrates From an OWT System

In conjunction with development of the conceptual site model, the spatial extent and temporal scope of the risk assessment should be defined. The spatial extent of the risk assessment is defined by the boundaries of the site on which the OWT system is located and the migration pathways to potentially exposed persons (such as offsite publics or tourists exposed at the site boundary). Likewise, the assessor should specify the temporal scale of analysis based on the lifetime of the OWT system or components of interest (Chapter 3, *Engineering Component Framework*) or on the travel time of wastewater constituents of concern to a potential exposure point. Local or state regulations may specify timeframes for re-evaluations of OWT system performance, which maybe be guidelines for the temporal scale of the assessment.

Example

As an example of how to use this public health risk component to estimate potential health risks, a scenario will be evaluated based on a single residential OWT system that is functioning properly. OWT systems and expected concentrations of constituents of concern in wastewater effluent during normal system operation as well as system dysfunction are described in Chapter 3, *Engineering Component Framework*. For the public health risk component, these concentrations of constituents of concern represent the initial concentrations that maybe released to the environment and may subsequently impact public health.

For this example, effluent from the residential OWT system has a

- Nitrate-nitrogen concentration of 45 mg/L
- Fecal coliform count of 5×10^5 colony forming unit (cfu) per 100 milliliters (cfu/100 mL)
- Rotavirus concentration of 4×10^3 plaque forming unit per liter (pfu/L)

The effluent is released from a properly functioning wastewater treatment system to the subsurface environment, which consists of soil. The subsurface soil depth extends 10 feet below the effluent discharge point before reaching the water table. Groundwater flow is downgradient and reaches the surface at a perennial spring and surface water body located at the edge of the site boundary.

This example scenario is used to evaluate potential exposure pathways, daily intakes of constituents of concern, and assessment of endpoints in each of the follow sections.

Stressors or Constituents of Concern

Stressors or constituents of concern for public health risk assessment endpoints consist of chemical and microbial agents. Chemicals of most concern with respect to adverse impact to public health in wastewater effluent are the nitrogen containing compounds nitrate and nitrite. Migration of surface or groundwater contaminated with nitrate and subsequent ingestion by humans can produce adverse health effects. Cyanosis among infants who drink well water is a commonly encountered clinical manifestation of nitrate toxicity. Natural geochemical and microbial processes in wastewater treatment systems and the subsurface can either decrease or increase concentrations of nitrogen compounds from onsite wastewater effluent. Thus, these compounds are readily available for environmental transport to potential exposure points such as surface waters and groundwater supply wells.

Microbial pathogens of concern include viruses, bacteria, and protozoans. Detection and monitoring of some of these microbes in environmental samples can be difficult, thus indicators are frequently used whose presence in soil or water denotes the likelihood that pathogens are present. Also, nonpathogenic microbes are monitored as indicator organisms in evaluating the potential presence of pathogens in wastewater treatment systems and effluent. Research continues to improve pathogen detection methods as well as selection of valid indicators for pathogens in environmental samples.

Viruses

Research conducted over the past several decades indicates the primary microbial risk from drinking water supplied by groundwater wells and from recreational exposure to waterborne pathogens is gastroenteritis from viral contamination (Gerba *et al.* 1996 and Soller *et al.* 2003). Gastroenteritis continues as a major cause of mortality worldwide and is the second most common cause of acute illness in families in the US (Monto *et al.* 1983). As noted by Sobsey *et al.* in 1995, recovery and detection of enteric viruses in soil and water is a technological challenge, time-consuming, and expensive. However, recent advances in molecular and genetic analytical techniques for water samples have produced improved monitoring results (see Appendix D, *Supporting Information: Public Health*).

While detection and routine monitoring of viruses in environmental samples is ideal for exposure and risk assessment purposes, the use of other data and models can be a valuable resource for microbial risk assessment. Soller *et al.* (2003) linked limited virus concentration data with numerical simulations (exposure-response models) of enteric viruses and risk of gastroenteritis during recreational exposures to determine if incremental wastewater treatment in a facility would result in substantial reductions in risk to public health. Results from their model-based risk assessment for viruses supported risk management decisions with regard to selection of wastewater treatment approaches. The public health risk assessment for OWT systems presented herein follows a similar approach for assessing public health risks from enteric viruses.

Bacteria

The most commonly measured bacteria to indicate contamination of soil and water by wastewater effluent are in the family Enterobacteriaceace and are commonly referred to as coliforms. Several methods are available for detecting coliforms in environmental samples. One of the preferred analytical water methods is the MI method, which is a membrane-filtration method that allows the simultaneous detection of total coliforms and *Escherichia coli* (*E. coli*) on one medium. Another commonly used method is the enumeration of fecal coliforms, which is indicative of coliforms that live at higher temperatures. Likewise, incubation of MI agar at 41 °C enables detection of other enterococci associated with warm-blooded animals.

Protozoans

Another suite of emerging pathogens of concern in water is the pathogenic protozoans. The two protozoans of most interest are *Cryptosporidium parvum* and *Giardia lamblia*. For pathogenic protozoans, direct measurement of *Cryptosporidium* oocysts and *Giardia* cysts in water samples determine their presence or absence, which is critical for exposure and risk assessment. However, much research is needed in the water monitoring methods to differentiate between live, infectious, and the human strains of these parasites. Approaches for quantitative risk assessments for waterborne pathogenic protozoans have linked illness outbreaks with exposure to drinking water (Haas 2000).

Example Stressors of Concern

The examples in this framework focus on a chemical of concern (nitrate), an indicator of bacterial pathogens (fecal coliforms), and a model virus (rotavirus). Actual risk assessments for OWT systems will typically address multiple chemicals and microbes of concern, such as those identified in this section. Those potential constituents of concern can be evaluated using the information provided in this public health risk assessment framework.

Exposure Pathways and Exposure Points

Another key component in this public health risk framework is the evaluation of exposure pathways and points where humans may come in contact with water or soil containing constituents of concern from wastewater. Exposure pathways generally fall into two categories: onsite and offsite.

Onsite exposure pathways and points are within the residence or the residential lot. For this framework, offsite exposure points are at the site boundary (that is, at the edge of the residential lot) and are downgradient of the OWT system. As constituents of concern in wastewater effluent travel through the environment, several media may be impacted and may subsequently retard or facilitate the transport of constituents of concern to human exposure points. The environmental media commonly evaluated are soil, surface water, and groundwater.

Onsite exposure points may exist within the residence as well as within the residential lot boundary (see Figure 4-1 and Figure 4-2). During normal operation of the OWT system the potential for exposure within the residence is low and would only occur as the result of a system dysfunction, which may cause backup and overflow within the residence. Likewise, treatment system dysfunction can result in several possible exposure pathways within the residential lot. For example, a release of wastewater effluent to surface soil can pollute surface and subsurface soils as well as create runoff, which may impact surface waters. Treatment system dysfunction may also result in direct discharge to surface waters. Treatment system failures that result in subsurface discharges may impact subsurface soils and/or a groundwater supply well via subsurface migration. The most common exposure pathway exists when there is subsurface discharge to the soil and groundwater followed by migration of wastewater constituents of concern to a downgradient groundwater supply well.

Offsite exposure pathways are at the residential lot boundary and primarily downgradient of the OWT system. Both normal OWT system operation and system failures can result in migration of wastewater constituents of concern to exposure points at the site boundary (see Figure 4-1 and Figure 4-2). For example, a release of wastewater effluent to surface soil can pollute surface and subsurface soils as well as create runoff to downgradient receptors at the edge of the residential lot. Treatment system failures that result in subsurface discharges may impact subsurface soils and/or a groundwater supply well via subsurface migration. The most common site boundary exposure pathway consists of subsurface migration of constituents of concern and impacts to subsurface soils, a groundwater supply well, or recharge of groundwater to surface water.

Environmental media typically impacted by wastewater effluent are soil, groundwater, and surface water. These media are also the most frequent human points of contact where exposure occurs. Exposed individuals may have inadvertent dermal contact with soil that contains wastewater constituents of concern. Also, ingestion of groundwater or surface water is a potential human route of exposure as well as dermal contact with surface water while participating in water recreational activities (such as wading or swimming at the site boundary).

Soil

Soil may be contaminated by two transport pathways. Direct discharge of wastewater effluent to surface soils would constitute a potential exposure to high concentrations of constituents of concern, especially pathogens. Subsurface soils maybe impacted by subsurface effluent discharged directly to the soil matrix. Exposure to subsurface soils containing constituents of concern is not likely unless excavation activities occur.

Surface Water

Surface water may be contaminated by several transport pathways. Discharge of groundwater containing constituents of concern would contaminate surface water. Overland flow of wastewater for a short distance before contact with surface water is another potential exposure pathway. Direct discharge of effluent to surface water where exposure can occur is unlikely, but would potentially result in exposure to high concentrations of constituents of concern. As a viable transport medium, surface water can transport constituents of concern great distances or, in contrast, can greatly dilute the concentration of constituents of concern.

Groundwater Supply Well

Subsurface migration of constituents of concern to a groundwater supply well is a commonly studied transport pathway. Hydrologic monitoring or fate and transport modeling in the subsurface provide estimates of concentrations of constituents of concern at groundwater well intakes. Ingestion of groundwater containing wastewater constituents of concern is a potential human exposure route.

Public Health Risk Assessment Endpoints

The two categories of public health assessment endpoints identified for this framework are based on exposure to chemical and microbial wastewater constituents of concern (Table 4-1). Assessment entities include potentially exposed populations and can be evaluated based on several subcategories such as age (children, adults, geriatrics). In addition, sensitive subpopulations may be evaluated based on:

- Gender
- Ethnicity

- Baseline health status (for example, immunocompromised, hereditary diseases)
- Any other site-specific health characteristic of the potentially exposed population that warrants consideration

The exposure pathways define routes of exposure to wastewater effluent or to environmental media that could be contaminated by wastewater effluent. The attribute evaluated quantitatively for chemical exposure is systemic toxicity. The level of effect guideline for chemicals in this public health risk assessment is defined as exceedence of the RfDs for systemic toxicity (US EPA 1989a). The attribute evaluated for microbial exposures is risk of infection. Likewise, the microbial risk of infection guideline is defined as greater than 1×10^{-4} (Macler and Regli 1993).

Table 4-1
Overview of Example Public Health Risk Assessment Endpoints

Entity	Exposure Pathways	Chemical Exposure		Microbial Exposure	
		Attribute	Level of Effect	Attribute	Level of Effect
Onsite resident or visitor	Dermal contact and ingestion of surface and subsurface soil[1]	Systematic Toxicity	Hazard Quotient[3] > 1	Infection	Risk of Infection $> 1 \times 10^{-4}$
Offsite resident or visitor (downgradient)	Groundwater—ingestion Surface Soil—dermal contact and ingestion[2]	Systematic Toxicity	Hazard Quotient[3] > 1	Infection	Risk of Infection $> 1 \times 10^{-4}$
Offsite recreationer (downgradient)	Surface water dermal contact and ingestion	Systematic Toxicity	Hazard Quotient[3] > 1	Infection	Risk of Infection $> 1 \times 10^{-4}$

[1]As a result of OWT system dysfunction, exposure to wastewater may occur inside the residence as well as with surface soil containing wastewater constituents within the residential boundary.

[2]As a result of OWT system dysfunction, exposure to wastewater constituents in surface soil may occur due to runoff migrating outside the residential boundary.

[3]A hazard quotient is the ratio of the exposure concentration divided by the threshold for effects (the RfD). Values greater than 1.0 indicate that the threshold level for effects has been exceeded by the exposure concentration.

Analysis

This step of the risk assessment process focuses on tools and techniques used to measure or estimate exposure and risk. The exposure and effects issues identified during the problem formulation step are evaluated in more detail in the analysis step. To the extent practical, the focus of this step is on quantitative methods.

Environmental Concentrations of Constituents of Concern

In conjunction with the assessment of potential exposure pathways, exposure concentrations for constituents of concern need to be measured or estimated at points of human exposure. Current exposure concentrations for some constituents of concern can be directly measured quantitatively such as the concentration of nitrate in soil, surface water, or groundwater. Likewise, current concentrations of some microbial pathogens can be measured in soil, surface water, or groundwater. Other constituents are more difficult to measure in environmental samples such as viruses. Also, samples may not be available at the specified point of exposure.

When concentrations of constituents of concern cannot be measured at the exposure point, fate and transport models can be utilized to predict exposure concentrations. Likewise, fate and transport modeling can be beneficial to determine half-lives of toxic chemicals as well as survivability and die-off of microbial pathogens. As migration occurs in the environment, certain chemicals degrade and become nontoxic while some become more toxic as they undergo transformation or degradation. Some pathogens rapidly die off while cyst-producing protozoans can survive indefinitely and be transported great distances from where the wastewater effluent initially contacts environmental media. Much research continues to be focused on the transport, survival (die-off), and infectivity potential of pathogens in various environmental settings and conditions.

A primary factor in determining concentrations of wastewater constituents of concern is the assessment of background or naturally occurring concentrations of chemicals and microbes. Risk assessments for OWT units are complicated by the multiple sources of nutrients (nitrogen containing compounds) and microbes that may lead to much greater exposure than the OWT unit of interest. In addition to effluent from OWT systems, environmental inputs of nitrogen include:

- Fertilizer application in agriculture and on lawns
- Animal and livestock waste
- Effluents from industrial and wastewater treatment facilities
- Urban storm water runoff

Likewise, there are numerous microbial sources that may impact soil or water including:

- Pets
- Livestock

- Industrial and wastewater treatment facilities
- Wildlife

Furthermore, the variety and concentration of microorganisms can change in environmental media over time through die-off or re-growth. Seasonal and climatic fluctuations also influence microbial concentrations.

Risk assessors must be aware of additional sources of chemicals and microbes as well as the variety of potential influences on their concentrations when evaluating impacts of OWT leachate to public health. Although these background sources are most relevant at the macro-level (for example, the watershed scale), they may also be important at the micro-level. For example, pets may contribute substantially to the nutrient and microbial loading to onsite soils (with runoff to surface water or recharge of shallow groundwater) and the surface water at the site boundary may already be loaded with nutrients or pathogens from upstream sources. Including these background levels in an assessment of an individual OWT system is appropriate when they are used to define the receiving environment. A detailed evaluation of alternative sources is more appropriate for macro-level assessments.

In the absence of chemical data from sampling and analysis at exposure points, there are robust fate and transport models for estimating and predicting concentrations of chemicals of concern in soil, groundwater, and/or surface water. Site-specific geological, hydrologic, and soil property information is needed to effectively utilize fate and transport models. Several key parameters are inputs for modeling the fate and transport of both chemicals and microbes in the subsurface, including:

- Soil properties (such as porosity, bulk density, soil type, and location of impermeable soil layers)
- Groundwater properties (such as temperature, flow rate and volume of groundwater, amount of suspended solids, and amount of natural organic material)
- Aquifer characteristics
- Other environmental factors (such as rainfall and evapotranspiration)

If site-specific field information is not available, data from similar sites in the region or from public domain databases maybe applicable for modeling purposes.

Subsurface microbial fate and transport modeling research is not as advanced as chemical modeling research. There are a few subsurface microbial transport models, but identifying and measuring key modeling inputs remains a challenge. However, this research area continues to evolve and valuable inputs for microbial fate and transport models are being generated in laboratory and field tests.

Resources for water quality models, fate and transport models, as well as modeling expertise are available in the state and local colleges and universities, state and federal environmental regulatory agencies and regional EPA, USGS, or USDA research offices, including:

- US EPA list of water quality models that can be found at: www.epa.gov/waterscience/wqm/.
- US EPA listing of vadose zone and groundwater models that can be found at: www.epa.gov/ada/csmos.html.
- USGS listing of water models at: http://smig.usgs.gov/SMIC/.

Selection of the appropriate site-specific model is a function of what question is being asked and the amount of data and information available about the OWT system and surrounding area to be modeled.

Example

In this example, a properly functioning OWT system limits onsite residential exposure pathways to subsurface soils and groundwater that come in contact with wastewater containing nitrate and fecal coliforms (see Figure 4-1 and Figure 4-2). For onsite subsurface soils and onsite groundwater, the only potential route of exposure is contact during an excavation event. If it is assumed that excavation will not occur, then no human contact will occur with the subsurface soils or groundwater located within the residential boundary.

Two hypothetical exposure pathways for a properly functioning traditional OWT system are considered. The first is a groundwater supply well located at the site boundary. The other is a surface water body that is recharged by groundwater at the site boundary. Both exposure points are assumed to be 100 feet downgradient of the OWT system discharge point (that is, the wastewater soil absorption system (WSAS) boundary), as per Figure 2-2. The measured concentration of nitrate-nitrogen in the groundwater supply well is 25 mg/L, fecal coliform count is 5×10^3 cfu/100 mL and the rotavirus concentration is 4 pfu/L. Likewise, measured nitrate-nitrogen concentrations in the surface water at the site boundary are 10 mg/L, the fecal coliform count is 4×10^2 cfu/100 mL and the rotavirus concentration is 30 pfu/L.

Also as part of this example, it is assumed the traditional OWT system fails and a dysfunction scenario is evaluated (see Figure 4-1 and Figure 4-2). In this hypothetical scenario wastewater has flowed onto the surface soils within the residential boundary and surface runoff has migrated to the site boundary. The measured concentration of nitrate in onsite residential surface soil is 50 mg/kg, the fecal coliform count is 5×10^5 cfu/g soil and the rotavirus concentration is 4×10^2 pfu/g soil. Likewise, the measured concentration of nitrate-nitrogen in offsite (at site boundary) residential surface soils is 30 mg/kg, fecal coliforms are 2×10^4 cfu/g soil and rotavirus is 40 pfu/g soil.

Exposure Concentrations, Routes of Exposure, and Exposed Populations

For adverse impact to human health to occur, a potentially exposed population is required at the exposure point. Potential receptors include residents and visitors onsite and residents and tourists at the site boundary. Subcategories of receptors include children, adults, geriatrics, and sensitive subpopulations such as immunocompromised individuals and recreationers such as swimmers or hikers.

Routes of exposure for potential receptors include ingestion, dermal contact, and inhalation. For each exposure route, an exposure model is constructed and applied to estimate a daily intake for the wastewater constituent of concern. Daily intakes can be estimated for acute and/or chronic exposures. Also, separate daily intakes can be developed for children and adults due to differences in exposure factors such as body weight, ingestion rates, and exposure frequencies. Daily intakes represent the estimated dose of the constituent of concern. The estimated doses are compared to health toxicity values or guideline values to determine if adverse health impacts are predicted and the level of the effect (RAIS 2003). See http://risk.lsd.ornl.gov/rap_hp.shtml for further information.

Example

To estimate human exposure from contact with soil, groundwater, or surface water containing wastewater constituents of concern, exposure pathway models are utilized to estimate human intakes/doses. Exposure factors and exposure models for estimating daily intakes of nitrate and fecal coliforms from surface soils at the onsite residence and the offsite downgradient residence are presented in Appendix D, *Supporting Information: Public Health*. The OWT is assumed to have failed, wastewater has flowed onto the surface soils within the residential boundary, and runoff has migrated to the site boundary.

The generic exposure factors and models listed in Appendix D, *Supporting Information: Public Health* have been developed by US EPA for assessing human health risk from exposure to hazardous chemicals. As such, the exposure factors are conservative and generally assume exposure for long durations. Site-specific exposure factors are highly recommended to be utilized in exposure modeling and when estimating daily intakes of wastewater constituents of concern.

Using the exposure factors given in Appendix D, *Supporting Information: Public Health* and the previously mentioned hypothetical nitrate-nitrogen, fecal coliform, and rotavirus concentrations measured in the surface soils, the estimated daily intake of nitrate for an onsite adult resident is 6.7×10^{-5} mg/kg-d, fecal coliforms is 685 cfu, and rotavirus is 5.5×10^{-1} pfu. Ingestion of soil by an onsite child resident results in a daily nitrate intake of 6.3×10^{-4} mg/kg-d, fecal coliforms is 6,392 cfu, and rotavirus 5.1 pfu. Likewise, the estimated daily intake of nitrate for an offsite adult resident at the site boundary is 4.1×10^{-5} mg/kg-d, fecal coliforms is 27 cfu, and rotavirus is 5.5×10^{-2} pfu. Ingestion of soil by an offsite child resident at the site boundary results in an estimated daily nitrate intake of 3.8×10^{-4} mg/kg-d, fecal coliforms is 255 cfu, and rotavirus is 0.51 pfu.

The offsite residence also has a groundwater supply well at the site boundary, which supplies water to the household. Exposure factors and models for estimating daily intakes of nitrate and microbes via groundwater ingestion in the adjacent residence where groundwater is the primary water source are presented in Appendix D, *Supporting Information: Public Health*. Using those exposure factors and the previously mentioned hypothetical nitrate-nitrogen, coliform, and rotavirus concentrations measured in the groundwater well, the estimated daily intakes for an adult are 0.68 mg/kg-d, 1,370 cfu, and 0.11 pfu, respectively. A similar groundwater ingestion scenario for a child results in a nitrate concentration of 1.6 mg/kg-d.

Exposure factors and models for estimating daily intakes of nitrate and microbes via surface water as the result of recreational activity (for example, swimming) are also presented in Appendix D, *Supporting Information: Public Health*. Based on the previously mentioned hypothetical nitrate-nitrogen and microbes measured in the surface water, the estimated daily nitrate intake is 0.34 mg/kg-d, fecal coliforms is 1.8 cfu, and rotavirus is 2.6×10^{-3} pfu as the result of recreational activity.

Risk Characterization

The risk characterization step of the risk assessment process evaluates the exposure and effects data developed in the analysis step while producing estimates of risk as well as explanations of results and uncertainties associated with the risk estimates. The risk characterization is used by risk managers and policy makers to determine if actions or policies are needed to reduce or to minimize risks.

Calculating Risk Estimates for Chemical and Microbial Exposures

As discussed previously, the public health risk assessment is used to evaluate exposure to chemical and microbial wastewater constituents. The public health endpoint evaluated for chemical exposures is systemic toxicity and the endpoint for microbial exposures is risk of infection (see Table 4-1). As previously described, if the calculated chemical intake exceeds the chemical-specific RfD, adverse health effects maybe expected (that is, the hazard quotient is greater than 1.0).

Quantification of microbial risk from exposure to wastewater effluent or environmental media that has contacted wastewater effluent is more challenging. Humans are exposed to microbes by numerous mechanisms such as through the air, by foods, and via hand contact for example. Thus, it is difficult to ascertain what percentage of the total microbial dose is attributed to exposure to microbes originating in wastewater effluent. Likewise, there are differing numbers of organisms in soil, air, or water and differing amounts of media consumption per individual. These factors contribute to variable doses to individuals. Another confounding parameter to quantification of microbial risk is the ability of infectious agents to propagate within a susceptible host with resulting signs of pathogenicity. There may be a latency period before the microbial population achieves significant numbers to cause infection and illness, which is also influenced by the health of the host.

For this public health risk assessment framework, the microbial assessment endpoint evaluated is risk or probability of infection. The use of probability of infection models for development of standards for bacteria, viruses, and protozoa in water is the basis for establishing the surface water treatment rule to address performance-based standards for control of *Giardia* (US EPA 1989b). To quantify the risk of infection, estimates of microbial doses and dose/response data for microbial species of concern is required. Dose/response data are available for several pathogenic microbial species associated with wastewater such as poliovirus, rotavirus, adenovirus, *E. coli*, *Salmonella*, *Shigella, Cryptosporidium parvum*, and *Giardia lamblia* (Haas *et al.* 1999 and WHO 2001).

To assess risks of infection from microbial exposures, the measured fecal coliforms in environmental samples were assumed to be *E. coli* to estimate the risk of infection. While most *E. coli* are not pathogenic, the presence of *E. coli* suggests the potential presence of pathogenic strains. To estimate the risk of microbial infection from ingestion and/or contact with soil, groundwater or surface water, the following beta-Poisson dose-response model, described in Haas *et al.* 1999, was used.

$$P = 1 - (1+(N/\beta))^{-\alpha} \qquad \textbf{Equation 4-1}$$

In this model, P is the probability of infection, N is the microbial exposure, and α and β are values defined by the dose response curve specific to an individual microorganism. In this case, the dose response curve for *E. coli* indicates $\alpha = 0.1705$ and $\beta = 1.61 \times 10^6$ (Pepper *et al.* 1996). Using the microbial intakes (doses) from the exposure models as input for the beta-Poisson model, risks of *E. coli* infection were estimated for the example exposure pathways.

The beta-Poisson model was also used to estimate risk of rotavirus infection. Measured rotavirus concentrations were used to determine microbial intakes (doses) from the exposure models. Based on the estimated dose and the beta-Poisson model, risks of rotavirus infection were estimated for the example exposure pathways. Haas *et al.* (1993) estimated the rotavirus dose-response parameters of α to be 0.26 and β to be 0.42.

The risk of infection estimates should be performed for each microbial pathogen of concern following the *E. coli* and rotavirus examples described previously. However, dose/response data for wastewater pathogens is limited to several bacterial species, one or two protozoans, and select viruses that also possess the potential for infecting exposed individuals. As microbial dose/response research continues to define infectious and pathogenic relationships, this information should be incorporated into quantitative microbial risk assessments, thus providing a cumulative risk estimate from exposure to multiple microorganisms.

Example Risk Characterization

This section provides example risk characterizations for an onsite resident or visitor and an offsite resident or visitor at the site boundary.

Onsite Resident or Visitor

Continuing the example risk calculations, the estimated onsite residential daily intake for the adult onsite resident or visitor of nitrate in soil is 6.7×10^{-5} mg/kg-d based on exposure factors and exposure models defined in Appendix D, *Supporting Information: Public Health*. To determine potential for systemic toxicity, the daily intake is compared to the RfD for nitrate, which is 1.6 mg/kg-d (US EPA 2002 and RAIS 2003). If the daily intake is greater than the RfD, systemic toxicity may occur. In this example, the daily intake of 6.7×10^{-5} mg/kg-d is less than the RfD and systemic toxicity is not expected from exposure to onsite soil containing nitrate (see Table 4-2).

In addition to the adult scenario, a child soil ingestion scenario is evaluated to demonstrate risk estimates for a sensitive subpopulation. The estimated onsite child resident intake of nitrate in soil is 6.3×10^{-4} mg/kg-d based on the child exposure factors and models defined in Appendix D, *Supporting Information: Public Health*. The child nitrate intake is less than the RfD indicating systemic toxicity is not expected from exposure to soil containing nitrate (Table 4-2).

Table 4-2
Example of Chemical Risk Characterization Summary for OWT Public Health Risk Assessment

Receptors and Exposure Pathways	Wastewater Chemicals of Concern		
	Nitrate-N		
	Measured Concentration	Intake (Dose)	Hazard Quotient[1]
Onsite Resident/Visitor			
Soil (Adult)	50 mg/kg	6.7×10^{-5} mg/kg-d	4.3×10^{-5}
Soil (Child)	50 mg/kg	6.3×10^{-4} mg/kg-d	4.0×10^{-4}
Offsite Resident/Visitor			
Soil (Adult)	30 mg/kg	4.1×10^{-5} mg/kg-d	2.5×10^{-5}
Soil (Child)	30 mg/kg	3.8×10^{-4} mg/kg-d	2.4×10^{-4}
Groundwater			
(Adult)	25 mg/L	0.68 mg/kg-d	0.42
(Child)	25 mg/L	1.6 mg/kg-d	1.0
Offsite Recreationer			
Surface Water	10 mg/L	0.34 mg/kg-d	0.21

[1]The hazard quotient is the intake/reference dose (RfD). The nitrate RfD is 1.6 mg/kg-d.

For the microbial risk endpoint, the fecal coliform intake estimated for the adult onsite residential ingestion of soil scenario is 685 cfu and the rotavirus intake is 0.55 pfu (Table 4-3). Using the beta-Poisson model for *E. coli* and rotavirus, the estimated risk of infection is 7.2×10^{-5} for *E. coli* and 1.9×10^{-1} for rotavirus (see Appendix D, *Supporting Information:*

Public Health for a breakdown of the example calculations). The rotavirus estimate exceeds the guideline risk of infection level of 1.0×10^{-4} (Table 4-3).

The fecal coliform intake estimate for the child onsite residential ingestion of soil scenario is 6,392 cfu, which yields a risk of 6.7×10^{-4}. Likewise, the rotavirus risk of infection estimate for child from soil ingestion is 4.8×10^{-1}. Both risk of infection estimates exceed the guideline risk level of 1.0×10^{-4}.

Table 4-3
Example of Microbial Risk Characterization Summary for OWT Public Health Risk Assessment

Receptors and Exposure Pathways	Wastewater Microbials of Concern					
	Rotavirus			Fecal Coliforms		
	Measured Conc.[1]	Intake (Dose)	Risk of Infection	Measured Conc.[2]	Intake (Dose)	Risk of Infection
Onsite Resident/ Visitor						
Soil (Adult)	4×10^{2} pfu/g	0.55 pfu	1.9×10^{-1}	5×10^{5} cfu/g	685 cfu	7.2×10^{-5}
Soil (Child)	4×10^{-2} pfu/g	5.1 pfu	4.8×10^{-1}	5×10^{5} cfu/g	6,392 cfu	6.7×10^{-4}
Offsite Resident/ Visitor						
Soil (Adult)	40 pfu/g	5.5×10^{-2} pfu	3.1×10^{-2}	2×10^{4} cfu/g	27 cfu	3.0×10^{-6}
Soil (Child)	40 pfu/g	0.51 pfu	1.8×10^{-1}	2×10^{4} cfu/g	255 cfu	2.7×10^{-5}
Groundwater	4 pfu/g	1.1×10^{-1} pfu	1.8×10^{-1}	5×10^{3} cfu/ 100 mL	1,370 cfu	1.2×10^{-4}
Offsite Recreationer						
Surface Water	30 pfu/L	2.6×10^{-3} pfu	1.6×10^{-3}	2×10^{3} cfu/ 100 mL	1.8 cfu	1.0×10^{-6}

[1] Rotavirus concentrations are reported as plaque forming units (pfu).

[2] For this public health risk assessment example, measured fecal coliforms are assumed to be *E. coli.* Fecal coliform concentrations are reported as colony forming units (cfu).

Offsite Resident or Visitor at the Site Boundary

The soil ingestion exposure pathway was evaluated for an offsite adult and child resident or visitor at the site boundary. The intake of nitrate in soil by an adult resident or visitor is 4.5×10^{-5} mg/kg-d and by a child is 3.8×10^{-4} mg/kg-d. Both the estimated adult and child intakes are less than nitrate RfD (Table 4-2).

For the microbial risk endpoint, the fecal coliform intake estimated for the offsite residential adult soil scenario is 27 cfu, which results in a risk of infection estimate of 3.0×10^{-6}, which does not exceed the risk guideline of 1.0×10^{-4} (Table 4-3). The rotavirus risk estimate is 3.1×10^{-2}

for the adult and 1.8×10^{-1} for the child, both of which exceed the guideline value. The fecal coliform intake estimate for the child offsite residential ingestion of soil scenario is 255 cfu, which yields a risk of 2.7×10^{-5}.

Also, the offsite residential groundwater at the site boundary was assessed for chemical and microbial risks. The exposure pathway evaluated is ingestion of groundwater, which results in a nitrate intake by the adult resident of 0.68 mg/kg-d based on exposure factors and models defined in Appendix D, *Supporting Information: Public Health*. The resulting nitrate intake is less than the RfD. However, ingestion of groundwater by a child results in an intake of 1.6 mg/kg-d, which is equal to the RfD. The intake of fecal coliforms via groundwater ingestion by the adult resident at the site boundary is 1,370 cfu, which yields a risk of 1.2×10^{-4}. Similarly, the rotavirus estimate from groundwater ingestion is 1.8×10^{-1}, which indicates both the fecal coliform and rotavirus risk of infection estimates from groundwater ingestion are greater than the guideline risk level of 1.0×10^{-4}.

Offsite Recreationers at the Site Boundary

A recreational scenario was evaluated based on migration of groundwater downgradient with discharge to surface water at the site boundary; in this example, a surface water body that is used for swimming. Recreational exposure factors for water ingestion and dermal contact while swimming are defined in Appendix D, *Supporting Information: Public Health*. For the recreational exposure example, the daily intake of nitrate from incidental ingestion while swimming and via dermal contact would be 0.34 mg/kg-d (Table 4-2). When the daily intake is compared to the nitrate RfD, it is less than the RfD.

Recreational exposure while swimming results in an intake of 1.8 cfu fecal coliforms and 2.6×10^{-3} pfu rotavirus. The risk of infection is 1.0×10^{-6} for fecal coliforms and 1.6×10^{-3} for rotavirus, which exceeds the guideline of 1.0×10^{-4} (Table 4-3).

Public Health Risk Summary

For this public health risk assessment component, risks are characterized for potential exposure to chemical and microbial constituents of concern in wastewater effluent. For adverse impact to public health to occur, human exposure and intake of constituents of concern must also occur.

The risk summary describes estimated risks for completed exposure pathways and identified potentially exposed populations, risks to sensitive populations if present, and uncertainties associated with risk estimates. Quantitative risks to public health are described in this framework for onsite and offsite residents and visitors as well as offsite recreationers.

For onsite and offsite residents as well as offsite recreationers exposed to nitrate or other chemicals of concern, systemic toxicity endpoints are evaluated. For chemicals with the potential to cause systemic toxicity (that is, cyanosis), exposures greater than the reference dose (RfD) are a health concern. Exposure to microbes that results in a risk of infection greater than 1×10^{-4} exceeds the example guideline for this public health risk framework. For microbial risks greater than 1×10^{-4} or exceedence of RfDs, risk management preventative measures may be warranted to reduce the concentration of wastewater constituents of concern at the source, reduce migration of wastewater constituents of concern, or mitigate public exposure to wastewater constituents of concern (Kolluru *et al.* 1996).

Quantitative Assessment

In the example risk calculations for chemical exposures, evaluations of nitrate in onsite and offsite soils at the site boundary for adult and child scenarios do not exceed the nitrate RfD guideline value (Table 4-2). Evaluation of nitrate in groundwater at the site boundary for the adult in the downgradient residence did not result in an exposure greater than the guideline. The primary public health concern from exposure to nitrate in groundwater is adverse effects in children. Based on measured nitrate concentrations in the groundwater, the estimated intake for a child is equal to the RfD guideline indicating the need for risk management to mitigate potential adverse health impacts. Because carcinogenic effects from nitrate have not been documented in dose/response toxicity studies and the US EPA does not identify nitrate as a carcinogen (RAIS 2003), carcinogenic risks from nitrate exposure were not evaluated. (See Appendix D, *Supporting Information: Public Health* for a brief discussion of carcinogenic chemical risk assessments.)

For the onsite and offsite residents at the site boundary as well as recreationers exposed to microbes at the site boundary that originate from wastewater effluent, the rate of infection is the endpoint evaluated. Risks of infection greater than 1×10^{-4} exceed the example guideline for this public health risk framework. In the example risk calculations for microbial exposures, infection risks from exposure to fecal coliforms and rotavirus were evaluated for onsite and offsite soils at the site boundary for the adult and child scenarios. An assumption that the measured concentrations of fecal coliforms were *E. coli* was made to provide a conservative, quantitative assessment of infection risks. Based on this assumption, adult exposures to onsite soils containing fecal coliforms did not exceed the risk of infection guideline. However, ingestion by a child of onsite soil containing fecal coliforms as a result of OWT dysfunction indicates a risk of infection of 6.7×10^{-4}, which is greater than the guideline (Table 4-3).

Adult and child exposures to soils that have been contaminated with fecal coliforms at the site boundary as a result of OWT dysfunction do not result in exceedence of the example risk of infection guideline. However, the groundwater ingestion scenario for an offsite resident at the site boundary indicates a risk of infection of 1.4×10^{-4} from exposure to fecal coliforms. Because this risk assessment was based on the assumption that the measured fecal coliforms were *E. coli*, the risk estimates are conservative and most likely overestimated, which is a key consideration for risk managers.

The adult and child estimates for exposure to rotavirus in onsite and offsite soils at the site boundary resulted in risk estimates greater than the guideline of 1×10^{-4} (Table 4-3). In addition, the groundwater ingestion and surface water recreation scenarios for rotavirus exposure also indicate risks that exceed the guideline value. The primary reason for these exceedances in the risk estimates is the high rate of infectivity associated with rotavirus.

Based on human dose response data, it appears that rotavirus is the most infectious enteric virus; thus, high rates of infectivity are expected (Regli *et al.* 1991, Gerba *et al.* 1996, Haas *et al.* 1999). Furthermore, the use of rotavirus as a model for enteric viruses, in this framework as well as by other researchers, provides a conservative estimate of risks from OWT systems to onsite and offsite residents and recreationers at the site boundary. According to Gerba *et al.* (1996), rotavirus has unique characteristics that make it a significant agent of human illness and a model viral pathogen to assess waterborne illness from wastewater effluent. Some of these characteristics are:

- Most common cause of viral gastroenteritis worldwide
- Highest infectivity of any waterborne virus
- Highest known case fatality for viral gastroenteritis
- 100-fold greater case fatality rates in the elderly
- Greater case fatality rates in the immunocompromised (as high as 50%) compared to the general population (0.01%)
- Greatest resistance to inactivation by UV light disinfection of all enteric RNA viruses

The use of rotavirus as a conservative model virus is a key consideration for risk managers when discussing policies or approaches for protection of public health from wastewater constituents of concern.

Qualitative Assessment

In addition to the quantitative risk estimates, this public health risk assessment includes a qualitative evaluation of risks from shellfish. The public health risk goal is to prevent harvesting of shellfish that contain elevated numbers of microbial pathogens (bacteria and viruses) thus reducing the likelihood of infection in consumers of shellfish.

For this public health risk assessment (which addresses a single OWT) the concentration of microbial pathogens released in wastewater from one OWT is unlikely to detrimentally impact the water quality of shellfish. The microbial load would be greatly reduced due to the dilution that would occur in the waters inhabited by the shellfish. However, the shellfish endpoint should be evaluated at the macro-level when multiple OWT systems or a wastewater treatment facility is evaluated, especially if they are located within a watershed that recharges or feeds surface waters inhabited by shellfish. The water quality of shellfish-inhabited waters is typically determined by sampling and analysis for fecal coliforms or by quantitative risk assessment based on shellfish consumption (Rose and Sobsey 1993 and Lee and Younger 2002). A concentration of fecal coliforms greater than a few tens to a few hundreds cfu/100 mL in waters of shellfish beds is generally unacceptable. When this criterion is exceeded, the shellfish bed is closed for harvest until the concentration of fecal coliforms does not exceed the guideline. State regulatory agencies typically determine shellfish bed closure criteria.

Uncertainty in Quantitative Risk Estimates

The quantitative risk assessment approach presented in this public health risk framework contains conservative assumptions in the exposure models and in development of toxicity values for chemicals and microbial constituents of concern. Conservative assumptions are intended to provide a margin of safety due to the uncertainty associated in the estimates of risk to the public. Because precise information is not know about all exposure parameters such as the amount of groundwater ingested, exposure durations, or the amount of time recreationers spend in the water while swimming, best estimates and conservative assumptions are made during the risk assessment process (US EPA 1995). Also, there is uncertainty in the human toxicity data as well as the chemical and microbial dose response models used to estimate health effects. The use of conservative assumptions tends to overestimate the risk to the general public while providing a margin of safety for individuals more susceptible to adverse health effects. If time and resources permit, a quantitative uncertainty analysis of the parameters and models used to estimate risk may provide a better understanding of technical issues associated with the risk estimates. Further refinement of parameters and models that are the most uncertain will reduce the overall uncertainty in the risk estimates.

In addition, if the public health risk assessment indicates the potential for risks from exposure to wastewater constituents of concern, a quantitative uncertainty evaluation maybe conducted to better refine the risk estimate. A sensitivity analysis of parameters used in the risk estimate will reveal which parameters most greatly influence the risk estimate. Refining the most sensitive parameters will reduce the uncertainty in the risk estimate. Likewise, collecting additional data for parameters utilized in estimating exposure and risk generally reduces the uncertainty in the risk estimate. A tutorial for quantitative uncertainty analysis for chemical risk assessment is provided in *Guiding Principles for Monte Carlo Analysis* (US EPA 1997) and for microbial risk assessment in *Quantitative Microbial Risk Assessment* (Haas *et al.* 1999).

Other Risk Metrics and Applicable Guidelines

The question of acceptable risk is determined by site-specific conditions and local or federal applicable regulations for OWT systems. However, reviewing other water quality risk levels and/or concentration-based guidelines provides a basis for consistency in risk assessment approaches. For example, microbial risk levels have been promulgated by US EPA or state regulatory agencies for water used for recreational purposes and drinking water (US EPA 1999 and US EPA 2002). Reducing the risk of pathogens in drinking water is paramount for US EPA. The US EPA approach for microbes in drinking water is treatment technology dependent to achieve 99 to 99.99 percent removal/inactivation of the specified microbe because of the need to protect potentially exposed persons with undeveloped immune systems and sensitive subpopulations with impaired immunity. Thus, the drinking water Maximum Contaminant Level Goal (MCLG) is technology dependent as long as greater than 99 percent removal efficiency is achieved (US EPA 2002 and US EPA 2003a). The objective of the reduction requirements is to reduce risk of infection to 1×10^{-4} per year from the consumption of tap water.

In addition to microbial guidelines for drinking water, most state regulatory agencies have specified microbial guidelines for surface waters used for recreational purposes. Each state specifies a monitoring strategy and microbial limits based on several factors such as climatology, hydrologic conditions, types and frequency of use of the surface water, and incidence of illness from contact with surface waters. As such, the microbial concentration limits vary from state to state. When comparing state regulatory guidelines that specify microbial concentration limits in surface water, the most commonly used metric for assessing microbial water quality is fecal coliforms. Guidelines for fecal coliform counts in surface waters that come in contact with the public range from 100 to 1000 cfu/100 mL with 100 to 200 cfu/100 mL the most commonly specified values (US EPA 2003a). State recreational microbial water quality guidelines for recreational waters can be found in US EPA (2003b).

Some states also specify guidelines for other indicator organisms such as enterococci and *E. coli*. Guidelines for enterococci counts in surface waters range from 8 to 50 cfu/100 mL and *E. coli* from 50 to 200 cfu/100 mL. When these state recreational water quality criteria are exceeded, public access is restricted until future sampling indicates a decrease in the number of fecal coliforms, enterococci, and/or *E. coli*. Note that the US EPA has targeted the year 2003 when states are to start using enterococci and *E. coli* as the primary indicator organisms instead of total coliforms or fecal coliforms (US EPA 1999).

5 ECOLOGICAL COMPONENT FRAMEWORK

The ecological component framework is used to evaluate the potential adverse impacts on non-human biota and ecosystems. The three-stage risk assessment format used throughout this framework (problem formulation, analysis, and risk characterization) is also used in this component framework. Each aspect of the framework is described in this chapter, and additional supporting information, especially that which is relevant to the macro-scale of assessment, is detailed in Appendix E, *Supporting Information: Ecological*.

Problem Formulation

The generic problem formulation for the integrated risk assessment was presented in Chapter 2, *General Problem Formulation*. The problem formulation for the ecological risk assessment component includes the development of the conceptual model, the selection of assessment endpoints, and the development of an analysis plan. Without this planning phase, the ecological risk assessment may be unfocused and may not meet the goals of the decision makers (risk managers).

Assessment and Management Goals

The overall assessment and management goals are discussed in Chapter 2, *General Problem Formulation*. The implications of those goals for the ecological assessment are best highlighted in the problem formulation of this component framework. That is, the risk assessment should be planned to address the overall assessment and management goals. For example:

- The risk assessment may be retrospective (concerned with impacts of past nutrient releases), prospective (concerned with prediction), or concerned with current conditions.
- The goal may be to determine if adverse effects are likely, or to determine the likely magnitude of the adverse effects.
- The assessor should specify whether a broad range of ecological effects are of concern, or only those that relate to existing water quality criteria.
- In the context of wastewater treatment, the assessment may be comparative, that is, intended to compare risks associated with different treatment technologies, or not comparative. In non-comparative assessments, the goal may be to identify any potential for risk, and therefore, it may be appropriate to use a high estimate of exposure and a high estimate of the effects that are potentially caused by that exposure. In comparative assessments, it is better to make accurate estimates of exposures and effects, or the ranking of alternatives may be biased.

Each of these example assessment and management goals could be supported by use of this framework.

Spatial and Temporal Bounds

As stated in the general problem formulation, this framework is designed to addresses micro-scale OWT systems, that is, an individual residential lot with an OWT system. Although some ecological impacts will only be evident at the macro-scale (such as population-level impacts on wide-ranging, mobile species), others can potentially be manifest at smaller geographic scales (such as impacts on plants and sensitive receptors). This circumstance is particularly true for sites where the dilution of OWT effluent at the exposure point is not great.

For example, the user may be interested in the potential risks to frogs inhabiting a shoreline were potentially contaminated groundwater is discharged to a small pond (at the site boundary, as shown in Figure 2-2). This scenario is particularly relevant, because several species of frogs appear to be sensitive to nitrate concentrations in water and there is increasing public concern regarding the widely observed decline in amphibian populations (note that these general declines are not necessarily related to nitrate concentrations). Although the potential impacts to amphibians in this example may be especially localized, they could still be of concern to the decision makers (such as, owner, residents, neighbors, and public officials).

The micro-scale, which is addressed in this framework, is pertinent to residential treatment systems located adjacent to small ponds, streams, or lagoons and some parts of shallow estuaries (such as coves where tidal water exchange is exceptionally limited). In the case of amphibians, it is also relevant for sites with small or temporary ditches onsite or at the site boundary. Users of this framework should specify the spatial bounds of the assessment to include the areas of adjoining surface waters that could potentially be contaminated by the OWT system being evaluated. This spatial extent is consistent with recommendations in Suter *et al.* (2000) for specifying the spatial extent for an ecological risk assessment to include areas that are believed to be contaminated and to extend to the point where sufficient dilution volume ensures negligible risks. Thus, the spatial bounds for the ecological assessment may extend somewhat beyond the site boundary, but it is still a micro-level assessment because only one OWT system is being evaluated.

Although the exposure models and exposure-response models presented here could also be used for macro-level assessments, the localized scale of analysis is the primary justification for the selection of stressors and assessment endpoints emphasized in this component framework. Most potential effects at the local scale are direct effects (such as increased plant biomass), rather than secondary effects (such as those from losses of forage or habitat or from increases in predation). For example, losses of abundance or production of eelgrass are addressed, but relationships between eelgrass production (or area) and abundance of the many populations that depend on eelgrass are not considered in this micro-level framework.

In addition to the spatial scope, the assessor should specify the time scale of analysis. The temporal bounds may be based on several factors, including:

- The lifetime of a treatment system
- The lifespan of a particular receptor
- Regulatory requirements
- A decision by a risk manager (decision maker)

Particular events in time may be emphasized, such as storm events that cause treatment failure. Periods of high releases of nutrients that coincide with sensitive life stages of organisms should be considered. For example, high inputs of nutrients may occur in the spring, during thaws of frozen soils and fertilizer runoff. This time period coincides with periods of vertebrate breeding or larval development (such as amphibians as documented in Hecnar 1995).

Many effects that could result from multiple residential lots with OWT systems are described in Appendix E, *Supporting Information: Ecological*, because they are beyond the scope of the micro-scale focus of this chapter.

Stressors

Stressors are agents, such as chemical constituents, that adversely affect ecological receptors. The same agent that is beneficial for one ecological receptor may act as a stressor with respect to another. Nutrients in surface water may be viewed as ecological stressors if they are directly toxic to aquatic organisms (such as nitrate for frogs) or if they reduce the productivity of one species by increasing the productivity of another species (for example, excessive growth of algae attached to macrophytes can “shade” out the macrophytes).

The nutrients nitrogen and phosphorus are the principal ecological stressors associated with residential OWT systems. Nutrient inputs to a surface water body have the greatest impact if background concentrations limit production or growth rates (such as primary production). In general, nitrogen is a limiting nutrient in estuarine waters in temperate environments and phosphorus is a limiting nutrient in most fresh waters in temperate environments. However, the user must identify stressors based on site-specific conditions. For example, phosphorus was identified as the most limiting nutrient in part of the Indian River Lagoon in Florida (Phlips *et al.* 2002). Detailed information that may be useful for identifying site-specific stressors is provided in Appendix E, *Supporting Information: Ecological*. However, the following points are worth noting here:

- Nitrogen is expected to enter surface water as nitrate, because oxidation of ammonia, nitrite, and organic forms of nitrogen usually occurs rapidly and nitrate is the most stable form of nitrogen in surface waters.
- Most of the phosphorus released in wastewater effluent (about 85 percent) is in the bioavailable form of soluble orthophosphate (Gold and Sims 2001).

- A substantial fraction of the phosphorus in wastewater effluent may precipitate with aluminum, iron, or calcium or sorb to clay particles, especially if it travels a substantial distance through soil.
- Organic matter in wastewater effluent is another ecological stressor addressed in this framework, because it can cause localized ecological impacts by reducing the amount of dissolved oxygen in surface waters.
- Other chemical stressors (such as household chemicals and antibiotic pharmaceuticals) may be constituents of OWT system effluent, but are not emphasized in this framework.

The user must identify the stressors that are the focus of each particular risk assessment and provide a rationale for why some potential constituents of the OWT system effluent are considered in the assessment and others are not.

Background Levels of Stressors

As stated previously, the risk assessor must define the background or reference conditions for the risk assessment. Exposures and risks are calculated with respect to background levels, which may be nutrient concentrations in the receiving water in the absence of the OWT system that is being evaluated or concentrations at reference locations that have no anthropogenic sources of nutrients.

In many ecological risk assessments for anthropogenic chemicals (such as PCBs), the source of concern is typically the major source of the chemical to surface water. However, risk assessments for OWT systems are complicated by the multiple sources of nutrients that may lead to much greater nutrient concentrations in the receiving water than the OWT system of interest. The user should be aware of other sources of nutrients and how they may affect the outcome of the risk assessment.

Ecosystems

Two types of surface water ecosystems are distinguished in this risk assessment framework:

- Freshwater systems (such as ponds)
- Estuarine systems (such as coastal lagoons)

The dichotomy in this framework between salt and freshwater systems is based on differences in prevailing nutrient dynamics (such as the nutrient limitations described previously). In the characterization of exposure, differences between lotic (flowing) and lentic (still) waters also are noted.

The ecological risk assessor should describe the receiving environment of interest in any ecological risk assessment. Many of the ecosystem components will be identified in the conceptual model, but not all will be considered in detail in every assessment.

Conceptual Models

As stated in the generic problem formulation (Chapter 2, *General Problem Formulation*), the conceptual model is used to describe and visually depict the expected relationships among stressors, exposure pathways, and receptors. Conceptual models are working hypotheses about how a chemical or other stressor may affect assessment endpoint entities (Suter *et al.* 2000). The assessor should depict all routes of exposure that are of potential concern. The models presented here are generic for assessments of OWT systems (Figure 5-1 and Figure 5-2). The conceptual models provide an aggregate picture of potential effects, but some of the assessment endpoint entities (such as mangroves and manatees) have exceptionally limited geographic ranges. The user should tailor a conceptual model to a site with more specific ecological receptors and stressors after surveying that site and studying its ecology. All ecological properties depicted in these models are candidate assessment endpoints for risk assessments of OWT systems. The endpoint properties that are emphasized in this risk assessment framework for individual OWT systems are indicated on the figures.

Note that in Figure 5-1 and Figure 5-2 potential assessment endpoint entities and properties are denoted by rectangles. Exposure pathways of focal assessment endpoint properties are indicated with bold lines.

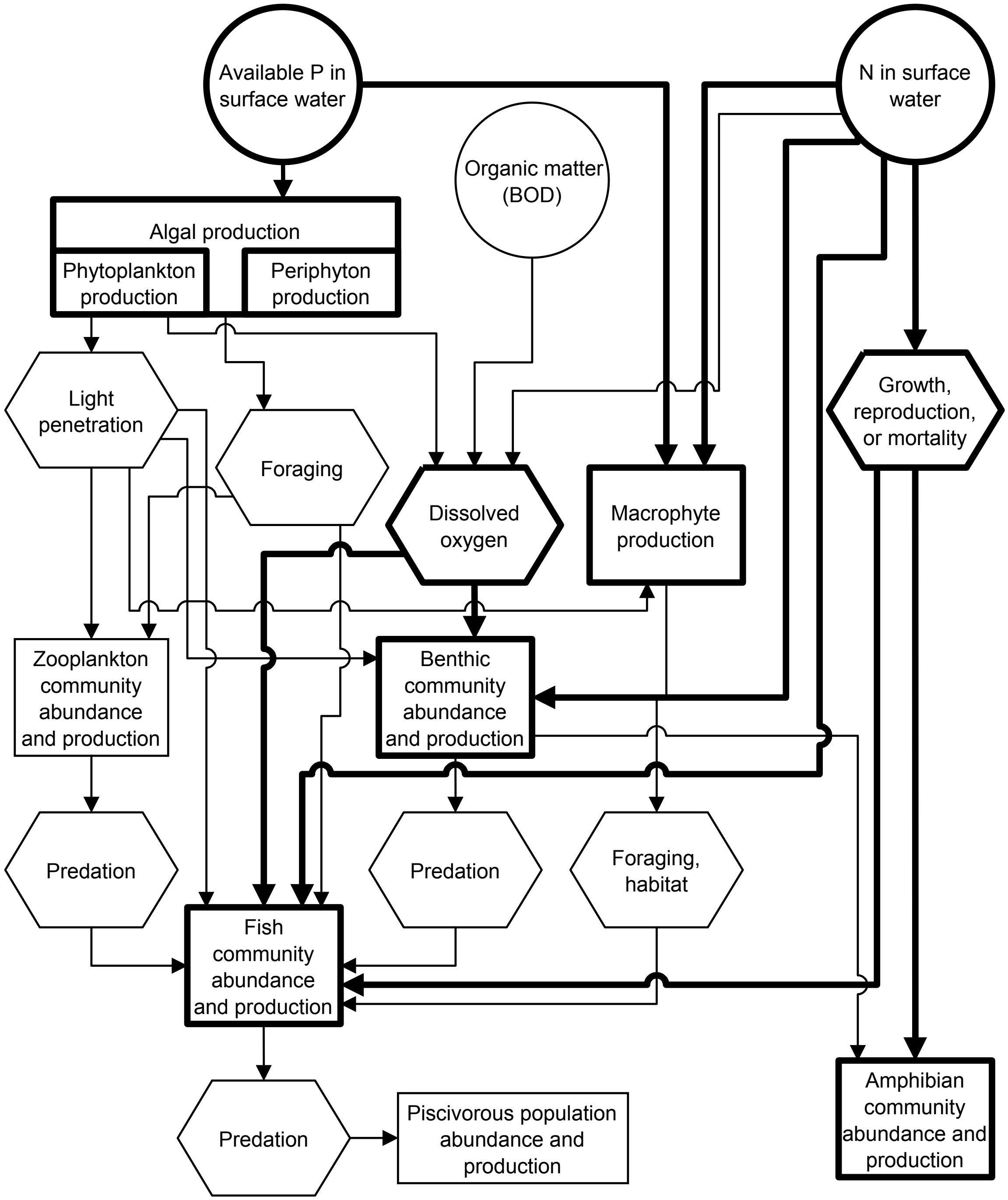

Figure 5-1
Conceptual Model for Ecological Risk Assessment of Wastewater Treatment Unit in a Freshwater Lake, Stream, or Pond

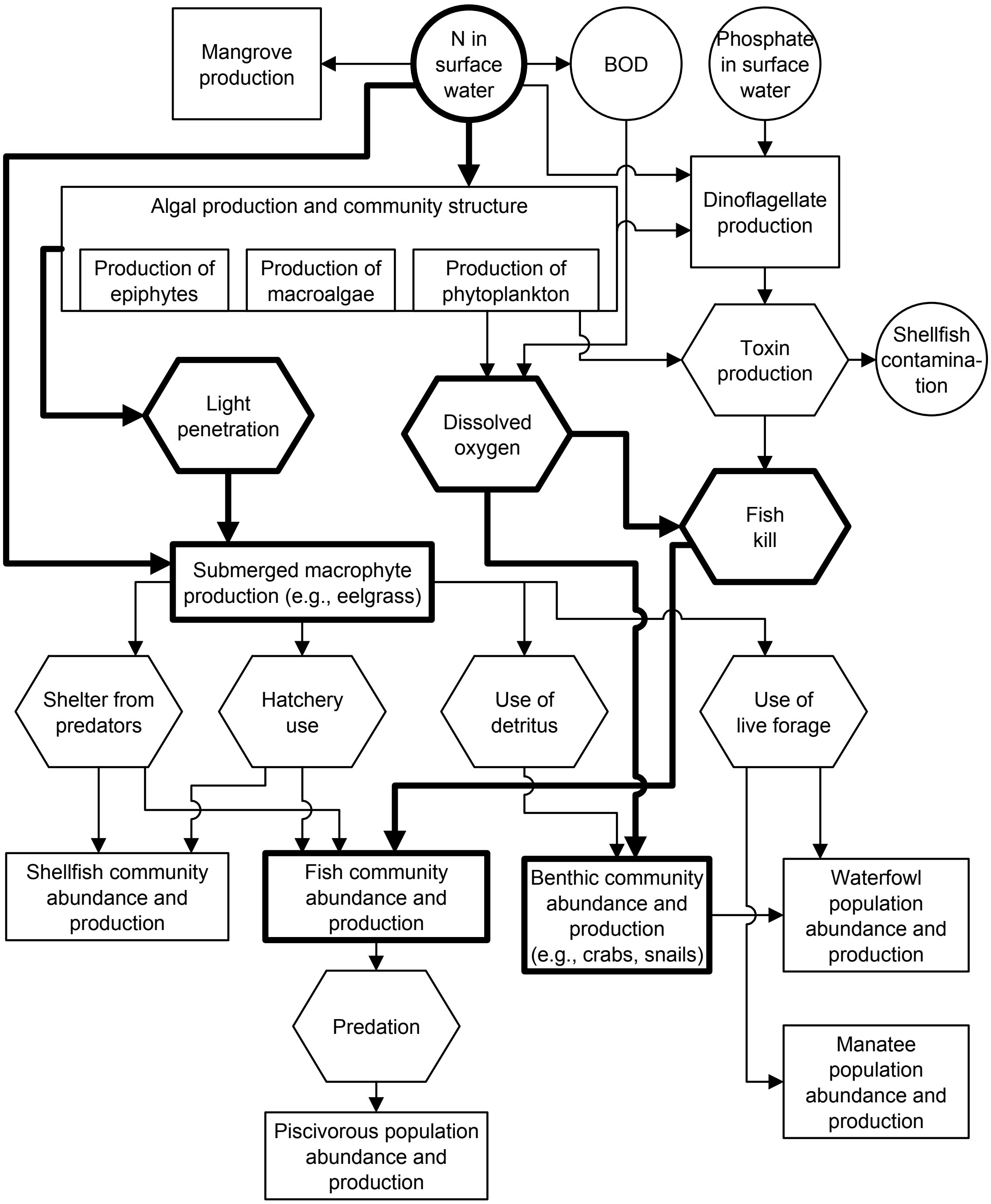

Figure 5-2
Conceptual Model for Ecological Risk Assessment of Wastewater Treatment Unit in a Shallow Estuary or Lagoon

Fresh Water

A generic conceptual model for the potential effects of an OWT system on a freshwater receiving environment (such as a pond) is depicted in Figure 5-1. Phosphorus exposure is the major determinant of phytoplankton production in most North American lakes. This nutrient may also be limiting in streams, but high water flows and flood events may overwhelm the effects of nutrients. Various forms of nitrogen can be directly toxic to aquatic biota, especially reduced forms such as ammonia, though the primary exposures of aquatic organisms and amphibians to nitrogen from an OWT system following release and oxidation in soil are exposures to nitrate. Organic matter that is associated with wastewater and directly released to surface water bodies is an additional stressor that can cause oxygen limitation. A detailed discussion of this conceptual model is provided in Appendix E, *Supporting Information: Ecological*.

Estuary/Lagoon

A generic conceptual model for wastewater treatment unit effects in a shallow estuary or lagoon is depicted in Figure 5-2. Nitrogen is the primary stressor, which can be directly toxic or can interact with biota to produce secondary stressors (limited light penetration, oxygen limitation, reduction in habitat, or reduction in forage vegetation or prey). Organic matter that is associated with wastewater and directly released to surface water bodies is an additional stressor that can cause oxygen limitation. A detailed discussion of this conceptual model is provided in Appendix E, *Supporting Information: Ecological*.

Assessment Endpoints

An ecological assessment endpoint is an explicit expression of a value that is to be protected and that is the subject of analysis in an ecological risk assessment. An ecological assessment endpoint consists of an entity (such as an individual, population, or community), a property of that entity that can be measured or estimated (such as abundance, production, diversity), and, to the extent possible, a level of effect on that property (for example, 20 percent), which constitutes an unacceptable risk. The criteria for selecting assessment endpoints are discussed in Chapter 2, *General Problem Formulation*.

The rectangular boxes in the generic conceptual models in Figure 5-1 and Figure 5-2 are options for assessment endpoint entities and properties for ecological risk assessments for OWT systems. In this ecological risk assessment framework for individual OWT systems, some properties and pathways are selected for emphasis because they are direct effects of nutrients, light limitation, or oxygen limitation. The user may also choose endpoint entities that do not appear on these figures.

The assessment endpoints that are emphasized in this risk assessment framework are presented in Table 5-1. In many ecological risk assessment precedents, the entity that is of concern for plants, fish, and invertebrates is the community (except for threatened or endangered species), whereas populations of terrestrial vertebrates are deemed to merit protection. For this reason, the population is listed as the endpoint entity only for amphibians in Table 5-1. Risks to individual organisms are typically only assessed in cases of threatened or endangered species. However, the user (risk assessor) and decision makers (risk managers) should select endpoint entities that are consistent with the goals of each risk assessment.

Table 5-1
Potential Focal Assessment Endpoint Entities, Properties and Measures for Ecological Risk Assessment of Onsite Wastewater Treatment

Type of Surface Water	Entity[1,2]	Property[1]	Measure of Endpoint Property[1]
Estuary/lagoon	Seagrass population	Decrease in production	Macrophyte biomass density N loading Nitrate-N loading N concentration Epiphyte biomass density Chlorophyll *a* Total suspended solids Number of houses in watershed
	Benthic invertebrate community	Decrease in abundance or production	Concentration of dissolved oxygen
	Fish community	Decrease in abundance or production	Concentration of dissolved oxygen
Fresh water	Phytoplankton community[3]	Increase in production	Algal biomass density Chlorophyll *a* Concentration of available phosphorus
	Macrophyte community	Change in production	Macrophyte biomass density Concentration of available phosphorus

Table 5-1
Potential Focal Assessment Endpoint Entities, Properties and Measures for Ecological Risk Assessment of Onsite Wastewater Treatment (Cont.)

Type of Surface Water	Entity[1,2]	Property[1]	Measure of Endpoint Property[1]
Fresh water (Cont.)	Fish community	Decrease in abundance or production	Concentration of nitrate Concentration of dissolved oxygen
	Benthic invertebrate community	Decrease in abundance or production	Concentration of dissolved oxygen
	Amphibian populations	Decrease in abundance and production	Concentration of nitrate

[1]Endpoint entities, properties and measures are not intended to be an exhaustive list. Entities and properties are the focus of the characterization of exposure and the characterization of effects in this ecological risk assessment framework. Also, an acceptable level of effect is not specific in this component framework, because widely accepted default values for ecological endpoints are not available.

[2]Note that many of these endpoint entities may only be affected at the macro-scale, that is, if large numbers of OWT systems are present. Although exposures to and effects on these entities are generally beyond the scope of this report, many of these exposure-response relationships are described in Appendix E, *Supporting Information: Ecological*.

[3]Particular populations of the phytoplankton community may be additional endpoint entities, such as cyanobacteria.

The ecological relevance of most of these candidate endpoints is described in the conceptual model section. For example, the seagrass community is emphasized because of its role as a unique habitat and breeding ground for many species and because it has been shown to be susceptible to nitrogen in many shallow marine systems. Amphibians are included because of their particular susceptibility to nitrate, which is described in the Characterization of Effects section.

The measures of the endpoint properties presented in Table 5-1 are only examples of the types of exposure-response relationships that might be available for any given assessment. Not all of these measures are useful in all environments. For example, nitrate concentration may not be a useful measure of effects such as seagrass production where nitrate is not observed to increase in water with increases in nitrate or nitrogen loading (see the Characterization of Exposure section and Nixon *et al.* 2001).

Moreover, the risk assessor is not limited to the measures in Table 5-1. Nevertheless, as shown in Figure 5-1 and Figure 5-2, only particular pathways are emphasized here as the focal assessment endpoint properties for this ecological risk assessment framework (for example, the effect of hypoxia on benthic and fish communities, rather than trophic effects).

The risk assessor should state the direction of change that is of concern when defining the assessment endpoint. For example, usually a decrease in production of seagrass is of concern, rather than an increase in production. However, some of these directional designations may vary from assessment to assessment. For example, the loss of macrophytes in freshwater systems may have negative ecological consequences for species relying on that habitat type. However, if the ecological risk assessment feeds into a socioeconomic analysis, an increase in macrophyte production may be of greater concern, because macrophytes may interfere with use of water for fisheries, recreation, industry, agriculture, and drinking (Carpenter *et al.* 1998).

The level of effect on each property that is of concern should be specified in an ecological risk assessment. Ecological levels of effect are not specified in this framework, because these are risk-management decisions that should be made on a site-specific basis. However, the user of this framework should be aware that toxicity data are often reported as LC50 values (concentrations causing mortality to 50 percent of individuals), and the actual concentration of concern may be a lower concentration that affects fewer than 50 percent of individuals of a population.

Measures of Exposure and Measures of Effects

Measures of exposure and measures of effects are the numerical outputs of environmental sampling, analysis, testing, and modeling. Measures of effects are statistical or arithmetic summaries of observations used to estimate the effects of exposure on the assessment endpoint property (Suter *et al.* 2000). They may include test endpoints such as median lethal concentration, LC50, summaries of field measurements or dose-response relationships. The property that is measured is an estimate of the assessment endpoint property, and often the measured property must be extrapolated to obtain the assessment endpoint property. Test species are not necessarily the same as assessment endpoint entities (Suter *et al.* 2000).

Suter *et al.* (2000) review factors that may be considered in selecting measures of effect, which include:

- Correspondence to an assessment endpoint
- Quantifiable relationship to an assessment endpoint
- Availability of existing data
- Simplicity of measurement
- Appropriate scale
- Relationship to exposure pathway

- Relationship to mode of action of stressor
- Specificity to a particular causation factor
- Low variability
- Broad applicability
- Availability of standard test method

Although measures of exposure and effects should be agreed upon during the problem formulation, they are discussed in the characterization of exposure and characterization of effects sections of this ecological risk assessment framework.

Analysis Plan

The analysis plan includes data collection, modeling, and logical analyses that are described or implicit in the risk assessment framework. The plan should contain

- A detailed explanation of how exposure will be measured or modeled
- The exposure-response models that will be used
- How evidence about exposure or effects will be weighed
- How uncertainty will be treated and presented
- A statement of data quality objectives

Analysis

Analysis includes:

- Characterization of exposure
- Characterization of effects

Characterization of Exposure

The characterization of exposure is the phase of an ecological risk assessment in which the spatial and temporal distributions of the intensity of the contact of endpoint entities with stressors (such as nutrients) are estimated (Suter *et al.* 2000). In this phase of assessment, exposure must be characterized in terms that are useful for estimating effects. That is, if the average annual input of phosphorus is known, it may need to be converted to the average annual concentration of phosphorus in the water body if the exposure-response relationship is based on this latter unit.

Exposure of ecological receptors to nutrients is characterized by measurement or modeling. Measurements of most forms of nutrients and dissolved oxygen are easy (with the exception of soluble reactive phosphorus), and if sufficient measurements are taken to characterize spatial and temporal variability, measurement is clearly more accurate than modeling for a risk assessment of current nutrient releases. (Note that measurement cannot distinguish the incremental exposure associated with wastewater treatment releases from other sources of nutrients.)

Prospective risk assessments require modeling of concentrations of nutrients in surface water at the exposure point. Retrospective risk assessments require modeling if historical measurements are not available. In some risk assessments, the modeling would include estimates of nutrient runoff, leaching to groundwater, and possible attenuation in groundwater. Dynamic modeling of nutrient transport is recommended if failure events may cause acute exposure or if responses in exposure-response models are acute or based on particular organism life stages or growing seasons. If dynamic models are used, field-verified models are most reliable.

The output of the characterization of exposure is not usually a single nutrient concentration or loading rate. Exposure is best characterized by a distribution of nutrient concentrations, loading rates, or other measures or exposure through time or space. These may be compared in the risk characterization to distributions of concentrations known to cause effects.

As stated in the problem formulation, in the ecological risk assessment framework that surface water bodies are assumed to be at the edge of the site (at the site boundary). Models that characterize the transport of nutrients through soil or groundwater beyond the edge of the treatment field are not discussed in this characterization of exposure, because they are not unique to ecological risk assessment. An exception is the transport of organic carbon, which can create carbonaceous biochemical oxygen demand (CBOD). This potential stressor is unique to the ecological risk assessment framework and is discussed in this characterization of exposure.

As nutrients and organic carbon enter the surface water, dilution is not instantaneous (unless the point of exposure is a small ditch or vernal pool). Water quality simulation models take loading rates or concentrations at points of entry in the water body and descriptions of mixing and reaction kinetics in a stream reach or other water body segment, and estimate pollutant concentration in a particular water body segment. Additional information regarding nutrient loading to surface waters is provided in Appendix E, *Supporting Information: Ecological*.

Most exposure-response models for ecological receptors in surface water require concentrations of nutrients as measures of exposure. Ecotoxicologists tend to measure or model nutrient concentrations rather than loading rates, though loading rates may be the starting point. However, nutrient loading rates to surface water bodies are not always easily converted to concentrations. (see Appendix E, *Supporting Information: Ecological*). Therefore, assessors should use caution when normalizing volumetric nitrogen loading for residence time to yield an expected or potential concentration. Most effects of concentrations for nitrate are expressed as mass of nitrate-nitrogen per volume, rather than on a nitrate mass basis, which is important to note.

Clearly, CBOD caused by organic carbon from OWT systems is one factor that may contribute to decreased dissolved oxygen concentrations in surface water. OWT system effluent that migrates to surface water through the soil will not likely retain enough organic carbon to create substantial CBOD in receiving waters, based on data from sand filters and the behavior of dissolved organic carbon in sand aquifers (Robertson *et al.* 1991). However, untreated wastewater that breaks through to the soil surface and either flows to the receiving water or is transported in surface runoff could produce locally high CBOD and, therefore, localized areas of hypoxia.

Although low oxygen levels and light attenuation are potential effects of nutrient-enhanced production, we treat them as exposure parameters here because they are not assessment endpoint properties, but rather, they affect assessment endpoint properties. Temperature, reaeration, and rates of photosynthesis and decomposition of nutrient-stimulated phytoplankton and periphyton are also predictors of dissolved oxygen concentrations. Additional information regarding nutrient loading to surface waters is provided in Appendix E, *Supporting Information: Ecological*.

Background Conditions

As stated previously, the risk assessor must decide how to define reference or background concentrations of nutrients, and determine if these concentrations include anthropogenic inputs. If non-anthropogenic background concentrations are unknown, US EPA suggests plotting concentrations in all lakes, streams, or estuaries, and selecting the 25^{th} percentile concentration or plotting concentrations in reference lakes, streams, or estuaries, and selecting the 75^{th} percentile to represent reference concentrations (US EPA 2000b, 2000c, and 2001b).

Characterization of Effects

The characterization of effects is the determination of the nature of adverse effects of contaminants and their magnitude as a function of exposure. The effects are related to the assessment endpoints. These exposure-response relationships may be available or derived from field observations, laboratory, or mesocosm tests with site-specific media, or relationships from published studies. These latter relationships may focus on exposure measures, ecological receptors, and locations that are somewhat different from those of concern in a particular assessment; they may be the only relationships available for retrospective or prospective assessments for which field observations or surface water samples are not available.

Exposure-response models may be

- Empirical models derived from measurements at one or more sites
- Mechanistic models derived from first principles
- Thresholds (exposure levels above which effects occur at a defined level)

In the risk assessment framework presented in this report, only one mechanistic model was identified to characterize ecological effects (see Appendix E). Also, several principles worthy of consideration when choosing among exposure-response models are presented in Appendix E, *Supporting Information: Ecological*.

When field observations are used, it may not be possible to attribute causation, if multiple stressors are present or if multiple sources of one stressor are present.

Thresholds for effects and other exposure-response relationships for the focal assessment endpoint entities are presented in Appendix E, *Supporting Information: Ecological*. No specific level of protection is assumed. For example, because the level-of-effect component of the assessment endpoint is selected by the risk manager, this framework does not arbitrarily present concentrations of nitrate that are associated with a specific percentage loss of a seagrass bed. Examples of many different levels of effect are provided in Appendix E, *Supporting Information: Ecological*.

Water Quality Criteria for Nutrients

Water quality criteria for nutrients may be derived to protect ecological receptors or human health. Those that are derived to protect ecological receptors are candidates for use in the characterization of ecological effects. As with any threshold or other effects level, the relevance to the particular site and ecological receptor of interest should be determined prior to using these values. A detailed discussion of potentially relevant water quality criteria is presented in Appendix E, *Supporting Information: Ecological*.

Exposure-Response Relationships

Exposure-response relationships for the stressors and ecological assessment endpoints (receptors) identified previously as being of particular interest for this framework for individual OWT systems are presented and discussed in detail in Appendix E, *Supporting Information: Ecological*. This section of the framework provides an abbreviated overview of the effects data for a few selected stressors and receptors. The user of this framework would provide a more substantial discussion of the available effects data in the characterization of effects section of an actual risk assessment. Also, it is important for the user to read the original references cited in Appendix E, *Supporting Information: Ecological* to determine if the relationships are valid for the range of exposure concentrations observed or predicted at the site of concern.

The available exposure-response relationships range from thresholds to continuous relationships (Appendix E). Many of these relationships are for single stressors (such as nitrate), although it is clear that combinations of stressors (such as multiple nutrients) or other factors can alter exposure-response relationships. Effects on reproduction, growth, and survival are assumed to relate most directly to the endpoint properties of abundance and production. Deformities are usually related to reproduction as well, and may be considered important by themselves. Therefore, tests that focus on these effects are more pertinent to and useful in the risk assessment than tests of behavior.

As stated in the description of the conceptual model for shallow estuaries, effects on benthic and water column populations are possible due to trophic level interactions caused by changing vegetation (such as relative dominance of seagrass and algae). However, these exposure-response relationships are not presented in Appendix E because these effects are unlikely to occur as a result of releases of nutrients from one OWT system. A more direct effect would be mortality due to low oxygen levels. Low oxygen tends to affect sessile benthic organisms first, because they are exposed to the lowest concentrations of oxygen. Hypoxia is generally defined as a water column oxygen concentrations less than 2 mg/L (Kelly 2001).

Periphyton biomass is not always related to nutrient concentration, and Bourassa and Cattaneo (1998) review several of the studies that observed significant relationships and those that did not. If periphyton biomass or production is an assessment endpoint property in an ecological risk assessment, the assessor should examine all available studies to determine which exposure-response relationships are appropriate to use (or if nutrient inputs are likely to be significant at all).

Recent research has indicated that amphibians may be at risk from nitrate in vernal ponds, lakes and streams. Effects levels for toxicity of nitrate to amphibians are presented in Appendix E, *Supporting Information: Ecological*. Water quality criteria intended to protect human health (10 mg/L) may not be protective of some amphibians. An assessor may choose one or more of these effects levels for use in a risk assessment (depending on the amphibian species of concern), or similar effects levels may be plotted in a species sensitivity distribution analogous to that in Figure 5-3, and the sensitivity of an untested species may be assumed to be a random variate in that distribution. This plot would facilitate comparisons with distributions of nitrate concentrations from the characterization of exposure. However, the assessor should note that nitrate tolerance of amphibians such as the common frog may vary based on the level of adaptation in a particular region (Johansson *et al.* 2001).

Potential synergistic effects are discussed in the Integration of Ecological Risks from Multiple Stressors section, which appears later in this chapter.

Risk Characterization

In risk characterization the information in the characterization of exposure and the characterization of effects is combined to estimate risks. The evidence is often presented in a weight-of-evidence table, with qualitative or quantitative uncertainty. For each line of evidence, several factors are considered, including:

- Data quality
- The relationship of measures of effect to the assessment endpoint
- The relevance of measures of exposure at the site to those that feed into the exposure-response relationship

Screening-Level Risk Assessment

In some ecological risk assessments, the goal may be to eliminate sources, nutrients, or ecological receptors that have no potential for risk. This screening-level ecological risk assessment may consist of comparisons of nutrient concentrations or loading rates to reference concentrations or rates, as well as comparisons to conservative estimates of thresholds for ecological risk (for example, low estimates of effective concentrations of nutrients; high estimates of effective concentrations of dissolved oxygen). Note that “reference” concentrations can refer to natural background or anthropogenic background concentrations, depending on the goal of the risk assessment. The screening-level risk assessment may be the goal of the entire undertaking, or it may be the first phase in a tiered risk assessment, to help focus the assessment on potential problems.

Implementing Exposure-Response Models

Measurements of nutrient concentrations in surface water bodies, modeled loading rates to surface water bodies, or other measures of exposure are available for risk characterization. These values may be available as single values or distributions of variable concentrations with respect to time or space. Effects may be measured in biological surveys or in tests with site-specific media, or using many of the published exposure-response models presented previously that may not be specific to the site where the risk assessment is being conducted.

If sufficient data are available, the risk characterization should consist of a comparison of distributions of exposure and those of probable effects for each assessment endpoint. Exposure and effects characterizations should be performed with concordant spatial and temporal dimensions. An example of a graph used to support a risk characterization for saltwater fish exposed to potentially low dissolved oxygen levels is presented in Figure 5-3.

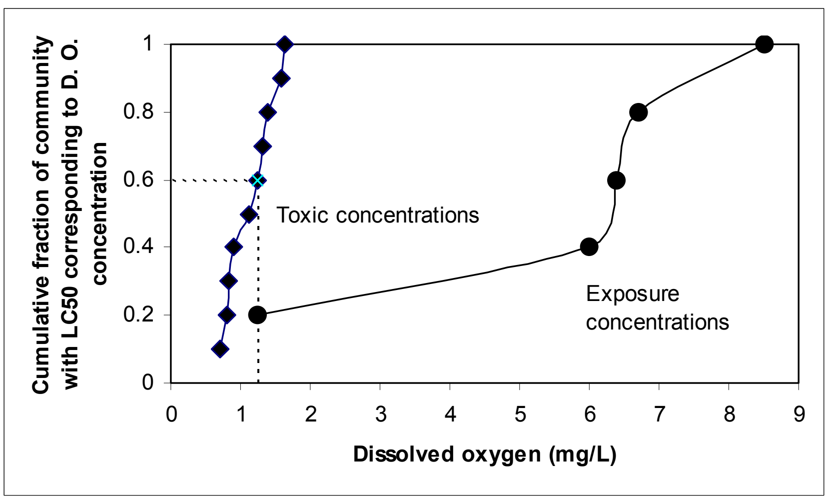

Figure 5-3
Species Sensitivity Distribution of LC50s for Saltwater Fish Exposed to Low Concentrations of Dissolved Oxygen, Compared to Concentrations Measured at Five Locations, Each Representing 20% of the Lagoon Area, at the Sediment-Water Interface of a Hypothetical Lagoon

As shown in Figure 5-3, exposure concentrations are added to the species sensitivity distribution (Figure 5-3). The Y axis is reinterpreted as the fraction of the community affected (rather than fraction of species affected), because exposure concentrations in the example represent five different spatial fractions (each 20 percent) of the fish community (the fish inhabiting this small lagoon). Measurements at four of five locations are well above the median lethal dose (are non-toxic) for any portion of the community, but the dissolved oxygen level at one location

(20 percent of the area) is likely to be lethal for a large fraction of fish species in the community. At one location the dissolved oxygen concentration of 1.25 mg/L would be acutely toxic to approximately 40 percent of the fish community (that is, the LC50s for approximately 60 percent of the tested species were below the exposure concentration). The assessor is cautioned that impacts on growth are likely to occur at higher concentrations of oxygen than those concentrations that are associated with mortality. For example, the US EPA water quality criteria for dissolved oxygen are 6 mg/L and higher (see previous discussion).

Many of the relationships presented in the characterization of effects section are not direct measures of the assessment endpoint property. For example, if the effluent from one treatment system flows into a ditch where tadpoles are located, death of all tadpoles in that ditch probably will not significantly affect the entire local population of frogs. However, such localized impacts would be of concern to many decision makers (risk managers) for OWT issues. That is, residents, neighbors, and local regulatory authorities may not tolerate even highly localized impacts. This intolerance is especially true for obvious impacts such as dead or deformed tadpoles, fish kills, and excessive abundance of aquatic macrophytes or algae.

Weight of Evidence

In many ecological risk assessments, only one line of evidence is available for the risk characterization of an ecological endpoint property and stressor. However, if multiple measures of exposure exist, or if multiple, empirical, exposure-response models are available, all of these may be used to obtain distinct estimates of risks to the assessment endpoints. One example is the use of effects models based on total nitrogen loading and those based on nitrate concentration in lagoons. In addition, biological surveys at the site may be used or performed.

Lines of evidence may be weighted differentially, if the assessor has more confidence in one than in another. The quality of a line of evidence depends on the inherent quality of the models employed and on the quantity and quality of data used to implement them. If differential weighting is implemented, the assessor should present the weights clearly. Ultimately, a determination should be made about whether an adverse effect is likely for a particular endpoint property and nutrient combination. If the goal of the ecological risk assessment is to estimate the magnitude of effect, the estimates of magnitude that result from using different methods to characterize exposure or effects may be weighted, and the result may be a weighted average estimate of the magnitude of effects.

Criteria that may be used to weight evidence include (Suter *et al.* 2000):

- Relevance of data to the assessment endpoint
- Strength of an exposure-response relationship
- Congruence of the temporal and spatial scope of the evidence with the endpoint
- Quality of the data in terms of sampling protocols, expertise, quality assurance, and reasonableness of conclusions

Examples

Two example applications of the ecological weight-of-evidence for individual OWT systems are presented in this section. The first example expands upon the hypothetical exposure-response model presented in Figure 5-3. In this example an emerging OWT system is proposed for a new home being built at the edge of a small estuarine lagoon. The site is considered to be unsuitable for a traditional or emerging OWT system, because the set-back from the lagoon is extremely small. (The porous media biofilter (PMB) and ultraviolet (UV) disinfection units are approximately ten feet from the site boundary at which point the terrain slopes steeply downward for a short distance to the edge of the lagoon.)

The worst-case failure scenario is considered in this example. More specifically, the OWT system is assumed to fail suddenly and catastrophically, discharging untreated wastewater that flows onto the soil surface and then into the lagoon.

At this point it is important to note that:

- This example is strictly hypothetical
- The frequency of the failure event is assumed to be rare, but the consequences are considered to be severe (see Chapter 3, *Engineering Component Framework*)
- The common process controls that would prevent such an event are not considered in this example (that is, the unmitigated risks are considered)

Conservative assumptions and a simple dilution/mixing model are used to predict dissolved oxygen concentrations at five locations in the lagoon. These modeled concentrations are then compared with the effects data for saltwater fish (Figure 5-3). The results of this comparison are then considered in the context of the exposure scenario. In an actual assessment, this would include an extensive discussion of the uncertainties associated with the predicted risks and a concise summary of the evidence and conclusions in a weight-of-evidence table. Only the weight-of-evidence table is presented in this framework example (Table 5-2).

Table 5-2
Example Weight-of-Evidence Summary for Risks to Aquatic Animals in a Small Estuarine Lagoon Exposed to Untreated Wastewater in Run-Off From a Failing Emerging OWT System

Evidence	Result	Level of Confidence in Effect or Non-Effect	Level of Confidence in Cause-Effect Relationship	Explanation
Biological "survey"	N/A	N/A	N/A	Biological surveys are not available because this is a prospective assessment for a new installation (the OWT system in question has not actually failed yet).
Predicted dissolved oxygen (DO) concentrations in the lagoon	+	Medium	High	The failure scenario assumes that untreated wastewater breaks through to the soil surface and flows into the lagoon. The CBOD of the wastewater is assumed to be very high (approximately 300 mg/L). Surface water DO levels are predicted for five locations based on the known volume and flushing rate of the lagoon in question, both of which are relatively small. DO levels at one of the five locations (the one nearest the OWT system) is expected to be acutely toxic to almost half of the fish at that location (the LC50 is exceeded for 40% of the tested species; Figure 5-3). A similar exposure-response analysis for benthic invertebrates also indicates that acute toxicity is likely at the location closest to the OWT system.
Weight of evidence	+	Medium	High	Only one line of evidence is available, but it suggests that a total failure of the OWT system in question would pose unacceptable risks to the aquatic animals (fish and invertebrates) in the lagoon. There is substantial uncertainty associated with the exposure scenario, which was accounted for by using conservative assumptions (the exposures are probably biased high). However, the effects data are considered to be reliable and a fish kill of any size is considered to be unacceptable by the decision makers.

The second example is for a traditional OWT system with a small farm pond located at the edge of the site boundary. The system was installed when a new home was built on a lot that was previously part of a small farm; the previous owners still reside in an adjacent home and retain ownership of the pond. The offsite residents (owners of the pond) and their visitors have noticed a substantial decrease in the number of frogs inhabiting the pond since the OWT system in question was installed. This assessment is based only on their casual observances. They are concerned that wastewater effluent is responsible for the apparent decline in frogs in the pond and that swimming in the pond may not be safe for their young children. Based on this concern, county officials inspect the OWT system and sample the pond for wastewater constituents (such as pathogens and nitrates). The system appears to be functioning properly and the substantial set-back from the pond (100 feet from the drainfield) leads to the conclusion that the OWT system is unlikely to be a significant source of contamination to the pond. The water samples confirm this assumption. Thus, the weight-of-evidence strongly suggests that amphibians inhabiting the pond are not at risk from the OWT system in question (Table 5-3).

Table 5-3
Example Weight-of-Evidence Summary for Risks to Amphibians Inhabiting a Small Pond Potentially Exposed to Nitrogen from a Traditional OWT System

Evidence	Result	Level of Confidence in Effect or Non-effect	Level of Confidence in Cause-Effect Relationship	Explanation
Biological "survey"	+	Low	Low	Residents and visitors have seen and heard fewer frogs in the pond over the last several years. This apparent reduction in abundance coincides with the building of a new house with a traditional OWT system immediately upgradient of the pond approximately five years earlier. However, a scientific survey has not been performed, and amphibian populations are known to be in decline throughout the state in which the site is located.
Predicted nitrate concentrations in groundwater discharging into the pond	–	Low	High	Installation of a traditional OWT system with the drain field located 100 feet from the pond was approved by the county. This set-back distance is based on conservative loading and treatment assumptions for the protection of human health. The nitrate concentration for groundwater entering the pond is assumed to not exceed 10 mg/L (a regulatory threshold for water supply wells in the area). This concentration is below those reported to be toxic in most of the published studies that were determined to be relevant for this site. Another assumption is that the groundwater will be diluted by a factor of ten when discharged into the pond.

Table 5-3
Example Weight-of-Evidence Summary for Risks to Amphibians Inhabiting a Small Pond Potentially Exposed to Nitrogen from a Traditional OWT System (Cont.)

Evidence	Result	Level of Confidence in Effect or Non-effect	Level of Confidence in Cause-Effect Relationship	Explanation
Measured nitrate concentrations in the pond	_	High	Moderate	Water samples were collected from the side of the pond closest to the OWT system on two separate occasions. The measured concentrations were well below all those reported to be toxic in the published studies that were determined to be relevant for this site. There is only moderate confidence that the OWT system in question is the source of the measured nitrate concentrations, because the site is in a farming community (agricultural inputs of nitrogen cannot be ruled out).
Weight of evidence	_	High	High	The weight of evidence strongly suggests that the OWT system in question is not currently having an adverse effect on amphibians inhabiting the pond.

Integration of Ecological Risks from Multiple Stressors

After risks from nutrient and more indirect (such as trophic) stressors are characterized, the risk assessor should integrate these risks for each assessment endpoint property. For example, the stressors produced by wastewater treatment could act together to exert effects. For example, De Solla *et al.* (2002) found that low hatching success of amphibians at agricultural sites in British Columbia may be due to a combination of ammonia, biochemical oxygen demand (BOD), and possibly organophosphates. Hatch and Blaustein (2000) observed interactions between pH and nitrate in determining survival of larval Cascades frogs (*Rana cascadae*), interactions between ultraviolet B (UV-B) and nitrate on the activity level of larvae, and interactions of the three stressors in determining survival in some experiments.

Because risks from trophic interactions are not presented, detailed guidance regarding the summation of risks from multiple stressors is not presented here. However, the assessor may find such guidance in Suter (1999). In this paper, Suter walks the assessor though a number of questions, including:

- Do the stressors or their effects overlap in space or time?
- Are risks from all but one of the stressors relatively inconsequential?

- Are the effects additive?
- Can exposures be added and risks be recalculated?

If effects or exposures are not easily added, a mechanistic model or regression incorporating multiple stressor variables can be used. Regressions that incorporate multiple stressors are available for a few ecological receptors in a few surface water bodies. For example, chlorophyll *a*, a surrogate for phytoplankton biomass, may be predicted in the South Indian River Lagoon, Florida, using a regression that involves three variables: orthophosphate, total nitrogen, and turbidity (Sigua *et al.* 2000).

Uncertainty and Variability

In any ecological risk assessment, sources of variability and uncertainty in results must be described, and wherever possible, quantified. Uncertainty and variability may be associated with both exposure estimates and effects estimates for nutrients. Field biologists often know the approximate magnitude of uncertainty associated with their measurements and the spatial and temporal variability. The error associated with regressions is usually expressed by the researchers who derived the equation, and it may be greater if the relationship is extrapolated from one location to an untested environment, from a mesocosm to the field, or from one species to another. Factors that are not incorporated into models but are known to influence effects that contribute to uncertainty should be presented and discussed.

Summaries of options for qualitative and quantitative uncertainty analysis are found in Warren-Hicks and Moore (1998) and Suter *et al.* (2000). Resampling methods such as Monte Carlo analysis can be time consuming, and the assessor should consider the extent to which a decision will be based on the level of uncertainty before launching numerous quantitative uncertainty analyses.

6 SOCIOECONOMIC COMPONENT FRAMEWORK

The socioeconomic impact and risk assessment component evaluates potential socioeconomic impacts and risks from exposure to wastewater effluent or environmental media that has come in contact with wastewater effluent and efforts to manage those effluents with a wastewater treatment system.

A general framework for conducting a socioeconomic impact and risk assessment of OWT systems is described in some detail along with some examples and hypothetical values. The goal of this framework is to provide guidance for developing qualitative and, where possible, quantitative impact and risk estimates for wastewater effluent of OWT systems and for the OWT systems themselves.

The three-stage risk assessment format used throughout this framework (problem formulation, analysis, and risk characterization) is also used in this component framework. Each aspect of the framework is described in this chapter.

Problem Formulation

In a socioeconomic impact and risk assessment, the problem formulation includes the development of the conceptual understanding of the problem, the selection of assessment endpoints, and the development of an analysis plan. Only the impact and risk assessment at the micro-scale (that is, a single residential OWT system) are addressed in this document; the impacts and risks of OWT systems at larger scales are not addressed (that is, either larger single facilities such as commercial or industrial facilities or OWT systems in an aggregate, such as neighborhoods, communities, regions, or states). Many of the most challenging problems facing the assessment of impacts and risks of OWT systems occur at the macro-scale (Etnier *et al.* 2001 and Wisconsin Department of Commerce 1998) as the result of the cumulative impacts of individual systems and are to be addressed in future research.

However, that does not mean that macro-level issues can be ignored when performing a micro-level assessment. Macro-level issues are relevant in a micro-level assessment to the extent that they create or influence stressors and other aspects of risk assessment that must be addressed in an assessment of an individual OWT system. For example, the presence, capabilities, services, and costs of a maintenance contractor or a responsible management entity are macro-level factors that will affect the likelihood of system failures and will determine what, if any, fees are paid out by the individual system for maintenance and/or oversight.

As discussed in the following sections, many of the impacts and risks wastewater treatment systems pose to the socioeconomic environment grow out of concerns related to the engineering, public health, and ecological dimensions of the system being studied. People value their money, their health, and their ecosystems, meaning that impacts or risks to any of these phenomena result directly or indirectly to impacts or risks to the socioeconomic environment.

Assessment and Management Goals

The overall assessment and management goals are discussed in Chapter 2, *General Problem Formulation*. The implications of those goals for the socioeconomic assessment are best highlighted in the problem formulation of this component framework. That is, the component risk assessment should be planned to address the overall assessment and management goals. In general, decisions made by an individual household may not significantly alter risks to populations or communities, although they may well result in impacts and risks to neighbors. Such offsite impacts are within the purview of a micro-level assessment, provided those impacts are only considered with respect to the individual OWT system being evaluated. For example, it would not be appropriate to evaluate the potential impacts on the community associated with increased residential developments that are only made possible by using emerging OWT systems. However, it may be appropriate to consider the aesthetic impacts (such as sights, sounds, and odors) of an individual emerging OWT system on the adjoining neighbor.

Spatial and Temporal Bounds

As stated in the general problem formulation, this framework is used to address individual OWT systems (the micro-scale). The user of this framework should specify the spatial bounds of the assessment as a particular piece of land (hosting a residence) and that piece of land's immediate neighbors. This framework also is pertinent to treatment systems located adjacent to water bodies and people using those water bodies.

Although the methods presented here may be appropriate for use in assessing impacts and risks to socioeconomic receptors aggregated at a larger scale (such as a neighborhood, community, or region), the focus of this section is on stressors and assessment endpoints at the micro-scale.

The user also should specify the temporal scope of the analysis, which may be based on the lifetime of a treatment system, the lifespan of a particular receptor (such as a homeowner), regulatory requirements, or some other information needed by the decision maker. Particular events in time may be emphasized, such as storm events that cause treatment failure (for example, occurrence of single or multiple one-hundred year floods within a time period of interest). Periods of high releases of nutrients that coincide with particular or unique uses of the property (for example, property owned or used by young or new families with young children or by older residents) should be mentioned or modeled to account for the potential for impacts or risks to particularly vulnerable populations. Periodic use patterns or events characterized by much higher than average rates of use, such as with seasonal residences, may also be important to consider if such patterns affect the operating performance of OWT systems.

Existing Socioeconomic Environment

The impacts or risks of a wastewater treatment system are understood to occur in the context of an existing socioeconomic environment. Thus, the assessor needs to understand and be able to characterize that environment to be able to assess the differences in impacts or risks associated with the given wastewater treatment system as compared with the pre-existing condition of that environment. In typical socioeconomic impact assessment frameworks, the existing socioeconomic environment consists of the pre-existing status of human beings, families, residents, communities, and any other relevant organizational components, and the defining characteristics of those persons (entities or receptors).

For the assessment of impacts and risks of wastewater treatment systems at the micro-scale, the existing socioeconomic environment would include but not be limited to the following characteristics:

- Economic status of the receptors and the receptor's neighbors (see the Receptors section)
- Presence or absence of vulnerable populations among receptors and the receptor's neighbors (that is, vulnerable in terms of susceptibility to health or economic stresses)
- Development status of the receptor's property (as a permanent, temporary, or seasonal residential property) and neighboring properties, including the current value of the properties and the aesthetic qualities of existing land uses
- Existing wastewater treatment capacity/capability of the receptor's and neighbor's environment
- Existing wastewater treatment capacity/capability of the source OWT system, including hydraulic capacity and capability of accepting household chemicals, high CBOD loads, and other components
- Presence, capabilities, services, and costs of a maintenance contractor or a responsible management entity (will affect the likelihood of system failures and will determine what, if any, fees are paid out by the individual system for maintenance and/or oversight)
- Capabilities and willingness of receptors (including those of absentee property owners) to maintain existing or new wastewater treatment systems
- Sensitivity of receptors to intervention(s) taken by outside agents (for example, to inspect or maintain an onsite wastewater treatment system or otherwise take action on the property)
- Temporal and climatic variability of the receptor's environment (such as seasonal, diurnal, or meteorological variation)
- Potential for catastrophic natural events (such as flood, earthquake, landslide, or hurricane)

The assessor needs to identify those characteristics of the existing socioeconomic environment that have the potential for change due to the construction and maintenance of a wastewater treatment system and the values or status of those characteristics prior to taking that action. If the assessor does not have an understanding of the socioeconomic environment prior to the introduction of a change agent (that is, installation or change of wastewater treatment system(s) or initiation of OWT system management), assessing the impacts and risks of that change agent on the socioeconomic environment will be impossible. The acquisition of data for the assessment is addressed in the Assessment Endpoints and Measures of Effects sections.

Receptors

Receptors are entities that are potentially exposed to one or more stressors and can refer to a person, a group of people, or a social or political construct (such as a neighborhood or community). Some or all receptors can be selected as assessment endpoint entities. At the micro-scale of this assessment framework, the receptors include:

- Individuals who are property owners or occupants (permanent, temporary, or seasonally transient)
- Vulnerable subgroups or populations
- Adjacent populations (including any vulnerable populations)

The resources of those individuals or groups of individuals and the relevant characteristics of those receptors (such as socioeconomic status, happiness, wealth, and health) are the important attributes for which endpoints are needed in the assessment.

Stressors

Stressors are any physical, chemical, or biological entity that can induce an adverse response in a receptor, either directly or indirectly. Note that in many cases the effect can be positive or beneficial as well as adverse. Benefits can include increases in property value, increases in development potential, and improved health status (and reductions in health-care costs associated with that improved health status), among others as discussed in the following section. In the context of the socioeconomic impact and risk assessment for wastewater treatment systems, stressors are more likely to indirectly affect the socioeconomic environment—that is, the physical, chemical, or biological change agent that stresses the physical environment, potentially including human health, causes a change in the value of socioeconomic characteristics and resources.

Socioeconomic stressors can be both tangible (such as real monetary costs or changes in property value) and intangible (such as "psychic costs" of allowing others on one's property for periodic system inspection). For many of the stressors (and attributes), the interrelationships among concepts and variables are complicated.

For instance, one argument is that property value is simply a metric for a bundle of characteristics, some of which are physical and tangible (such as size of lot, size of house, number of bedrooms, *and* the kind of wastewater treatment system) while others are more perceptual (such as perception of sanitation, healthiness of a home, aesthetics of landscaping, and sense of well being). For the assessment of impacts and risks of wastewater treatment systems at the micro-scale, stressors would include, but not be limited to, the following:

- Monetary cost of the design and installation or replacement of the wastewater treatment system (Etnier *et al.* 2001 and Wisconsin Department of Commerce 1998)
- Maintenance costs of the wastewater treatment system (Etnier *et al.* 2001)
- Opportunity costs of the wastewater treatment system, which represent value foregone by using an economic resource in a particular way. For instance, opportunity costs include the "cost of money" for the funds used to build and operate the system. Opportunity costs also include the value of any uses of the property that are desired but foregone upon construction of the system; for instance, the space required for a wastewater soil absorption system (WSAS) cannot also be used for a garage. If a garage cannot be built elsewhere on the property, this loss is a real cost due to the OWT system[1]
- Time costs of the wastewater treatment system (such as time required to maintain the system) (Etnier *et al.* 2001)
- Water use (that is, the potential for re-use of graywater or blackwater) (Etnier *et al.* 2001)
- Regulatory compliance costs (such as obtaining permits, variances, time-of-sale inspections, and other costs) (NSFC 1998)
- Restrictions on use due to OWT technology limitations (such as restrictions on use of garbage disposals, water softeners, and other restrictions) (Nelson *et al.* 2000)
- Restricted water use during peak load periods that could affect the use of Jacuzzi tubs and spas in the house
- Restricted organic and suspended solids loading to the system that could restrict use of kitchen facilities
- Uncertainty over system performance and chances of failure, and attendant unpredictability of costs (Otis 1998)
- Intrusiveness of regulation/management (such as periodic access to the property by non-owners for inspections) (Nelson *et al.* 2000)

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[1] Opportunity costs could also include any loss in property value resulting from the failure of an OWT system. A property may not be salable with a malfunctioning or poorly functioning OWT system. In such a case, the opportunity cost of the OWT system would be the entire property value, which is no longer available to the property owner. This highlights the importance of properly maintaining an OWT system (the increment in property value from poor to proper maintenance can be quite large), and the reason why expensive emerging OWT systems must be installed in cases of septic system failure where replacement with a conventional system is not possible due to property size, soils, or other factors. Opportunity costs other than the time value of money are highly situation-dependent; therefore, the example risk assessments given later in this chapter include only the time value of money as an opportunity cost.

- Aesthetic impacts, including noise, smell, and visual impacts of wastewater systems (Etnier *et al.* 2001, Miller 2000, and Wisconsin Department of Commerce 1998)
- Monetary costs to the owner resulting from the impacts of a wastewater system on subsurface and adjacent water bodies; for instance, property value changes due to onsite well contamination, eutrophication of water frontage, or lack of water clarity
- Real or perceived inequities relative to the wastewater treatment systems used by neighbors (that is, whether the benefits and costs/risks are borne by the appropriate actors) (Etnier *et al.* 2001)

The assessor needs to identify those aspects of the wastewater treatment system that could adversely or beneficially affect the socioeconomic environment. Moreover, the assessor must identify those aspects of the system that could affect the environment if the system operates or works successfully and if the system fails (whether due to a design flaw, improper maintenance, or capacity overload due to climatic or behavioral changes). If the assessor does not have a comprehensive listing of those aspects of the wastewater treatment system that can stress the receptors and their socioeconomic environment, it will not be possible to conduct a comprehensive impact and risk assessment for the socioeconomic environment.

The distribution of impacts and risks may vary for different receptors in the socioeconomic environment, which is important to recognize. Depending on the socioeconomic status, life stage, and health status of the individual(s) potentially affected by the successful or unsuccessful wastewater treatment system, the stressor may have a greater or lesser adverse or beneficial impact and risk to the receptor. For example, treatment systems characterized by higher installation and/or maintenance costs would have a greater adverse impact on individuals having less resources than those having more resources. Also, vulnerable populations are more likely to be adversely (in terms of costs) *and* beneficially (in terms of health benefits) affected by advanced and more costly systems. As another example, the development status of a property is likely to have an impact on the acceptability of management schemes. Centralized management, particularly if inspections by government officials are involved, is thought to be more acceptable to homeowners in a new development context, where the management scheme is part of the entire property "package" being purchased, than situations where homeowners are asked to accept central management of OWT systems they have lived with for years (Nelson *et al.* 2000).

In addition to considering the stressors to individuals (as receptors) consideration of stressors to the resources and characteristics of the receptors is important. Table 6-1 provides a conceptual matrix or checklist that an assessor might use to map out the impacts and risks associated with different stressors for the collection of receptors that might be affected by the wastewater treatment system.

Table 6-1
Checklist for OWT System Socioeconomic Receptors and Stressors

Stressors Associated With OWT System	Populations Potentially Affected					
		Occupant				
	Absentee Property Owner	Permanent (Property Owner)	Temporary (Renter)	Seasonally Transient (Vacationers and Second Homes)	Vulnerable Populations	Adjacent Populations (Regular and Vulnerable)[a]
Monetary cost of the design, installation, or replacement of the system						
Maintenance costs of the system						
Opportunity costs of the system (including additional development costs required for system)						
Time costs of the system (time required to maintain the system)						
Information costs of the system (including public education and outreach)						
Regulatory compliance costs of the system						
Aesthetic impacts and risks of the system						
Real or perceived inequities relative to wastewater treatment systems used by neighbors						

[a]This receptor can also include visitors to the property (such as friends of occupants or delivery persons) who could be exposed to and adversely affected by aesthetic or other features of the OWT system.

Time and monetary costs are the principal socioeconomic stressors associated with the micro-scale of the OWT system (residential, onsite treatment systems). The time and monetary costs borne by the different receptors and the distribution of those costs by the receptors are the principal measurements that will need to be characterized in the assessment (see the Analysis section and the Risk Characterization section). Additional, intangible stressors can often be addressed by measuring related time and monetary costs as surrogates.

The risk assessor should specify the receptors and stressors that are the focus of the impact and risk assessment. For example, an impact or risk assessment for a treatment system on, adjacent to, or near seasonally-occupied housing or an industry dependent on water quality (such as recreational water activities or fishing) would usually assess impacts and risks from changes in water quality to seasonal housing occupants and owners, owners of recreational water activities, and the affected fishing industry.

Background Levels of Stressors

As stated previously, the user must define the background or baseline conditions, because impacts and risks are calculated with respect to baseline or background levels. Working with other members of the assessment team, the socioeconomic analyst must determine what activities are taking place that contribute to the existing status of the environment (for example, water quality) so that impacts and risks due to the OWT system of interest can be differentiated from those pre-existing levels.

For example, as pointed out in Chapter 5, *Ecological Component Framework*, in the Stressors section, in addition to effluent from an OWT system (such as septic tanks) nitrogen inputs include:

- Fertilizer application in agriculture and on lawns
- Livestock waste
- Effluents from industrial and wastewater treatment plants
- Atmospheric deposition of oxidized forms of nitrogen from the burning of fossil fuels
- Nitrogen fixation by leguminous crops
- Urban storm water runoff

In this example, the user needs to understand and be able to characterize the pre-existing water quality environment and its stressors to be able to put into perspective the impacts and risks associated with the OWT system of interest. Some of these physical stressors from offsite sources but contributing to the onsite and neighboring environment may have corresponding socioeconomic stressors, such as permitting costs, management system costs, and other costs that are part of the overall socioeconomic environment (though these are generally likely to be of greater relevance at the macro-scale).

Assessment Endpoints

An assessment endpoint is an explicit expression of a value that is to be protected and that is the subject of analysis in a socioeconomic impact and risk assessment. An assessment endpoint consists of:

- An entity (population potentially affected)
- A property or attribute of that entity that can be measured or estimated (such as socioeconomic status, wealth, and health)
- A level of effect on that property that constitutes an unacceptable impact or risk (for adverse effects, where acceptability can be measured as political or social acceptability or some measure commonly agreed to by relevant social and political organizations)

There would likely not be a threshold value (that is, a specified level of effect) for beneficial effects, which is important to note.

In contrast to the engineering, public health, and ecological dimensions of the assessment framework, where acceptability endpoints are better understood and more generally agreed to, assessment endpoints for the socioeconomic dimension of the problem are sometimes more difficult to identify and often more difficult to quantify (in general because they are less amenable to common understanding or agreement). This dilemma is particularly problematic when addressing the intangible values that are important to the assessment (for example, the “psychic costs” or stigma of alternative wastewater treatment systems).

The assessment endpoints emphasized in the socioeconomic component of this risk assessment framework are presented in Table 6-2. These assessment endpoints should be characterized for each entity or receptor, as appropriate (that is, property owner, resident, *and* adjacent property owner/resident, including consideration for vulnerable populations). The assessor, in consultation with the decision maker, may modify endpoint entities to be consistent with the goals of the impact and risk assessment.

Table 6-2
Potential Assessment Endpoint Properties/Attributes and Measures for Socioeconomic Impact and Risk Assessment of OWT

Property or Attribute	Measure	Metric(s)
Economic Status	Design fee	Dollars ($)
	Permit fees	Dollars ($)
	Filing fees (auditor recording of maintenance agreement)	Dollars ($)
	System costs	Dollars ($)

Table 6-2
Potential Assessment Endpoint Properties/Attributes and Measures for Socioeconomic Impact and Risk Assessment of OWT (Cont.)

Property or Attribute	Measure	Metric(s)
Economic Status (Cont.)	Annual Inspection fees	Dollars ($)
	OWT system maintenance and repair cost	Dollars ($)
	Opportunity cost (time value of money for design, installation, and maintenance costs)	Dollars ($) or discount rate (%)
	Change in property value associated with the OWT system	Dollars ($)
	Site evaluation costs	Dollars ($)
	Auditor filing fees	Dollars ($)
	Annual operating fees	Dollars ($)
	Higher design, permitting, and inspection fees for alternative (emerging) technologies	Dollars ($)
Convenience	Difficulty and time spent on system maintenance	Time (h/yr)
	Limitations on water use (on use of garbage disposal and washing machines)	Identity of water use limitation
Aesthetic Quality	Intrusiveness of OWT system in the visual landscape	Identity of OWT system features affecting visual landscape (mounds, risers, red alarm lights, and control panels and alarm boxes on the sides of houses or on posts in yards)
	Indirect impacts on visual landscape	Identity of secondary OWT system feature affecting visual landscape (pond or wetland created with re-use of graywater)
	Changes in noise levels (due to pumps, blowers, and alarms)	Decibel output of OWT system

Table 6-2
Potential Assessment Endpoint Properties/Attributes and Measures for Socioeconomic Impact and Risk Assessment of OWT (Cont.)

Property or Attribute	Measure	Metric(s)
Aesthetic Quality (Cont.)	Change in presence of odors	Nominal and chemical identity (ppm) and source of odors associated with OWT system Focus group measurement of samples of air taken from the odorous source Use of a scentometer to measure the odor threshold of the air in question
Privacy	Change in ability of property owner to determine land use	Identity of land use changes available to property owner attributable to OWT system
	Intrusions by outsiders to maintain/monitor the OWT system	Occurrences/year
Equity	Willingness to bear cost for other's benefit (pay higher cost for OWT system than neighbors have paid for wastewater disposal)	Dollars ($)

The endpoints identified in Table 6-2 and their measures are not always straightforward and require careful and systematic use and modification, as appropriate, by the assessor. Some of the endpoints are conceptually (and operationally) easy to quantify (such as costs), but even those that are easy to quantify, as well as the more intangible endpoints (such as aesthetic or equity considerations), should be understood as values, or objectives or concerns, held by the endpoint entity. Approaches to measurement are discussed further in the next section.

In particular, the assessor must consider carefully both the direct and indirect effects on adjacent properties and their receptors. For example, if the existing wastewater treatment system adversely affects offsite drinking water supplies (such as public or private wells), shellfish beds, and recreational water use, but the OWT system of interest would obviate those adverse impacts, those benefits have to be identified and evaluated. Conversely, if the OWT system of interest would result in adverse impacts to those offsite resources where they had not previously been affected, those adverse impacts have to be identified and evaluated.

The risk assessor should state the direction of change that is of concern when defining the assessment endpoint. In socioeconomic terms, direction is often indicated as a cost or benefit (regardless of whether specific monetary losses or gains are involved). For example, usually an increase in noise level would be considered a cost, and an increase in property value a benefit.

However, some of these directional designations may vary from assessment to assessment, depending on its focus and the overall issues of concern to risk managers. For instance, where affordability of property is a substantial concern, an increase in property value may have negative implications.

Indeed, in socioeconomic systems, a benefit to one entity is often a cost to another. Therefore, the perspective of the analysis—homeowner, neighbor, utility, society at large, or other perspectives—must be carefully articulated in order for costs and benefits to be properly counted. Multiple perspectives are often used in socioeconomic impact analysis, and are sufficient provided each perspective is clearly identified and the accounting for each done appropriate to its proper conceptual boundaries.

As mentioned previously, determining an acceptable level of socioeconomic effect or, conversely, the threshold at which the level of effect is considered unacceptable can be problematic. Even if the assessor can measure an endpoint (such as cost or hours/year), the acceptability of any given measure is not objectively knowable for many, if not all, socioeconomic endpoints. Whereas for public health and other components there may be a level of effect defined by science and/or outside regulatory process as acceptable, for socioeconomic impacts the acceptability of a level of effect is more likely to vary among individuals (based, for instance, on ability to pay for an OWT system).

There will likely be both adverse and beneficial impacts (that is, costs, risks, and benefits) of the OWT system of interest, and the assessor must identify and evaluate all such impacts. Although it may be possible in some cases to integrate these effects into a net effects conclusion, retaining as much information in the assessment as possible is important so that the analysis and all of its conclusions are transparent to the decision maker and other interested stakeholders. This is particularly important for socioeconomic assessments since acceptability in the socio-political sphere is often defined as a matter of trade-offs between effects on endpoints, rather than by absolute effects on specific endpoints. For instance, substantially increased maintenance and management costs may be accepted where they respond to clear threats to property values, public health, or the quality of a water body especially valued by the community (Herring 1996).

The impact and risk assessment should be designed to answer the question of whether the weight of the evidence is sufficient to determine whether the OWT system of interest results in changes in the assessment endpoints. A precise amount of change is not specified in this framework, because it is a management decision. However, the user of this framework should be aware that changes in values for the endpoints are susceptible to variable interpretation by different interested parties or stakeholders and, as pointed out previously, that both the real and perceived changes in value for some, if not all, of those endpoints may be of interest to the decision maker.

Reality of Perception

The source of uncertainty in the socioeconomic portion of the assessment is complicated even further by the reality of perception problem. That is, the perception of an impact or risk, whether imagined or real, constitutes one of the realities for assessing impacts. Thus, the assessor may be faced with the situation where, for any given endpoint, characterizing the objective reality or value of that endpoint (for example, measured by a change in average annual income for the population or entity of interest) as well as the entity's perception of that endpoint (for example, does the entity perceive that its average annual income has changed as a result of the OWT system) is important. For example, the perception may be that an individual system has a significant impact on the environment, but the reality is that this perceived impact is not true; the cumulative impact of a large number of systems has an impact. The decision maker can then take both values (real and perceived) into account in making a decision and attempt to bring those values into agreement where necessary.

Measures of Effects

Compared with the other subcomponents of this framework, some of the measures for socioeconomic impacts are relatively difficult to quantify. As is evident from Table 6-2, some of the measures may only be characterized at a nominal scale (such as if the property owner's power to determine land use will or will not be affected). Some other measures might be characterized on an ordinal scale (such as if the OWT system of interest may result in more or less limitations on the use of water than the existing system or alternative systems).

Depending upon the time and resources available to the socioeconomic assessor, it may be possible to develop quantitative measures for use in a survey instrument. For example, Likert or Guttman scales might be developed to measure the attitudes and perceptions that property owners, residents, and adjacent property owners and residents have regarding the OWT system of interest or its alternatives (Nachmias and Nachmias 1987). Another example would be that the adverse or beneficial impacts of the OWT system on the visual landscape can be measured with a comprehensive and systematic visual landscape assessment protocol involving survey research, the use of computer-enhanced imagery, and scaling procedures. Measures of willingness to pay might be developed and used in a survey instrument (such as a contingent valuation) to ascertain the value of environmental amenities (Anderson and Kobrin 1998). In and of themselves, however, the numbers resulting from such exercises may only be useful in comparative analyses. For instance, a respondent may indicate that an emerging OWT system, as compared to a traditional OWT system, is likely to result in a significant increase in property value, a modest increase in property value, no effect on property value, a modest decrease in property value, or a significant decrease in property value.

Alternatively, real estate appraisal techniques and hedonic valuation techniques may be used to quantify impacts on property values (Boykin and Ring 1993 and Boyle and Kiel 2001). This process is done by finding comparable properties that differ only in the attribute in question. The difference in property value is then considered to indicate the value or cost of the attribute. Finding comparables and isolating attributes is not always possible.

The assessor may be faced with the possibility that data are not readily available for the endpoints of interest or that such data are not available for the particular application being considered. In the absence of exactly applicable data (that is, if no data exist for the specific application and locale of interest), the assessor can identify and examine the suitability of using data from other analogous applications and locales. If the assessor decides to use such data, qualifying any conclusions by noting the differences between the application and locale of the source of the data and the assessment's application and locale is essential.

Parsimony should always be considered in socioeconomic measurements. For instance, techniques to determine dollar values for endpoint properties, such as through contingent or hedonic valuation, can be time and resource intensive. Risk assessors should consider whether a qualitative or semi-quantitative approach based on surveys, focus groups, or other public participation techniques may be simpler and sufficient for the purpose of the socioeconomic risk assessment.

One additional issue that must be considered is the necessity of identifying and characterizing both the actual (objective) values and perceived (subjective) values of the different measures. The assessor might be tempted to minimize these differences (at least for some measures). However, an important realization is that even a measure as seemingly objective as cost of design and installation of the OWT system is a relative measure in the sense that the person bearing the cost may or may not have the disposable income to incur such costs or may or may not be willing to spend their disposable income on an OWT system.

Analysis

Analysis includes characterization of exposure and characterization of effects.

Characterization of Exposure

The characterization of exposure is the phase of a socioeconomic impact and risk assessment in which the spatial and temporal distributions of the intensity of the contact of endpoint entities with the stressors are estimated. For the socioeconomic impact and risk assessment, the exposure is not based on discipline-specific modeling or estimation (as with the other components in this framework) but rather derives from either the presence (or absence) of the OWT system of interest (and results in the effects on the endpoints). The exposure could also derive from the findings of the exposure assessment of the other risk assessment domains—that is, many of the effects are secondary effects. For this approach, the socioeconomic risk assessor would characterize socioeconomic exposure in terms of the spatial and temporal distributions of the intensity of the engineering, human health, and ecological exposures and effects. Thus, for example, if the ecological and human health exposures are large, then the potential for ecological and human health effects may be large, and those, in turn, may lead to large socioeconomic effects (for example, if there is a significant ecological exposure and effect, the indirect effect on property value could be large).

Characterization of Effects

The characterization of effects is the determination of the nature of adverse and beneficial effects of the stressors on the receptors and the receptor's (entity's) properties and attributes. These effects may be available or derived from field observations and research [such as survey research and focus groups (Nachmias and Nachmias 1987 and Stewart and Shamdasani 1990)], secondary data sources (such as census data), or from published studies. In some cases information from these sources may focus on measures, receptors, and locations that are somewhat different from those of concern in a particular assessment, but they may provide the only information available for retrospective or prospective assessments for which site-specific data are not available.

Risk Characterization

In the impact and risk characterization portion of the impact and risk assessment the information in the characterization of effects is used to estimate impacts and risks. The evidence may be presented in a weight-of-evidence table, with consideration of qualitative or quantitative uncertainty (see the Weight of Evidence section). For each line of evidence, several factors are considered, including data quality and the relationship of measures of effect to the assessment endpoint.

Screening-Level Impact and Risk Assessment

In some socioeconomic impact and risk assessments, the goal may be to eliminate stressors and receptors that have no potential for impact or risk. This screening-level impact and risk assessment may consist of comparisons of stressors to existing stressors, as well as comparisons to conservative estimates of thresholds for socioeconomic impact and risk (if any such threshold exists). For example, if the OWT system of interest would result in negligible impact in terms of cost, or if the system would have little or no aesthetic profile that impinges on the aesthetic environment, then the assessor may ignore these features and focus on any other stressors that have the potential for socioeconomic impact or risk. The screening-level impact and risk assessment may be the goal of the entire undertaking, or it may be the first phase in a tiered impact and risk assessment to help focus the assessment on potential problems.

Implementing the Full Impact and Risk Characterization

Measurements of stressors associated with OWT systems are often available for the risk characterization. The engineering subcomponent should provide information regarding the design, installation, and maintenance and repair costs of the OWT system of interest. The assessor can stipulate, by assumption, the cost of money or assume variable costs of money (for opportunity cost). Likewise, the engineering subcomponent should supply information regarding the time cost of the OWT system (in terms of hours per month to maintain the system) and whether that time is a cost to the property owner or to a firm contracted for OWT system maintenance. For socioeconomic risk assessment of operational failure, the engineering subcomponent would provide estimates of rates of failure, which then translate to frequency and severity of socioeconomic stressors such as cost to repair a system, aesthetic (such as olfactory) insult from failure, and other stressors.

If enough data are available, the impact and risk characterization should consist of a comparison of distributions of probable effects for each assessment endpoint, comparing the OWT system of interest to other wastewater treatment alternatives or to the status quo. Some of the descriptions presented in the characterization of effects section may not be direct measures of the assessment endpoint property. The assessor may have to extrapolate from the property value of a similar property (as property assessors and real estate agents typically do in valuing properties) to the value of the property of interest and identify a threshold value of OWT system (or existing wastewater system) output that results in diminution or enhanced value.

In addition, in some cases, the assessor may have to extrapolate from a different part of the assessment. For example, if the ecological risk assessment portion of the framework indicates that the effluent from one treatment system flows into a ditch where tadpoles are located, the assessor needs to evaluate if death of all tadpoles in that ditch would significantly affect the property's

- Aesthetic quality
- Monetary value
- Recreational use

Weight of Evidence

In many socioeconomic impact and risk assessments, only one line of evidence may be available for the impact and risk characterization of a socioeconomic endpoint property and stressor. However, if multiple measures are available, all of these may be used to obtain distinct estimates of impacts and risks to the assessment endpoints. As a general rule of thumb, the more lines of evidence available, the better, because they tend to decrease the uncertainty in the estimate. However, if the multiple lines of evidence indicate different conclusions (in direction or magnitude of effect), the assessor will have to evaluate the quality of the data and their confidence in its accuracy.

Lines of evidence may be weighted differentially, if the assessor has more confidence in one line of evidence than in another. The quality of a line of evidence depends on the state of the knowledge and understanding about socioeconomic phenomena and on the quantity and quality of data available to understand them. If differential weighting is implemented, the assessor should present the weights clearly and explain his or her rationale for differential weighting. Ultimately, a determination should be made about whether an adverse effect is likely for a particular endpoint property. If the goal of the socioeconomic impact and risk assessment is to estimate the magnitude of effect, the estimates of magnitude that result from using different methods to characterize effects may be weighted, and the result may be a weighted average estimate of the magnitude of effects.

Hypothetical Socioeconomic Impact and Risk Characterization of a Traditional OWT System

The socioeconomic impacts and risks of a traditional OWT system (shown in Figure 2-2 and referred to as Scenario 2 in this document, which is a residence on relatively flat terrain using a septic tank and a septic tank effluent line to a wastewater soil absorption system) are addressed in this section. With the exception of a few data points (such as typical cost for installation), this characterization is purely hypothetical and is provided simply to walk the reader through the concepts and information presented in this section.

Before conducting the assessment itself, the assessor needs to characterize the baseline socioeconomic environment (see the Existing Socioeconomic Environment section). This characterization must include information regarding

- The economic status of the property owner/resident and of the property's neighbors
- Whether there are any vulnerable populations living in the residential property or in neighboring properties
- The development status of the property in question (in this case a single-family residence) and its neighbors (including the current value of the property and its aesthetic quality)
- The existing wastewater treatment capacity of the property and of neighboring properties
- The property owner's capabilities and willingness to maintain the OWT system
- The sensitivity of the property owner to allow outsiders to be on the property for inspecting and maintaining the OWT system
- The temporal and climatic variability of the property's environment
- The potential for catastrophic natural events

As an example, a summary of the socioeconomic impacts and risks of a properly installed, maintained, and functioning traditional OWT system (that is, a Type 2 system) for an application at a house occupied by the property owner is provided in Table 6-2. Recall also other potential receptors identified in Table 6-1 need to be considered in the assessment.

The impacts on economic status include a number of separate items. The following assumptions were made for Scenario 2:

- The monetary cost of designing and installing this system is estimated at between $2,500 and $3,500 (Etnier *et al.* 2001); Etnier *et al.* report that costs can vary from a low of approximately $1,500 in southern states to $6,000 to $8,000 or more in the more urbanized East and West Coast
- The maintenance (and repair) costs are assumed to cost $100 on an annual basis; as Etnier *et al.* (2001) report, the assessor can use the predictive cost-estimation model Costs of Onsite Management Options(COSMO) developed at North Carolina State University to estimate these costs (Renkow and Hoover 1996)

- The discount rate used to estimate the opportunity cost is assumed to be 5 percent, meaning that the cost of money for design and installation varies between $125 and $175 (assuming a one-year payback)
- No additional development costs are required for installation of the system (such as removal of a failed septic tank)
- The impact of the septic system on property value is a gain of $5,000 as compared to the property being vacant or undeveloped; if the alternative is to be on sewer, Etnier *et al.* (2001) report on a study in Massachusetts that homeowners believe that being connected to a sewer increases the value of the property by $9,000 over a property on a septic system[2]. Property value is a summary metric for a bundle of characteristics, some of which would be affected by the installation of a traditional OWT system (such as perception of sanitation and healthiness of the home) but others would be unaffected by such an installation (such as lot size). If the system is being installed in an area where other properties have no wastewater treatment (that is, effluent is piped directly to surface waters or land), the impact on property value may be greater. The property owner, however, might be hesitant to invest resources when his neighbors' failure to protect the environment would lower the value of his or her property. Property values can vary greatly depending on geographic location or ease of OWT development. For example, $5,000 may be an appropriate value for a half-acre parcel in rural Kentucky, but it is not appropriate for a one-acre parcel in Monterey, California.

Table 6-3
Impacts and Risks of a Traditional OWT System (Scenario 2)

Property or Attribute	Measure	Property Owner/Resident	Adjacent Population
Economic Status	Design fee	(–$100)–(–$350)	$0
	Permit fees	(–$0)–(–$100)	$0
	Filing fees (auditor recording of maintenance agreement)	(–$0)–(–$50)	$0
	System costs	(–$2,500)–(–$3,500)	$0
	Annual inspection fees	(–$25)–(–$100)	$0
	OWT system maintenance and repair cost	(–$100) per year	$0

[2] A different kind of impact on property value can occur in the case of poor maintenance of an OWT system. In some cases of OWT system failure, the property may no longer be salable, and the property value on the market becomes zero. Poor OWT system functioning, short of failure, may leave a property salable, but at a reduced value. These situations point out that proper OWT system maintenance, including any fees for management paid to a local government or responsible management entity, should be seen as an investment necessary to maintaining one's property value.

Table 6-3
Impacts and Risks of a Traditional OWT System (Scenario 2) (Cont.)

Property or Attribute	Measure	Property Owner/Resident	Adjacent Population
Economic Status (Cont.)	Opportunity cost (time value of money for design, installation and maintenance costs)	(–$125)–(–$175)	$0
	Change in property value associated with the OWT system	+$5,000	Unknown but real
Convenience	Difficulty and time spent on system maintenance	(–5) hours/year	0
	Limitations on water use (on use of garbage disposal and washing machines)	Cannot have a garbage disposal and restricted use of waster softener	0
Aesthetic Quality	Intrusiveness of OWT system in the visual landscape	Construction debris during installation, then benign	Construction debris during installation, then benign
	Indirect impacts on visual landscape (if pond or wetland is created with re-use of graywater)	0	0
Privacy	Change in ability of property owner to determine land use	Favorable	Unknown
	Intrusions by outsiders to maintain/monitor the OWT system	Once/year (minor)	0
Equity	Willingness to bear cost for other's benefit (pay higher cost for OWT system than neighbors have paid for their wastewater disposal)	Unfavorable	Favorable

The impacts of the septic system in terms of convenience are also important to consider. The time cost for the system is assumed to be approximately five hours per year (for the homeowner to contract for septic tank pumping). A septic system might restrict the homeowner's ability to utilize certain modern conveniences, at least as compared to being on sewer—the septic system might not be able to accommodate refuse from a garbage disposal (Nelson *et al.* 2000) or the system might restrict the use of a water softener, thereby compromising the convenience and water supply aesthetics for the resident (NSFC 2001).

The impact of the septic system in terms of aesthetics should be temporary during construction, but benign during routine operations. If the septic tank is not pumped or maintained, or if the capacity of the wastewater soil absorption system is exceeded, system failure can result in noxious odors that will persist until the system is repaired. These odors can adversely affect not only the property owner but neighbors as well.

In Scenario 2, there is no assumption regarding the use of graywater or blackwater, because such use would likely be prohibited for health as well as aesthetic reasons. Minor enhancements to the traditional septic system might allow the use of graywater for irrigation and landscape features (Etnier *et al.* 2001), but the use of blackwater would require more advanced onsite wastewater treatment systems.

The impact of the septic system in terms of privacy should be favorable and minor. The favorable impact is that the property owner would be able to develop the property for residential use and, if it is assumed that the septic tank and septic tank effluent line are pumped and cleaned once per year, there would be minor intrusions on the property owner's privacy.

The equity impacts of the septic system depend entirely upon how and at what cost neighboring properties manage their wastewater. If the property owner implements a septic system while his or her neighbors are on sewer, the neighbors might perceive that having a neighbor on a septic system reduces their property's value and that they (the neighbors of the new septic system) are having to pay more than their fair share of reducing adverse environmental impacts.

Hypothetical Socioeconomic Impact and Risk Characterization of an Emerging OWT System

The socioeconomic impacts and risks of an emerging OWT system (as shown in Figure 2-4 and referred to as Scenario 4 in this document, which is a residence on relatively flat terrain using a septic tank, a septic tank effluent line to a porous media biofilter (PMB) linked to a pump and ultraviolet lamp prior to discharge) are addressed in this section. This characterization is purely hypothetical and is provided simply to better explain the concepts and information presented in this section.

Before conducting the assessment itself, the assessor needs to characterize the baseline socioeconomic environment (see the Existing Socioeconomic Environment section). This characterization must include information regarding

- The economic status of the property owner/resident and of the property's neighbors
- Whether there are any vulnerable populations living in the residential property or in neighboring properties
- The development status of the property in question (in this case a single-family residence) and its neighbors (including the current value of the property and its aesthetic quality)
- The existing wastewater treatment capacity of the property and of neighboring properties
- The property owner's capabilities and willingness to maintain the OWT system

- The sensitivity of the property owner to allow outsiders to be on the property for inspecting and maintaining the OWT system
- The temporal and climatic variability of the environment
- The potential for catastrophic natural events

A summary of the socioeconomic impacts and risks of a properly installed, maintained, and functioning emerging system (Scenario 4) for an application at a house occupied by the property owner is provided in Table 6-4. (Other potential receptors identified in Table 6-1 need to be considered in the assessment.)

Table 6-4
Impacts and Risks of an Emerging OWT System (Scenario 4)

Property or Attribute	Measure	Property Owner/Resident	Adjacent Population
Economic Status	Design fee	(–$350)–(–$1500)	$0
	Permit fees	(–$380)–(–$500)	$0
	Filing fees (auditor recording of maintenance agreement)	(–$0)–(–$50)	$0
	System costs	(–$9000)–(–$25,000)	$0
	Annual inspection fees	(–$50)–(–$200)	$0
	OWT system maintenance and repair cost	(–$400) per year	$0
	Opportunity cost (time value of money for design, installation and maintenance costs)	(–$250)–(–$500)	$0
	Change in property value associated with the OWT system	+$5,000 – +$10,000	Unknown but real
Convenience	Difficulty and time spent on system maintenance	(–10) hours/year	0
	Limitations on water use (on use of garbage disposal and washing machines)	Cannot have a garbage disposal and restricted use of water softener	
Aesthetic Quality	Intrusiveness of OWT system in the visual landscape	Construction debris during installation, then benign	Construction debris during installation, then benign

Table 6-4
Impacts and Risks of an Emerging OWT System (Scenario 4) (Cont.)

Property or Attribute	Measure	Property Owner/Resident	Adjacent Population
Aesthetic Quality (Cont.)	Indirect impacts on visual landscape (if pond or wetland is created with re-use of graywater)	0	0
Privacy	Change in ability of property owner to determine land use	Favorable	Unknown
	Intrusions by outsiders to maintain/monitor the OWT system	Twice/year (minor)	0
Equity	Willingness to bear cost for other's benefit (pay higher cost for OWT system than neighbors have paid for their wastewater disposal)	Unfavorable	Favorable

The impacts on economic status include a number of separate items. The monetary cost of designing and installing this system (that is, system costs plus design, permit, and filing fees) is estimated at between approximately $10,000 and $27,000 (a cost assumed for this purpose to be at least four times as much as the traditional septic system).

The maintenance (and repair) costs are assumed to cost $400 on an annual basis (a cost assumed for this purpose to be approximately four times as much as the traditional septic system to account for UV disinfection and for PMB maintenance). The discount rate used to estimate the opportunity cost is assumed to be five percent, meaning that the cost of money for design and installation varies between $250 and $500 (assuming a one-year payback). The assumption for Scenario 4 is that no additional development costs are required for installation of the system (such as removal of a failed septic tank). The impact of the emerging system on property value is assumed to be comparable to the impact of septic system on property value—a gain of $5,000 to $10,000 as compared to the property being vacant or undeveloped. If the alternative is to be on sewer, Etnier *et al.* (2000) report on a study in Massachusetts that homeowners believe that being connected to a sewer increases the value of the property by $9,000 over a property on a septic system.

If the system is being installed in an area that has difficult soils and a history of septic pollution (ponding or odors or regular reports of failures and backups and repairs), the impact on property value may be greater than assumed here because the alternative system would carry additional value due to both the perceived reduction in health risks and the perceived reduction in economic risk that would be associated with a traditional system that is perceived to be susceptible to failure. Additionally, an alternative system (emerging system) could be the only way that the property can be developed (for example, due to depth to groundwater requirements being less strict for emerging technologies than for traditional septic systems). On the other hand, as previously discussed for Scenario 2 (the traditional OWT system), the property owner might be hesitant to invest his resources in a costlier emerging OWT system because his property's value would be diminished by the septic system failures of his neighbors.

The impacts of the emerging system in terms of convenience are also important to consider. The time cost for the system is assumed to be approximately 10 hours per year (for the homeowner to contract for annual cleaning of the PMB). Note that this is greater than for a conventional system due to the additional operational and maintenance requirements of an emerging system. An emerging system might restrict the homeowner's ability to utilize certain modern conveniences, at least as compared to being on sewer—the emerging system might not be able to accommodate waste from a garbage disposal (Nelson *et al.* 2000) or the system might restrict the use of a water softener, thereby compromising the convenience and water supply aesthetics for the resident.

According to the National Small Flows Clearinghouse (NSFC 2001), some experts believe that salty water softener regeneration brine may adversely affect the functioning of septic systems. Other experts disagree. If a softener caused problems, a dry well or other separate system for disposal of brine might be necessary, increasing costs.

The impact of the emerging system in terms of aesthetics should be temporary during construction but reasonably benign during routine normal operations, with only the system's pump and ultraviolet lamp being visible during operation.

The impact of the emerging system in terms of privacy should be favorable and minor. The favorable impact is that the property owner would be able to develop the property for residential use. An emerging system will likely require more frequent inspection and maintenance than a conventional system. The assumption in Scenario 4 is two visits per year. Such visits could be considered an invasion of privacy. Homeowners typically consider visits by personnel they choose through a maintenance contract to be less of an invasion than visits by government personnel inspecting the system. The two visits per year are assumed to be a minor (acceptable) invasion of privacy.

The equity impacts of the emerging system depend entirely upon how and at what cost neighboring properties manage their wastewater. If the property owner implements an emerging system while his or her neighbors are on septic systems, the property owner may perceive that they are unfairly having to pay to reduce their pollution while the neighbors' costs are less—the neighbors are free-riders. If the property owner implements an emerging system while his or her neighbors are on septic systems, the neighbors may perceive that having a neighbor on an emerging system enhances their property's value and that they (the neighbors of the new emerging system) are having to pay less than their fair share of reducing adverse environmental impacts. In short, the assessor must consider impacts to a multiplicity of stakeholders—the property owner and the neighbors for a micro-scale assessment, and many others (such as the neighborhood, community, and region) for a macro-scale assessment.

What should be apparent from evaluating Scenarios 2 and 4 is that the socioeconomic impact and risk assessment is, to a great extent, contextual. Demographic information about the socioeconomic status of the property owner (such as age, family size, and economic status) as well as the property owner's neighbors may be critical to the assessor's ability to characterize and, where possible, quantify the impacts and risks of a given OWT system.

Uncertainty and Variability

In any socioeconomic impact and risk assessment, sources of variability and uncertainty in results must be described, and if possible, quantified. Social scientists often know the approximate magnitude of uncertainty associated with their measurements and their spatial and temporal variability. The error may be greater if the information is extrapolated from

- One location to an untested environment (for example, from New England to the South or from a rural area to a suburban area)
- An individual having ample resources to one living barely above the poverty level
- An established residential suburban area to a vacation area offering water recreation amenities including fishing, boating, and swimming

Factors that are not incorporated into the assessment but are known to influence effects contribute to uncertainty.

Summaries of options for dealing with sources of error and qualitative and quantitative uncertainty analysis are found in standard social science methods references (such as Nachmias and Nachmias 1987 and Blalock 1960). If there is substantial uncertainty regarding variables and values that are deemed critical to the assessment, the assessor can conduct sensitivity analyses, thus providing the decision maker a sense of the potential variability in conclusions that might be reached.

In the final analysis, however, many of these sources of error or uncertainty are unavoidable (or too costly to overcome). Thus, it is the assessor's responsibility to identify all potential sources of error, uncertainty, and bias that he or she can so that the decision maker is aware of the potential impact of those uncertainties on the decision to be made.

7 GENERAL RISK CHARACTERIZATION

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

1. Summarizing the risks and uncertainties characterized in each component assessment

2. Characterizing the integrated risks and uncertainties for assessment endpoints that are potentially affected by one or more other assessment endpoints

3. Summarizing the integrated risks and uncertainties characterized in this section

This approach highlights the fact that risks can be divided into two general categories for the purposes of this framework:

- Independent risks
- Conditional risks

That is, the estimation of some risks is independent of the estimation of all other risks to a particular assessment endpoint, which is referred to as independent risks in this framework. Conditional risks are those for which the estimation of risk is conditional on the estimates for one or more other risks. This dichotomy is discussed further in the following section.

Principles of Risk characterization

Risk assessment is the scientific process of estimating the likelihood and magnitude of adverse effects. Risk characterization is the final step in the risk assessment process. Risk management is the process of deciding which actions to take in response to a particular risk. That is, risk characterization is intended to be an objective process, whereas risk management is intended to be a subjective process in which value judgments are made.

Four general goals for risk characterization and the potential means of achieving those goals are provided by US EPA (1998b):

- Clarity
  - Be brief; avoid jargon
  - Make language and organization understandable to risk managers and the informed lay person

- Fully discuss and explain unusual issues specific to a particular risk assessment

- Transparency
  - Identify the scientific conclusions separately from the policy judgments
  - Clearly articulate major differing viewpoints of scientific judgments
  - Define and explain the risk assessment purpose (regulatory purpose, policy analysis, priority setting)
  - Fully explain assumptions and biases (scientific and policy)
- Reasonableness
  - Integrate all components into an overall conclusion of risk that is complete, informative, and useful in decision making
  - Acknowledge uncertainties and assumptions in a forthright manner
  - Describe key data as experimental, state-of-the-art, or generally accepted scientific knowledge
  - Identify reasonable alternatives and conclusions that can be derived from the data
  - Define the level of effort (such as quick screen or extensive characterization) along with the reason(s) for selecting this level of effort
  - Explain the status of peer review
- Consistency with other risk characterizations
  - Describe how the risks posed by one set of stressors compare with the risks posed by a similar stressor(s) or similar environmental conditions
  - Indicate how the strengths and limitations of the assessment compare with past assessments

These are generally sound recommendations that can often be applied directly to risk assessments of onsite wastewater treatment (OWT) systems, without extensive explanation herein. Instead, exceptions and caveats to those methods for risk characterization of OWT systems are addressed in this section.

One way in which this risk characterization differs from others is that subjective factors are normally excluded from a risk assessment to avoid biasing the characterization of risk. However, this integrated framework explicitly includes subjective measures of impact via the socioeconomic component framework. This process is consistent with the overarching goal of providing risk managers with all the information needed to make sound and viable decisions. That is, the socioeconomic component gives decision makers a tool with which to gauge the potential for societal impacts.

A fundamental problem encountered when trying to achieve the aforementioned general goals for risk characterization is that the goals of being brief and being transparent conflict (Suter *et al*. 2000).

Providing all of the details needed to understand fully how the risk characterization results were derived could result in a large, cumbersome general risk characterization. This problem is partly avoided in this framework by providing a separate risk characterization for each component framework.

The means by which independent risks were characterized for each assessment endpoint can be discussed in sufficient detail in the component assessments to avoid discussing those details in the general risk characterization. The results still need to be presented in adequate detail for decision making (Suter *et al*. 2000). For the summary of previously characterized risks this information should include, at a minimum, a rating for both the estimated risks and the uncertainties associated with those estimates.

Conditional risks are best evaluated in the general risk characterization. Therefore, the user must provide additional details in this section regarding the characterization of risks that are dependent on the estimates for one or more other risks. Potential issues and methods are discussed in the following section.

Independent Risks

Independent risks, as noted previously, are those that can be estimated without first having to estimate the risks associated with a separate assessment endpoint. For example, the risk of treatment failure (dysfunction) due to seasonal flooding of the wastewater soil absorption systems (WSAS) can be estimated in the engineering risk assessment even if the other three component assessments are not performed. The user must make some assumptions regarding the severity of the impact of this type of failure event on other assessment endpoints (such as exposure of residents to pathogens), but the risk of treatment failure is not truly conditional on the estimated risk of infection of residents by a virus. In contrast, the risk of infection by a virus is, in part, conditional on the risk of having a treatment failure due to seasonal flooding of the WSAS, which then results in exposure of the residents to viruses (this example is discussed further in the section on conditional risks).

This difference is subtle, but the distinction is useful for purposes of streamlining the risk characterization process. Making a distinction between an assumed parameter (such as the criterion used to estimate the severity of a failure) and an estimated risk (such as the probability and magnitude of a specific exposure scenario) means that independent risks and conditional risks can be addressed in separate and different ways in the general risk characterization.

For example, the user might choose the concentration of viruses that defines failure of the OWT system (the failure criterion) based on a virus concentration that is assumed (or estimated) to pose a certain level of risk to the receptors of concern (such as 1:10,000 incidence of infection). That criterion concentration can then be used in the engineering risk assessment to estimate the probability and consequences of OWT system failure due to flooding. That risk estimate can then be summarized in the general risk characterization as the independent risk of failure of the OWT system due to flooding, with reference back to the engineering assessment for a detailed discussion of the risks and uncertainties.

Rating Systems

A rating system can be used to summarize the estimated risks for purposes of risk communication and decision making. Some component assessments include a rating system in their design, whereas others may just report the results without explicitly classifying the risks as being more or less than the assessment endpoint level. The former method is preferred over the latter. For example, the engineering framework in this assessment uses a Roman numeral ranking system ranging from I to IV (see Chapter 3, *Engineering Component Framework*) and the ecological framework uses a weight-of-evidence process to assign a plus/minus rating for each assessment endpoint (see Chapter 5, *Ecological Component Framework*). These ratings can be used without modification for the general risk characterization.

However, a rating should be assigned in the general risk characterization section if the component assessment does not do so directly. For example, the risk of infection assessed in the public health component framework is reported as an estimated rate of infection (such as 1:10,000). Such quantitative estimates should be reported in the component assessments wherever possible. Instead of repeating those estimates, risks can be characterized (rated) as being less than or greater than the selected assessment endpoint level.

A simple acceptable/unacceptable rating system is a useful and intuitive tool for this purpose (Suter *et al.* 2000). The decision rules for that system are:

- **U**—Indicates that the estimated risk exceeded the selected level of effect for the assessment endpoint and is unacceptable
- **A**—Indicates that the estimated risk did not exceed the selected level of effect for the assessment endpoint and is acceptable
- **I**—Indicates that insufficient evidence was available to conclude whether the selected level of effect for the assessment endpoint was exceeded (that is, the acceptability of the risk is indeterminate)

Summarization the uncertainties (that is, confidence) associated with the characterization of risk is also necessary. Uncertainties for each assessment endpoint are discussed in detail in the component assessments. Those uncertainties need to be summarized in a way that helps decision makers compare disparate types of risks and uncertainties in a holistic manner. That is, decision makers will need to weigh all risks (and associated uncertainties) for all assessment endpoints. To support the risk-management process, the general risk characterization needs to summarize the previously detailed uncertainties in a simple and consistent manner.

A simple and effective rating method entails classifying the level of confidence associated with a risk rating as being low, moderate, or high. These rankings must be applied consistently across all assessment endpoints to be useful. Each category should be defined clearly in the general risk characterization. These definitions need to relate back to the uncertainty descriptions provided for each assessment endpoint to ensure consistency within the overall assessment.

Assigning a confidence rating to each endpoint/risk combination in this section may require consensus among multiple assessors. That is, the assessor responsible for each component assessment should be consulted when assigning each rating, even those not associated with that assessor's component, to help ensure consistency in the ratings among the various assessment endpoints.

Combining the risk and confidence ratings in one table for each assessment endpoint is a useful practice for risk communication purposes. For example, the hypothetical socioeconomic impact and risk characterization of a traditional OWT system (shown in Figure 2-2 and referred to as Scenario 2 in Chapter 6, *Socioeconomic Component Framework*) is summarized in Table 7-1. The conditional risk approach is described in the following sections.

Table 7-1
Hypothetical Summary of Estimated Socioeconomic Risks and Associated Uncertainties to a Property Owner with a Traditional OWT System[a]

Endpoint Property	Risks[b]	Confidence[c]	Comments
Economic Status	I	L	Costs might be marginally offset by increased property value, but repair costs and future property value are somewhat uncertain
Convenience	U	M	Restricted water uses and maintenance effort can be estimated with a moderate degree of confidence
Aesthetic Quality	A	H	Visual appearance of traditional systems are well known
Privacy	A	H	Intrusion factors for traditional systems are well known
Equity	U	M	Willingness to pay is variable among individuals and can be difficult to estimate

[a] Based on results of hypothetical socioeconomic component assessment, as described in Chapter 6, *Socioeconomic Component Framework* and Table 6-3

[b] "U" indicates results are consistent with risk to assessment endpoint (unacceptable)
"A" indicates results are not consistent with risk to assessment endpoint (acceptable)
"I" indicates results are too ambiguous to reliably estimate risk to assessment endpoint (indeterminate acceptability of the risk)

[c] Confidence in the risk rating is classified as being low (L), moderate (M), or high (H)

Conditional Risks

As noted previously, conditional risks are those for which the estimation of risk is dependent on the estimates for one or more other risks. Using the example summarized in Table 7-1, the risk of infection by a virus is conditional on the risk of having a treatment failure, in this case due to seasonal flooding of the WSAS, which then results in exposure of the residents to viruses.

Two potential methods for integrating the risks from multiple component assessments were discussed in Jones *et al.* (2001):

- Mathematically propagating risks estimated in each component
- Logically weighing the evidence of risks presented in each component

Mathematical propagation is only possible when quantitative estimates of risk are calculated for all of the assessment endpoints included in the conditional risk calculation. However, only the public health framework is designed to result in probability estimates (such as a 1:10,000 risk of infection) as a component of the current integrated framework. Therefore, mathematical propagation of risks cannot be used in this framework.

Characterization

Characterization of conditional risks in this integrated framework is 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, where a line of evidence is any model, test, or observation that can be used to estimate the magnitude or likelihood of risks (US EPA 1998b and Suter *et al.* 2000).

In this general risk characterization section, the component assessment for each assessment endpoint is treated as a line of evidence for the conditional risks associated with two or more assessment endpoints. The user must logically evaluate the likely interactions between each assessment endpoint to see how these interactions support or refute the theory that an OWT system poses a risk to a particular assessment endpoint. This evaluation is accomplished by weighing the evidence for conditional risks based on one or more of the following criteria, which are adapted from Suter *et al.* (2000):

- **Relevance**—Are potential impacts/risks to one endpoint relevant to the potential impacts/risks to the other endpoint being evaluated?
- **Exposure/Response**—Does an increase in the impacts/risks to one endpoint lead to an increase in the impacts/risks to the other endpoint being evaluated (for example, increasing likelihood and magnitude of virus discharges into drinking water supplies results in increased risk of infection of users of that water supply)?
- **Temporal Scope**—Do the component assessments of concern address important variations with time (such as depth to water table during the wet season or seasonal use of vacation homes)?

- **Spatial Scope**—Do the component assessments of concern consider the same spatial scale (such as both being based on micro-level evaluations)?
- **Quality**—Were the data available for the component assessments generated using appropriate quality assurance and control procedures (such as appropriate analytical procedures being used)?
- **Quantity**—How much information was available for the component assessments (such as number of treatment systems tested)?
- **Uncertainty**—How reliable was the information available for the component assessments in terms of estimating risks (such as estimated viral densities varying by several orders of magnitude)?

The evaluation of this evidence for conditional risks should be documented fully in the general risk characterization. This characterization of risks is more than a simple summary of previously characterized risks, which was the case for independent risks (as discussed previously). Instead, the evidence that supports or refutes the theory that interactions among two assessment endpoints lead to increased risks/impacts to one of those assessment endpoints should be discussed in detail.

A rating should be assigned in the general risk characterization section for each of the conditional risks being evaluated. This rating system should be compatible with the rating systems used to summarize the independent risks. A variation of the acceptable/unacceptable rating system discussed above is used here to rate the conditional risks. The decision rules for this system are:

- **U**—Indicates that estimated risks to one assessment endpoint are likely to result in unacceptable risks to the second assessment endpoint
- **A**—Indicates that estimated risks to one assessment endpoint are not likely to result in unacceptable risks to the second assessment endpoint
- **I**—Indicates that insufficient evidence was available to conclude whether or not the estimated risks to one assessment endpoint are likely to result in unacceptable risks to the second assessment endpoint (that is, the acceptability of the risk is indeterminate)

Uncertainties

Uncertainties for each assessment endpoint are discussed in detail in the component assessments. This section of the assessment should focus on the uncertainties (level of confidence) associated with estimating conditional risks. Sufficient detail should be provided to help decision makers understand the origin, magnitude, and tractability of these uncertainties. Tractability refers to the level of effort that would be required to substantially reduce these uncertainties (increase confidence).

Uncertainties associated with conditional risks need to be summarized in a manner consistent with that used for the independent risks to help ensure comparability among all risks and uncertainties. The ratings of low, moderate, and high confidence are, therefore, also recommended for the characterization of conditional risks. The issues and methods discussed for assigning confidence levels to the independent risk ratings also apply to the ratings for conditional risks.

Summary

The risk and confidence ratings for conditional risks can be presented in a summary table similar to that used for independent risks. However, including two sets of risk and confidence ratings requires changing the summary table to a risk matrix. The independent risks are listed by assessment endpoint across the top of the matrix (column headings). The assessment endpoints for which conditional risks are being summarized are listed in the leftmost column (row headings). The conditional risk ratings are listed in the body of the matrix at the intersection of each row and column combination. All of the calculations and assumptions used to estimate the conditional risks should be described in the accompanying text, because they are not adequately addressed in the component assessments.

For example, the hypothetical risks to the public health assessment endpoints that are conditional on the hypothetical risks of an emerging OWT (Type 4) system failure are summarized in Table 7-2. In this example, it is assumed that failure (dysfunction) of the UV disinfection unit poses serious (II) and major (I) risks due to discharges of fecal coliform (FC) and viruses (V), respectively (unmitigated risks). However, risk of failure is assumed to be only marginal (III) after mitigation controls are put in place (such as automated detection of UV unit dysfunction with immediate shut-down of effluent discharge and notification of operator).

Confidence in the unmitigated risks due to fecal coliform discharge is high, because pathogens associated with fecal coliform are assumed to always be present in the wastewater effluent. Confidence in the unmitigated risks due to virus discharge is low, because virus loads in the wastewater are highly variable. That is, the consequences would be severe if high concentrations of viruses were discharged, but it is difficult to predict virus concentrations in OWT systems.

Risks to onsite and offsite residents and visitors are assumed to have been calculated based on the drinking water pathway. The conditional, unmitigated risks listed in Table 7-2 for these endpoints (U) assume that the specified failure occurs causing exposure of the public health receptors (that is, assessment endpoint entities) to harmful concentrations of pathogens in their drinking water. Confidence in these conditional risk ratings is the same as the confidence level associated with the engineering risk ratings. The exception is the rating for offsite residents exposed to pathogens associated with fecal coliforms. The longer pathogen travel time required for exposure of offsite residents is assumed to decrease the exposure concentrations and increase the variability in those concentrations. Similarly, the risk from fecal coliforms to the recreationer is assumed to be low in this example where dilution is assumed to be high.

Conditional public health risks due to pathogens are assumed to be acceptable (A) when the mitigation controls are in place. Confidence in these risk ratings is high, because the detection and correction component have been thoroughly tested.

Only marginal engineering risks are assumed for system failures resulting in total nitrate (TN) concentrations that exceed the engineering assessment endpoint (Table 7-2). The conditional risk to public health due to nitrate toxicity is assumed to be acceptable. Confidence in those risk ratings increases as distance to the potential receptors increases.

Table 7-2
Summary of Hypothetical Risk and Confidence Ratings for Public Health Risks That Are Conditional on Risks of OWT System Failures[a]

	Engineering Risk and Confidence Ratings [b] (Unmitigated/Mitigated)		
Public Health Assessment Endpoints	**Total Nitrate (TN)**	**Fecal Coliform (FC)**	**Viruses (V)**
	III (m)/ NA	**II (h) / III (h)**	**I (l) / III (h)**
Infection of onsite resident or visitor	NA	U (h) / A (h)	U (l) / A (h)
Infection of offsite resident or visitor	NA	U (m) / A (h)	U (l) / A (h)
Infection of offsite recreationers	NA	A (l) / A (h)	A (m) / A (h)
Toxicity to onsite resident or visitor	A (l)	NA	NA
Toxicity to offsite resident or visitor	A (m)	NA	NA
Toxicity to offsite recreationers	A (h)	NA	NA

[a] Conditional public health risks are rated as:
U—Results are consistent with risk to assessment endpoint
A—Results are not consistent with risk to assessment endpoint
I—Results are too ambiguous to reliably estimate risk to assessment endpoint
NA = Not Applicable
Confidence in the risk rating is classified as being low (l), moderate (m), or high (h)

[b]Engineering risks are rated as
I = major II = serious III = marginal IV = negligible
as discussed in the engineering component framework
Ratings for risks based on the use of additional detection and correction capabilities (such as mitigated risks) are included if the unmitigated risks are major or serious.
Confidence in the risk rating is classified as being low (l), moderate (m), or high (h)

8 DATA GAPS

The process of developing a risk assessment framework often reveals or re-emphasizes inconsistencies between the types information that is needed and the types information that is actually available. Following is a list of some important data gaps with respect to the assessment of individual onsite wastewater treatment (OWT) systems.

Additional information that is needed to improve assessments of engineering risks for OWT systems include the following:

- Failure rates for OWT system components under a wide range of real-world conditions (as opposed to certification test results) over extended periods of operation
- System performance information that has been collected in a way that supports development of continuous failure rates
- Additional relationships between performance of wastewater soil absorption systems (WSAS) and changes in environmental conditions (such as seasonal changes in precipitation and in the separation from the water table)

The primary gaps in the data required for assessments of risks to public health from OWT systems are (or continue to be) the following:

- Dose/response information to support quantitative microbial risk calculations
- Viral dose/response models and rates of human infectivity
- Information on survival of viral particles in the environment
- Environmental fate and transport of microbial pathogens (environmental factors have a great influence on the transport and survivability of pathogens as well as control mechanisms that can be manipulated to impact pathogen survival)

Research needs for assessments of risks to ecological receptors from OWT systems include the following:

- Field studies of amphibians in wet soils, ponds, streams, and other areas around septic tanks versus control areas
- Studies to develop relationships between multiple stressors (nutrients, low oxygen, organic matter) and effects on various aquatic trophic levels
- Improved technologies for remote sensing of nutrients, phytoplankton, and sea grass area and condition

Data for a socioeconomic risk assessment should be as localized as is possible and cost-effective to obtain. This can mean that each socioeconomic risk assessment requires its own data generation to a larger degree than for the other component analyses, where data is often already available or can be extrapolated from values in the literature. However, generalized data can also be useful in socioeconomic risk assessments for OWT systems. Such data can increase understanding of the dynamics of socioeconomic issues in ways that assist qualitative assessment, or can provide comparables to assist quantitative assessment.

Data on the non-monetary costs and benefits of a wastewater system are most needed. While monetary issues are of profound importance, comparables are in many cases available. For instance, capital and operation and maintenance (O&M) monetary costs of OWT systems are relatively easily calculated. As another example, there is vast literature on the property value impacts of environmental factors; this literature can in many cases help the analyst make reasonable estimates for a specific decentralized wastewater risk assessment.

On the other hand, general data and even qualitative understandings are lacking for many non-monetary aspects of decentralized wastewater systems. These include:

- Foregone land uses due to OWT system components such as drainfields
  - How often or under what conditions are desired uses of portions of a property foregone?
  - What desired uses are foregone?
  - Is an economic loss perceived?
- Regulatory compliance costs
  - How much time does it take system owners to comply with regulations for different types of OWT systems?
  - How much time does it take system owners to comply with different types of regulations?
- Value of privacy and costs of intrusions
  - Under what conditions do system owners consider inspection or maintenance services to be welcomed, accepted, or resented?
  - How do these perceptions vary with income, community character, and other factors?
- Restrictions on use (such as inabilities to use of garbage disposals, water softeners, or other appliances)
  - Under what conditions do homeowners consider these restrictions problems or not problems?
  - How extensive is the problem?
  - What is the economic cost of various restrictions—objectively and subjectively?

- Perceptions of uncertainty over system performance
  - To what degree do system owners' perceptions of the chances of failure track with actual reliability data?
  - What do they see as the costs of failure?
  - How do they value those costs, or value a more reliable system?
- Perceived inequities
  - Under what conditions do system owners perceive that the costs or benefits of wastewater treatment systems are unfair?
  - How do variations in income or other socioeconomic characteristics affect these perceptions?
- Aesthetic issues
  - How do system owners and neighbors perceive the aesthetic qualities of various features or results of OWT systems?
  - How do these perceptions vary according to lot size, income, and other factors?
  - What is the economic benefit of various measures (screening, landscape design, or other measures) to mitigate aesthetic problems?

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10 LIST OF ACRONYMS AND ABBREVIATIONS

µg	micrograms
µM	micromolar
ATU	Aerobic treatment unit
BOD	Biochemical oxygen demand
CBOD	Carbonaceous biochemical oxygen demand
CFU	Colony forming unit
DET	Detection
DQO	Data quality objectives
FC	Fecal coliform
FMEA	Failure modes and effects analysis
g	grams
ha	hectare
L	Liter
MCLG	Maximum contaminant level goal
m	Meter
mg	Milligram
ml	Milliliter
mmole	Millimole
N	Nitrogen
NBOD	Nitrogenous biochemical oxygen demand
OCC	Occurrence
OWT	Onsite wastewater treatment
P	Phosphorus
PFU	Plaque forming unit
PMB	Porous media biofilter

RfD	Reference Dose
RT-PCR	Reverse transcriptase-polymerase chain reaction
SEV	Severity
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UV	Ultraviolet
V	Virus
WSAS	Wastewater soil absorption system
yr	Year

A GLOSSARY

This section contains definitions of terms used throughout this framework. Where more than one definition is possible, the context for each usage is identified.

Aesthetics—Visual, auditory, or olfactory attributes of an environment that affect the perceived quality or value of that environment (for example, the aesthetics of a residential property can be degraded by the sight and smell of raw sewage on the soil surface).

Assessment endpoint—An explicit expression of a value that is to be protected, consisting of an entity, a property of that entity that can be measured or estimated, and, to the extent practical, a level of effect on that property that constitutes an unacceptable risk.

Assessment endpoint entity—A receptor that is selected for evaluation as part of a risk assessment process.

Attribute—A property or characteristic of a living or non-living thing. In this framework, it is typically used in reference to a measurable property of a receptor, treatment system, or receiving environment.

Bioassay—Test of the effect (such as mortality or growth inhibition) of a nutrient or other chemical on organisms in a laboratory or mesocosm.

Biochemical Oxygen Demand (BOD)—The amount of oxygen that would be consumed if all the organics in one liter of water were oxidized by bacteria and protozoa. BOD is a measure of the potential for an effluent to reduce the dissolved oxygen in a receiving surface water body. (Also see Carbonaceous and Nitrogenous BOD.)

Carbonaceous BOD—Biochemical oxygen demand from carbon compounds only.

Component—See system component.

Component framework—One of the four discipline-specific risk assessment frameworks included in this general framework (the engineering, public health, ecological, and socioeconomic frameworks).

Conceptual model—The expected relationships among the stressors, exposure pathways, and receptors (assessment endpoint entities) that are depicted visually and described as part of the problem formulation. The general framework and the component frameworks each include at least one conceptual model.

Contemporary systems—Traditional OWT systems that have been modified to enhance performance under less-than-favorable site conditions (such as low permeability, high groundwater levels, and other less favorable conditions).

Discharge point—The point at which wastewater effluent is released into the environment (for example, immediately beneath a drainfield or at the end of a straight pipe).

Dysfunction—Performance of an OWT system or system component that is below the expected level of performance.

Effect—A change in the state or dynamics of a receptor.

Emerging systems—OWT systems that include additional stages or technologies to enhance treatment beyond what can be achieved at a particular site with traditional or contemporary treatment systems. They are designed to require little or no soil, and generally include some type of disinfection process.

Exposure—The co-occurrence of a stressor and a receptor.

Exposure pathway—The physical route by which a nutrient or other stressor moves from a source to a biological receptor. A pathway may include transformation of the chemical.

Exposure point—The point at which a receptor is assumed to be exposed to the wastewater effluent or its constituents (for example, a potentially contaminated drinking water supply well).

Failure—Complete dysfunction of an OWT system or system component.

First-level cause—The immediate cause of the failure mode that will directly make the failure mode occur; whereas a root cause is below the first-level cause.

Hazard quotient—The ratio of an exposure concentration divided by a threshold for effects concentration (such as an RfD). Values greater than 1.0 indicate that the threshold level for effects has been exceeded by the exposure concentration.

Household chemical constituents—Chemical components of products that are commonly used in residential dwellings (such as cleaners, paint, and medicines) that may be disposed of via an OWT system. Examples include volatile organic compounds, endocrine disrupters, and antibiotics.

Incremental risk—Risk above the magnitude associated with the background level of a stressor or above another designated level.

Input variable—A variable used within a framework or model (such as the loading rate for an OWT system).

Intermediate stressor—Any physical, chemical, or biological entity that serves as a link between a secondary stressor and a receptor (for example, excessive growth of aquatic macrophytes due to nutrient loading from an OWT system can be an intermediate stressor for fish).

LC50—Lethal concentration for fifty percent of organisms tested.

Level of effect—A measure of the degree to which a receptor responds to a stressor.

Maintenance—The actions required to keep an OWT system or system component operating at or above the expected level of performance.

Macro-scale—The spatial scale that contains many individual onsite wastewater treatment systems, as well as other point and nonpoint sources of pollution.

Management—The process of operating and maintaining an OWT system or system component, either directly or through a designee.

Measure of ecosystem characteristics—A measure of an environmental attribute that influences the distribution of a stressor (such as soil temperature and depth to groundwater).

Measure of effect—A measurable change in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it was exposed.

Measure of exposure—A measure of stressor existence and movement in the environment and its contact or co-occurrence with the assessment endpoint.

Measures of receptor characteristics—A measure of a receptor attribute that influences exposure and response (such as age and behaviors).

Micro-scale—the spatial scale represented by an individual residential lot with an OWT system.

Nitrogenous BOD—Biochemical oxygen demand from the mineralization of ammonia.

Nominal performance—The expected level of performance of an OWT system or system component.

Non-performance—Any degree of functioning of an OWT system or system component that is less than the expected level of performance (dysfunction).

Outdated systems—OWT systems that are no longer considered adequate (such as cesspools and straight pipes) even at sites with favorable conditions for onsite treatment. They provide little or no treatment prior to discharge to the receiving environment.

Output variable—A variable estimated by a using a framework or model (such as the likelihood of an adverse effect on a receptor).

Parameter—A variable characteristic that can be used to describe the state or condition of an OWT system, system component, or environment.

Parameterized—The process of assigning a value or range of values to a parameter or variable.

Pathogens—Biological entities capable of causing illness in other organisms (such as bacteria, protozoa, and viruses).

Performance—The functioning of an OWT system or system component with respect to its ability to modify the attributes of wastewater.

Primary stressor—Any physical, chemical, or biological entity that can directly affect a receptor in an adverse way.

Property—A characteristic of a living or non-living entity. (Also see attribute.)

Prospective risk assessment—Risk assessment intended to evaluate risks under future conditions.

Receptor—An entity that is potentially exposed to one or more stressors. A receptor can refer to an organism, group of organisms, or a social or political construct. Some or all receptors may be selected as assessment endpoint entities.

Reference conditions—Characteristics of exposure or effects at a site that is not impacted by the source of interest but may be impacted by other sources of the type of stressor that is the subject of the assessment.

Retrofit—An OWT system that has been repaired by installing one or more new system components.

Retrospective risk assessment—A risk assessment intended to evaluate risks associated with past conditions.

Risk manager—An individual or institution with the authority to decide what actions will be taken in response to a risk.

Root cause—The underlying cause (second-level cause) of a failure mode. The root cause ultimately results in a first-level cause, which is the immediate cause of the failure mode.

Screening-level risk assessment—An assessment performed to determine potential risk or the scope of a definitive assessment by eliminating from further consideration chemicals and receptors that are clearly not associated with a potential risk.

Secondary stressor—Any physical, chemical, or biological entity that can indirectly affect a receptor in an adverse way (such as nutrients discharged to surface water can be secondary stressors for fish by increasing the growth of aquatic macrophytes).

Sensitivity—A measure of how readily a receptor responds to a particular stressor.

Source—The facility in which wastewater is initially generated. The default source in this framework is a single-family residence.

Straight pipe—An outdated OWT system that is assumed to provide little, if any, treatment prior to discharging the wastewater to the environment.

Stressor—Any physical, chemical, or biological entity that can induce an adverse response in a receptor, either directly or indirectly.

Surface breakthrough—The upwelling of untreated or partially-treated wastewater through the soil surface due to dysfunction of an OWT.

Susceptibility—A measure of the exposure and sensitivity of the receptor to a particular stressor.

System component—A manufactured or natural structure that performs a particular function as part of an OWT system [such as a septic tank, drainfield, or wastewater soil absorption system WSAS)].

System type—A classification used to distinguish general classes of OWT systems for purposes of framework development and for selecting example treatment trains for this framework. The terms selected for the four system types in this framework are:

- Outdated
- Traditional
- Contemporary
- Emerging

Traditional systems—OWT systems that are typically installed at sites with favorable conditions for onsite treatment (for example, a septic tank with a drainfield that discharges by infiltration/percolation through 60 cm to 120 cm of unsaturated soil).

Treatment train—One or more treatment components linked together in series to create an OWT system.

Value to be protected—Any entity, process, or concept about which risk management decisions will be made.

Variable—A characteristic with more than one value that can be used to describe the state or condition of an OWT system, system component, or environment.

Weight-of-evidence—(1) A type of analysis that considers all available evidence and reaches a conclusion based on the amount and quality of evidence supporting each alternative conclusion; (2) the result of a weight-of-evidence analysis, often presented in a table.

B SUPPORTING INFORMATION: GENERAL PROBLEM FORMULATION

This appendix provides reference values and assumptions used to select the assumed effluent quality characteristics, the specific treatment components selected for the example treatment trains, and the key factors considered in the selection process. This information is provided in the following tables:

- **Table B-1**—Effluent Quality for the Example Systems to be Included in This Framework
- **Table B-**—Assumed Effluent Quality Characteristics for the Wastewater Soil Absorption Systems (WSAS) in the Example Traditional and Contemporary Treatment Systems
- **Table B-3**—Treatment Components to be Used for Example Systems Included in This Framework

Table B-1
Effluent Quality for the Example Systems to be Included in This Framework

Example System Categories[a]		Example Effluent Quality[b]						Referenced Effluent Quality (BOD:TSS:TN:TP:FC)		
No.	**Name**	**BOD**	**TSS**	**TN**	**TP**	**FC**	**V**	**SWRCB[c] System/Ratings**	**NOWRA[d] Conc. Cat.**	**M&E Subcommittee[e]**
1	Straight Pipe	350	350	70	10	10^7	0–10^5	Untreated (450:503:70.4:17.3:10^6)	Untreated 350: 350:90:35: 10^6	Untreated 350:350:70:10:10^7
2	Traditional	250	150	60	9	10^6	0–10^5	Septic Tank (185:83:70:16:10^6)	Septic Tank 200:200:90:35: 10^5	Septic Tank 250:150:60:9:10^5
3	Contemporary	30	30	20	8	10^4	0–10^5	C:C:C:C:C (30:30:30:10:2000)	C:C:C:C:C 30:30:30:10:200	C:C:C:C:C 30:30:20:8:10^4
4	Emerging	10	10	10	6	10	0–10	A:A:B:B:A (10:10:20:5:2)	A:A:B:B:A 10:10:10:5:10	A:A:B:B:A 10:10:10:6:10

Note: The values in Table B-1 are not intended to be considered as research-proven values, due to the high variability of soils and design parameters.

[a] Proposed categories are intended to broadly represent the range of OWT systems in the US. Specific treatment components were selected as representative of each OWT system category.

[b] The proposed effluent quality characteristics were used to select specific components to be included in the example treatment systems.
BOD = 5-day biochemical oxygen demand (mg/L); for systems 3 and 4 CBOD
TSS = total suspended solids (mg/L)
TN = total nitrogen (mg/L)
TP = total phosphorus (mg/L)
FC = fecal coliform (colony forming units/100 ml)
V = virus (plaque forming units/ml)
Values are from Siegrist *et al.* (2001) and represent episodically high loading to the system.

[c] The California State Water Resources Control Board (SWRCB) funded the Review of Technologies for Onsite Treatment of Wastewater in California (Leverenz *et al.* 2002), which included system designations and effluent quality ratings: **Untreated**—Typical (average concentrations) residential wastewater prior to any onsite treatment; **Septic tank**—Typical (average concentrations) residential septic tank effluent, without effluent filter; **Ratings**—Classification system with values (A, B, C, and D) that are based on values obtained from operational systems, independent certifications, and experimental systems.

[d] The National Onsite Wastewater Recycling Association (NOWRA) is developing a classification matrix based on the expected concentrations of selected wastewater constituents in the effluent from OWT system components. Concentrations that represent the full range of expected values from untreated wastewater to the highest treatment level were divided into ordinal categories (for example, <200 mg/l, < 30 mg/L, <20 mg/L, and so on). Only a subset of the NOWRA concentration categories are used in Table B-1.

[e] These effluent quality characteristics were recommended by the NDWRCDP Management and Economics Subcommittee.

Table B-2
Assumed Effluent Quality Characteristics for the WSAS in the Example Traditional and Contemporary Treatment Systems

System Type[a]	WSAS Description[b]	Value Description[c]	Example Concentrations and Values[d]					
			BOD	TSS	TN	TP	FC	Virus
#2	3 ft. depth 280 ft.^{2} area	Influent	250	150	60	9	10^6	0–10^5
		Reduction	90%	90%	10%	50%	99.99%	99.9%
		Effluent	25	15	54	4.5	10^2	0–10^2
#3	18 in. depth 200 ft.^{2} area	Influent	30	30	20	8	10^4	0–10^5
		Reduction	80%	90%	10%	20%	90%	99%
		Effluent	6	3	18	6.4	10^3	0–10^3

Note: The values in Table B- are not intended to be considered as research-proven values, due to the high variability of soils and design parameters. The assumed percent reduction values may be excessively conservative in many locations. These values are intended for example purposes only.

[a] Example OWT system categories are described in Table 2-1. System type #2: Traditional and system type #3: Contemporary

[b] The assumed surface area of the WSAS component for the example traditional system (Type #2) was based on an assumed loading rate of 280 gallons/day and an application rate of 1 gallon per square foot/day. The area of the contemporary WSAS (Type #3) was assumed to be approximately 75% of the area for the Type #2. The depth (thickness) of the Type #3 WSAS was assumed to be half that of the Type #2.

[c] Influent concentrations are the example effluent quality values listed in Table 2-2 for the associated OWT category (Type #2 or #3). Reduction is the percentage of the influent constituent that is assumed to be removed by the WSAS. For Type #2, the reduction values are from Siegrist *et al.* (2001). The reduction values for Type #3 are based on an assumed effectiveness of the smaller WSAS relative to the effectiveness of the Type #2 WSAS. Specifically, WSAS #3 is assumed to be approximately 90% as effective as WSAS #2 for removal of BOD, TN, and FC; approximately 40% as effective as WSAS #2 for removal of TP; and approximately 100% as effective as WSAS #2 for removal of TSS and viruses. Estimated values were rounded to the nearest multiple of 10.

[d] BOD = 5-day biochemical oxygen demand (mg/L); for system 3 CBOD TSS = total suspended solids (mg/L) TN = total nitrogen (mg/L)
TP = total phosphorus (mg/L) FC = fecal coliform (colony forming units/100 ml) V = virus (plaque forming units/ml)

Table B-3
Treatment Components to be Used for Example Systems Included in This Framework

		Major System Components			
Component Type	**System Category**	**Example Components**[a]	**Effluent Quality**[b]	**Prevalence**[c]	**Availability of Data**[d]
Septic tank	2–4	Concrete tank	Meets all criteria (assumed)	Ubiquitous	High
PMB	4	Sand filter (or textile filter)	Meets all except FC	Common	High
ATU (NSF/ANSI STD 40)	3	Various proprietary systems (Whitewater or Nayadic)	Meets all	Common	High
Disinfection Unit	4	UV Radiation	Meets FC	Not Common	Low/Medium
WSAS, gravity-fed	2	Drain field	Not rated	Ubiquitous	High
WSAS (reduced sizing)	3	Pressure dosed trench system	Not rated	Less common	Medium/High

Source: Adapted from Table ES-3 and text in Leverenz *et al.* (2002)

[a] Leverenz *et al.* 2002

[b] Effluent quality based primarily on ratings in Table ES-3 (located in Laverenz *et al*. 2002) and, to a lesser extent, the referenced section of the text.

[c] Prevalence of use is among all OWT systems using this type of component rather than among all OWT systems in general.

[d] Data availability refers to the amount and quality of data regarding component performance and reliability.

C SUPPORTING INFORMATION: ENGINEERING

This appendix provides supporting information for the engineering component framework.

Probability of Failure Formulas

An assumption could be made that the probability of a 100-year flood occurring at least once over a 20-year onsite wastewater treatment (OWT) system lifetime would be 20 % 0.01 = 0.20. However, a more accurate probability is given by Vesely *et al.* (1991) as:

$Q = 1 – (1 – p)n$ **(Eq. C-1)**

Where:

Q = the probability of occurrence over an extended number of years

p = annual probability of occurrence

n = number of years

Thus, the probability of a 100-year flood occurring at least once over a 20-year OWT system lifetime would be:

$Q = 1 – (1 – 0.01)20 = 0.18$ **(Eq. C-2)**

Therefore, if a 100-year flood is the minimum size flood expected to fail the system (or subsystem), then the example occurrence (OCC) rank from Table 3-4 would be moderate (that is, approximately 5).

For this and many other similar events, it is worth noting that the simplified (for example, 20 × 0.01 = 0.20) will provide a sufficiently accurate likelihood estimate for purposes of assigning an OCC ranking. That is, both methods yield an OCC ranking of moderate (5) for the example of the 100-year flood occurring at least once over a 20-year OWT system lifetime.

Many of the failure (dysfunction) events associated with OWT systems are best described as being independent component failures. The failure probabilities for these events are based on the component failure rates. The failure rate, λ (*t*), is the probability of failure during time *t* to *t* + *dt*, given no failures before *t*. The time-dependent system failure rate describes the density of time to first system failure, and consequently, the distribution of time to first system failure.

The probability that the system has experienced one or more failures from time $t = 0$ to time t (its unavailability), for the non-repairable case (maintenance is not performed), is given by Fussell (1975):

Unavailability = $\bar{a}_i = 1 - e^{-\lambda i\ t}$ **(Eq. C-3)**

When $\lambda\ t \leq 0.1$, this can be approximated by (Fussell 1975)

$\bar{a}_i = \lambda_i t$ **(Eq. C-4)**

For the repairable case where $\lambda\ t$ is the mean down time for repair (Fussell 1975),

$\bar{a}_i = \lambda_i t_i /(1 + \lambda_i t_i)\ [1 - e^{-(\lambda i + 1/\ ti)t}\ ]$ **(Eq. C-5)**

Because FMEA analyses typically concentrate on single failures (the failure of a single component causes the system or subsystem to fail), $\lambda\ t$ is also the expected number of failures (ENF). Thus

$\mathrm{ENF}_i = \lambda_i t$ **(Eq. C-6)**

for both repairable and non-repairable systems (Fussell 1975).

D SUPPORTING INFORMATION: PUBLIC HEALTH

This appendix provides supporting information relevant to the public health component framework.

Viruses

Numerous gastroenteritis outbreaks within the US and worldwide have been linked to enteric viruses such as rotavirus, enterovirus, and adenoviruses associated with exposure to water.

As noted by Sobsey *et al*. in 1995, recovery and detection of enteric viruses in soil and water is a technological challenge, time-consuming, and expensive. However, detection and quantification of viruses in environmental samples is the preferred means for assessing public health risks from these pathogens.

Recent advances in molecular and genetic analytical techniques for water samples have produced improved monitoring results. The reverse transcriptase-polymerase chain reaction (RT-PCR) technique is one example (Grabow *et al.* 2001). RT-PCR is a gene-probe method that amplifies and recognizes the nucleic acids of target viruses. The enteric viruses detected by use of this method include enterovirus, reovirus, rotavirus, Hepatitis-A virus, and Norwalk virus. In addition to direct measurements of enteric viruses, two main groups of coliphage, which infect and replicate in coliform bacteria, are used as viral indicators. Somatic coliphage infect coliform bacteria by attachment to the outer cell membrane or cell wall. They are widely distributed in both fecal-contaminated and uncontaminated waters. Male-specific coliphage attach only to the F-pilus of coliforms that carry the F+ plasmid. F-pili are made only by bacteria grown at higher temperatures. Thus, male-specific coliphage presumably come from warm-blooded animals or wastewater.

Exposure Factors and Models

Table D-1 provides default factors and models (formulas) for estimating human exposures to contaminants in ambient media (such as water, soil, or groundwater). Site-specific factors and models be are highly recommended for use whenever possible.

Table D-1
Exposure Factors for Assessing Residential Daily Intakes Via Ingestion of Soil[a]

Ingestion of Soil Pathway		
Chronic daily intake carcinogen (mg/kg-d) = $[(CS \times FI \times EF)/AT] \times [(ED_c \times IR_c/BW_c) + (ED_a \times IR_a/BW_a)]$ Chronic daily intake noncarcinogen (mg/kg-d) = $[(CS \times FI \times EF \times IR_n \times ED_n) / (BW_n \times AT_n)]$		
Variable	**Value Used**	**Explanation/Source**
CS = Concentration in soil or sediment	Chemical-specific mg/kg	Concentration is derived from sample data
IR_c = Ingestion rate child	0.0002 kg/day	US EPA 1991b
IR_a = Ingestion rate adult	0.0001 kg/day	US EPA 1991b
CF = Conversion factor	10^3 g/kg	Necessary to convert to appropriate units
FI = Fraction ingested	1 (unitless)	Maximum value used; equivalent to 100%
EF = Exposure frequency	350 days/year	OSWER Directive (US EPA 1991a, 1991b)
ED_c = Exposure duration child ED_a = Exposure duration adult	6 years 24 years	Two parts (child and adults) exposure for a 30-year duration (OSWER Directive, US EPA 1991b)
BW_c = Body weight child	15 kg	Child (OSWER Directive, US EPA 1991b)
BW_a = Body weight adult	70 kg	Adult (OSWER Directive, US EPA 1991a)
AT = Averaging time	365 days/year $ED_{c \text{ or } a}$	Averaging time for noncarcinogens (US EPA 1989a, 1991b)
	365 days/year 70 years	Averaging time for carcinogens (US EPA 1989a, 1991b)
AT_n = Averaging time	365 days/year ED	Averaging time for noncarcinogens (US EPA 1989a, 1991b)
IR_n = Ingestion rate	0.0002 or 0.0001 kg/day	Child or adult rate (US EPA 1991b)
ED_n = Exposure duration	6 or 24	Child or adult (OSWER Directive, US EPA 1991b)
BW_n = Body weight	15 or 70 kg	Child or adult (OSWER Directive, US EPA 1991b)

[a] Parameter subscripts: “a” is for adult, “c” is for child, and “n” is for neither.

Table D-2
Exposure Factors for Estimating Residential Daily Intakes Via Ingestion of Groundwater

Ingestion of Groundwater Pathway		
Chronic daily intake (mg/kg-d) = [CW x IR x EF x ED / (AT x BW)]		
Variable	**Value Used**	**Explanation/Source**
CW = Concentration in water	Chemical-specific (mg/L)	Concentration is obtained from sample data or modeled
IR = Ingestion rate	2 L/day adult 1 L/day child	US EPA 1989a; OSWER Directive (US EPA 1991b)
EF = Exposure frequency	350 days/year	OSWER Directive (US EPA 1991b)
ED_C = Exposure duration	30 years	Residential exposure for a 30-year duration (OSWER Directive, US EPA 1991b)
BW = Body weight	70 kg	Adult (US EPA 1991b)
AT = Averaging time	365 days/year ED	Averaging time for noncarcinogens (US EPA 1989a, 1991b)
	365 days/year 70 years	Averaging time for carcinogens (US EPA 1989a, 1991b)

Table D-3
Exposure Factors for Estimating Daily Intakes From Ingestion of Water During Recreational Activity

Ingestion of Surface Water During Recreational Activity		
Chronic daily intake (mg/kg-d) = [CW x IR x EF x ET x ED / (AT x BW)]		
Variable	**Value Used**	**Explanation/Source**
CW = Concentration in water	Chemical-specific (mg/L)	Concentration is obtained from sample data or modeled
IR = Ingestion rate	0.05 L/hour	Region IV Supplemental Guidance to RAGS (US EPA 1995)
ED = Exposure duration	30 years	Residential exposure for a 30-year duration (OSWER Directive, US EPA 1991b)
ET = Exposure time	1 hour/day	US EPA 1991b
BW = Body weight	70 kg	Adult (US EPA 1991b)

Table D-3
Exposure Factors for Estimating Daily Intakes From Ingestion of Water During Recreational Activity (Cont.)

Ingestion of Surface Water during Recreational Activity		
Chronic daily intake (mg/kg-d) = [CW x IR x EF x ET x ED / (AT x BW)]		
Variable	**Value Used**	**Explanation/Source**
AT = Averaging time	365 days/year ED	Averaging time for noncarcinogens (US EPA 1989a, 1991b)
	365 days/year 70 years	Averaging time for carcinogens (US EPA 1989a, 1991b)

Table D-4
Exposure Factors for Estimating Dermal Contact With Water During Recreational Activity

Dermal Contact With Surface Water During Recreational Activity		
Chronic daily intake (mg/kg-d) = [CW x CF x PC x SA x ET x EF x ED / (AT x BW)]		
Variable	**Value Used**	**Explanation/Source**
CW = Concentration in water	Chemical-specific (mg/L)	Concentration is obtained from sample data or modeled
CF = Conversion factors	m/100 cm x 1000 L/m^3	Necessary to convert to appropriate units
SA = Available surface area	1.94 m^2	Average total body surface area for an adult (Dermal Exposure Assessment, US EPA 1992)
PC = Permeability constant	Chemical-specific (cm/hour)	Dermal Exposure Assessment (US EPA 1992)
ET = Exposure time	1 hour/day	Dermal Exposure Assessment (US EPA 1992)
EF = Exposure frequency	45 days/year	OSWER Directive (US EPA 1995)
ED = Exposure duration	30 years	Residential exposure for a 30-year duration (OSWER Directive, US EPA 1991b)
BW = Body weight	70 kg	Adult (US EPA 1991b)
AT = Averaging time	365 days/year ED	Averaging time for noncarcinogens (US EPA 1989a, 1991b)
	365 days/year 70 years	Averaging time for carcinogens (US EPA 1989a, 1991b)

Example Quantitative Microbial Risk Calculations

For the microbial risk endpoint, the fecal coliform intake estimated for the adult onsite residential ingestion of soil scenario is 685 cfu and the rotavirus intake is 0.55 pfu (Table 4-3). Using the beta-Poisson model for *E. coli* and rotavirus, the estimated risk of infection is 7.2 × 10-5 for *E. coli* and 1.9 × 10-1 for rotavirus. The rotavirus estimate exceeds the guideline risk of infection level of 1.0 × 10-4 (Table 4-3).

Bacterial Risk Calculation *(E. coli)*

An example of how to determine exposure concentration (intake) using equations as provided in the Example Quantitative Microbial Risk Calculations section is provided here. This example highlights exposure calculations for an adult soil ingestion scenario (see exposure parameters and equations in Table D-1). Sampling indicates the onsite soil contains *E. coli* at a measured concentration of 5×10^5 cfu/g (colony forming unit/gram).

Equation and parameters from Table D-1:

$[(CS \times FI \times EF)/AT] \times ED \times (IR/BW)] = \text{cfu}$

$[(5 \times 10^5\ \text{cfu/g}) \times 1 \times 350\text{d/yr})/(365\text{d}/70\ \text{yrs}) \times 24\ \text{yrs} \times (0.000\ 1\text{kg/d}\ /70\ \text{kg})] = 685\ \text{cfu}$

The risk of infection to an adult from exposure to soil containing *E. coli* is estimated using the exposure concentration of 685 cfu and the beta-Poisson dose/response model for *E. coli*.

$P = 1-(1+N/B))^{-\alpha}$

P = probability of infection

N = exposure concentration (685 cfu)

$B = 1.61 \times 10^6$

$\alpha = 0.1705$

$P = 1-(1+(685/1.61 \times 10^6))^{-0.1705}$

P (risk of infection) = 7.2×10^{-5}

Viral Risk Calculation (*Rotavirus*)

An example of how to determine exposure concentration (intake) using equations as provided in the Example Quantitative Microbial Risk Calculations section is provided here. This example highlights exposure calculations for an adult soil ingestion scenario (see exposure parameters and equations in Table D-1). Sampling indicates the onsite soil contains rotavirus at a measured concentration of 400 pfu/g (plaque forming unit/gram).

Equation and parameters from Table D-1:

$[(CS \times FI \times EF)/AT] \times ED \times (IR/BW)]$= pfu

$[(400\ \text{pfu/g}) \times 1 \times 350\text{d/yr})/(365\text{d}/70\ \text{yrs}) \times 24\ \text{yrs} \times (0.0001\ \text{kg/d}\ /70\ \text{kg})] = 0.55$ pfu

The risk of infection to an adult from exposure to soil containing rotavirus is estimated using the exposure concentration of 0.55 pfu and the beta-Poisson dose/response model for rotavirus.

$P = 1-(1+N/B))^{-\alpha}$

P = probability of infection

N = exposure concentration (0.55 pfu)

B = 0.42

$\alpha = 0.26$

$P = 1-(1+(0.55/0.42))^{-0.26}$

P (risk of infection) = 1.9×10^{-1}

For each pathogen and exposure pathway of concern, exposure concentrations and risk of infection estimates are calculated. The individual risk of infection estimates are summed to obtain a total risk of infection from exposure to all pathogens of concern. The examples in this section only focus on one bacterial species and one virus.

Public Health Risk Assessment Framework for Carcinogenic Endpoints

As a supplement to Chapter 4, *Public Health Component Framework*, additional information is provided to assess carcinogenic endpoints from exposure to wastewater chemicals of concern. Carcinogenic endpoints are addressed when the wastewater chemical of concern has been classified as a carcinogen. Chemicals that are carcinogens can enter wastewater and OWT systems as the result of their use in the household. Household cleaners and disinfectants contain organic compounds and chlorinated compounds, some of which are carcinogens. These chemicals can enter wastewater and subsequently OWT systems during typical household use.

As stated in Chapter 4, *Public Health Component Framework*, the human health properties evaluated as the result of exposure to chemicals originating in wastewater are systemic toxicity and/or cancer as defined in US EPA's Risk Assessment Guidance for Superfund (US EPA 1989a). In addition to the systemic toxicity (noncarcinogenic) endpoint described in Chapter 4, carcinogenicity is a second human health property evaluated from exposure to chemicals of concern.

Doses for carcinogenic chemicals are estimated in the same manner as for systemic toxins; however, the dose is multiplied by a cancer potency factor to estimate excess cancer risk. Chemical-specific cancer potency factors (or slope factors) can be obtained from US EPA or the Risk Assessment Information System (RAIS 2003). The resultant is a risk value defined as an increased incidence of cancer such as 1/10,000 (1×10^{-4}) from exposure to a carcinogenic chemical.

The steps outlined in Chapter 4, *Public Health Component Framework*, the Exposure Pathways and Exposure Points section for evaluating exposure pathways and exposure points and the Exposure Concentrations, Routes of Exposure, and Exposed Populations section, for evaluating exposure concentrations, routes of exposure and exposed populations are directly applicable for assessing carcinogenic chemicals. The examples for nitrate provide the guidance for assessing additional chemicals of concern, whether they are noncarcinogens or carcinogens. The development of the conceptual site model, fate and transport modeling, and exposure modeling are identical to those described for nitrate. However, additional exposure pathways may be relevant for chemicals such as inhalation and dermal absorption when compared to the normal wastewater chemicals of concern such as nitrate. The US EPA (1989a, 1995) and the RAIS (2003) provide details on additional exposure modeling for carcinogenic chemicals.

Once exposure concentrations are determined, doses for carcinogens are estimated in the same manner as noncarcinogens (see Chapter 4, *Public Health Component Framework*, the Exposure Pathways and Exposure Points section) and quantification of cancer risk is determined by multiplying the intake of a carcinogen by the cancer slope factor thus producing an estimate of excess cancer risk. For chemicals with the potential to cause cancer, risks greater than 1×10^{-4} are generally regarded as unacceptable and risk mitigation options are considered by risk managers and/or decision makers.

E SUPPORTING INFORMATION: ECOLOGICAL

This appendix provides additional supporting information for the ecological component framework, particularly information relevant to the macro-scale of risk assessment.

Problem Formulation Issues

Additional information that may help the user develop a problem formulation for the ecological risk assessment is provided in this section.

Identifying Potential Stressors

The nutrients nitrogen and phosphorus are the principal ecological stressors associated with residential OWT systems. Nutrient inputs to a surface water body have the greatest impact if background concentrations limit production or growth rates (primary production). In general, nitrogen is a limiting nutrient in estuarine waters in temperate environments, and phosphorus is a limiting nutrient in most fresh waters in temperate environments. However, the user must identify stressors based on site-specific conditions. The information in this section may be useful for that identification effort.

Most nitrogen is expected to enter surface water as nitrate, because oxidation of ammonia, nitrite, and organic forms of nitrogen usually occurs rapidly and nitrate is the most stable form of nitrogen in surface waters. Because of their chemical instability, nitrite and ammonia are generally only significant stressors when

- They are released in large quantities from major point sources such as industrial effluents, livestock feed lots, or urban centers that do not have denitrification systems (Rouse *et al.* 1999)
- An OWT system is located in wet soils or forested watersheds (Valiela *et al.* 2000)
- The surface waters in which oxidized forms of nitrogen are released are anoxic or hypoxic

Reduced forms of nitrogen are not discussed in detail in this framework.

Most of the phosphorus released in wastewater effluent (about 85 percent) is in the form of soluble orthophosphate, with the rest as organic and inorganic phosphorus in suspended solids (Gold and Sims 2001). If the phosphorus travels a distance through soil before reaching a surface water body, a substantial fraction may precipitate with aluminum, iron, or calcium or sorb to clay particles.

Nitrogen is usually limiting to primary production in environments where the ratio of inorganic nitrogen to phosphorus is below the Redfield ratio of 16 moles of nitrogen to one mole of phosphorus (Redfield 1958 and Howarth 1988). This is common in estuarine environments. However, phosphorus may also limit primary production in some marine ecosystems (Howarth 1988), including, for example, part of the Indian River Lagoon in Florida (Phlips *et al.* 2002).

More specifically, three factors determine whether nitrogen or phosphorus is more limiting:

- The ratio of nitrogen to phosphorus in surface water inputs
- The preferential removal of nitrogen or phosphorus from the photic zone because of
  - Denitrification
  - Preferential sedimentation of nitrogen in zooplankton fecal pellets
  - Adsorption of phosphorus
  - Other biogeochemical processes
- The extent to which nitrogen fixation balances other deficits in nitrogen availability (Howarth 1988)

Nutrients may not act directly on all potential ecological receptors. Nutrients may create algal blooms, which may cause light or oxygen limitations to macrophyte, fish, or benthic communities. Light limitation, oxygen limitation, changes in populations that affect other populations (by predation, forage availability), or changes in communities that affect habitat are referred to as secondary stressors because they are not produced directly by wastewater effluent, but rather as a consequence of excess nutrients. These phenomena are primarily relevant for macro-level (such as watershed scale) assessments, rather than in the micro-level assessments that are the focus of this framework.

Another potential stressor in wastewater effluent is organic matter, which is measured as CBOD. CBOD is the amount of oxygen that would be consumed if all of the organic carbon in one liter of water were oxidized by microorganisms. BOD is a measure of the potential for an effluent to reduce the dissolved oxygen in a receiving surface water body.

Other chemical stressors on ecological receptors may originate from wastewater treatment systems, but are not emphasized in this framework. These could include pathogens, household products such as detergents or paints, pharmaceuticals such as antibiotics, or undigested fat or sugar substitutes. For example, algal biomass and community structure were affected by amendments of an antibiotic, an anti-microbial agent, and a surfactant in recent laboratory studies (Wilson *et al.* 2003).

Conceptual Models

Two types of surface water ecosystems are distinguished in this risk assessment framework:

- Freshwater systems (such as ponds)
- Estuarine systems (such as coastal lagoons)

The dichotomy in this framework between salt and freshwater systems is based on differences in prevailing nutrient dynamics (for example, the nutrient limitations described above). Separate conceptual models for freshwater and estuarine systems are presented in Chapter 5, *Ecological Component Framework*. Detailed descriptions of each conceptual model are presented in the following sections.

Freshwater

A generic conceptual model for wastewater treatment unit effects in freshwater lakes, streams, and ponds is depicted in Figure 5-1. Phosphorus exposure is the major determinant of phytoplankton production in most North American lakes. The nutrient may also be limiting in streams, but high water flows and flood events may overwhelm the effects of nutrients. Lake eutrophication leading to increased phytoplankton biomass may result in increased hypolimnetic oxygen deficit, decreased water clarity, and changes in species composition. Phytoplankton diversity is rarely selected as an assessment endpoint (usually not relevant to management goals, see discussion in the following sections), but it is notable that species diversity and productivity are often inversely related (Interlandi and Kilham 2001). Periphyton biomass in lakes and streams is sometimes related to nutrient concentrations, but at other sites no relationship is evident (Bourassa and Cattaneo 1998).

The relationship between aquatic macrophytes and phytoplankton is not straightforward, but it appears that epiphytes and filamentous algae increase in the presence of high nutrient loads and compete with macrophytes by shading them (Phillips *et al.* 1978). In addition, macrophytes can secrete allelopathic chemicals that reduce the growth of cyanobacteria or other algae (Scheffer *et al.* 1997 and Phillips *et al.* 1978).

Zooplankton and fish biomass may be partly controlled by phytoplankton biomass, nutrient ratios in phytoplankton, and palatability of phytoplankton. Zooplankton densities would be expected to increase in the presence of diets that are not limited by phosphorus content (Brett *et al.* 2000). Cyanobacteria are less palatable to zooplankton than other algal species (Scheffer *et al.* 1997). Models exist that relate fish yield to phytoplankton production or standing crop (Oglesby 1977), but these are not described in the characterization of effects section because indirect effects are beyond the scope of this risk assessment framework.

The toxicity of nitrate to amphibians has been observed in several studies, and the low volumes of water in ditches or vernal ponds increase the likelihood of exposures to toxic concentrations of nitrate. Direct toxicity to fish is also observed, usually at concentrations higher than those to which amphibians are sensitive. Indirectly, nitrate-resistant adult fish may increase the predation pressure on amphibian eggs and tadpoles if the fish do not experience toxicity.

Estuary/Lagoon

A generic conceptual model for wastewater treatment unit effects in a shallow estuary or lagoon is depicted in Figure 5-2. Nitrate is the primary stressor, which can be directly toxic or can interact with biota to produce secondary stressors (limited light penetration, oxygen limitation, reduction in habitat, or reduction in forage vegetation or prey). Organic matter that is associated with wastewater and directly released to surface water bodies is an additional stressor that can cause oxygen limitation.

In very shallow estuarine receiving waters, most of the primary production is performed by seagrasses such as eelgrass (*Zostera marina*), epiphytic algae, drift and attached macroalgae (sea weeds), and epibenthic microalgae (Nixon *et al.* 2001). Phytoplankton is generally less important. Nutrient levels are the key determinants of the structure of the primary producing community.

The importance of seagrass beds lies in their use as habitats for fish and shellfish, temporary nurseries for fish and shellfish, sources of food for fish, food for waterfowl, detrital food for benthic invertebrates, food for manatees, and refuges from predation. Seagrasses require rather clear water, and they are found in sheltered lagoons just below the low-tide line, at maximum depths of usually only two or three meters. Seagrass reductions have been observed in numerous, nutrient-enriched shallow marine systems (Burkholder *et al.* 1992, Hauxwell *et al.* 2003, Short and Burdick 1996, and Stevenson *et al.* 1993) and are, therefore, of concern in an ecological risk assessment for OWT systems. Nitrate can act to increase the biomass of epiphytic algae and macroalgae, causing shading of seagrasses.

The majority of the epiphyte community consists of algae, but the epiphytic complex consists of epiphytic macrophytes, microorganisms, macroalgae, metazoans, the extracellular excretions of these organisms, and mineral and organic particles sorbed on the organic matrix (Drake *et al.* 2003). At low biomass, this layer may prevent damage from ultraviolet radiation or repel potential herbivores (Drake *et al.* 2003), but at higher biomass, the epiphytic layer may shade seagrasses or possibly affect nutrient uptake and gas exchange and reduce photosynthesis (Drake *et al.* 2003 and US EPA 2001b)

In addition to shading, epiphytes preferentially absorb light in the blue and red wavelengths and thus directly compete for light with seagrass leaves (Drake *et al.* 2003). Measurements of intact epiphytes on seagrasses showed that epiphytes on turtlegrass from an oligotrophic site absorbed a maximum of 36 percent of incident light in peak chlorophyll absorption bands, whereas higher epiphyte loads on eelgrass absorbed 60 percent of incident light in peak chlorophyll absorption bands (Drake *et al.* 2003). Shading of seagrasses by phytoplankton blooms is less common but occasionally observed (Nixon *et al.* 2001).

Although much research shows that nutrient enrichment stimulates the growth of epiphytes on seagrass leaves, Nixon *et al.* (2001), working in a colder region where epiphyte growth is delayed, has not observed stimulated growth of epiphytes.

Eelgrass responds to inorganic nitrogen enrichment and to shading through the elongation of leaves and a decrease in the allocation of biomass to below ground roots and rhizomes. The lateral branching of rhizomes decreases, causing a decline in the density of shoots (Nixon *et al.* 2001). Hauxwell *et al.* (2003) also postulate that recruitment is diminished.

In addition to light-mediated effects, nitrate can have a direct toxic effect on eelgrass, particularly at warm temperatures (Burkholder *et al.* 1992 and Touchette *et al.* 2003). The mechanism may involve uncontrolled nitrate uptake, which can lead to internal phosphorus limitation, carbon limitation, or other nutrient imbalances (Burkholder *et al.* 1992). Nitrogen amendment can stimulate growth of mangrove forest trees in nitrogen-limited areas (Feller *et al.* 2003).

Phosphorus-limited lagoons such as part of the Indian River lagoon are not reflected in the generic conceptual model in Figure 5-1, but should be included in a site-specific model if phosphorus-mediated effects are possible. Mesocosms in Rhode Island showed that even nitrate-limited systems could display secondary limitation associated with phosphorus; that is, phytoplankton blooms were larger in mesocosms treated with nitrate and phosphorus than in those treated with nitrate alone (Taylor *et al.* 1995).

Losses of seagrass may lead to impacts on higher trophic-level organisms. For example, shifts from eelgrass communities to macroalgal communities that were associated with high nutrient inputs resulted in decreases in abundance, biomass and diversity of fish (Hughes *et al.* 2002 and Deegan *et al.* 2002). Macroalgae grow in dense mats on estuary sediments that reduce oxygen levels and alter the benthic community (Deegan *et al.* 2002). Moreover, declines of migrant Canada geese (*Branta canadensis*) and common goldeneye (*Bucephala clangula*) have been associated with the collapse of eelgrass beds, because of the vegetation and invertebrate prey reduction, respectively (Seymour *et al.* 2002).

Furthermore, algal blooms that are associated with high nutrient levels can deplete oxygen, especially at the benthic boundary layer. For example, decapod (crab) abundance and biomass were reduced, apparently as a result of hypoxia (Deegan *et al.* 2002). Benthic and pelagic invertebrates and fish may be affected.

Dinoflagellates such as *Pfiesteria* are protists that prey on fish and are implicated in major fish kills in estuaries and coastal waters in the mid-Atlantic and southeastern US (Burkholder and Glasgow 1997). Nitrogen and phosphorus can directly stimulate growth of dinoflagellates or their algal prey. *Pfiesteria* outbreaks are observed in poorly flushed eutrophic estuaries that are impacted by human wastewater, among other sources (Glasgow and Burkholder 2000).

Exposure Analysis

Additional information that may help the user perform an exposure analysis for the ecological risk assessment is provided in this section.

As nutrients and organic carbon enter the surface water, dilution is not instantaneous (unless the point of exposure is a small ditch or vernal pool). Water quality simulation models take loading rates or concentrations at points of entry in the water body and descriptions of mixing and reaction kinetics in a stream reach or other water body segment, and estimate pollutant concentration in a particular water body segment. The models may be steady-state or time-varying. Water quality models are most often deterministic, but occasionally stochastic (Viessman and Hammer 1985). Simplifying assumptions are often made, such as steady stream flow, first order decay of organic matter, and no influence of biota on nutrient concentrations.

Nitrogen loading (mass or moles per unit volume or unit area per day or year entering a water body) is a common exposure parameter for exposure-response models in lagoons and shallow estuaries. Short and Burdick (1996) provide an empirical relationship for estimating nitrogen loading ($kg/km^2/yr$) from the number of houses in watersheds of Waquoit Bay, Massachusetts. This regression would probably be useful for a risk assessment in the Waquoit Bay watershed. In other watersheds, an analogous relationship could be derived if there were hundreds of houses with similar treatment systems or with a series of types of treatment systems that had the same distribution of nutrients discharged through space and time. However, this type of analysis will really only be useful for macro-level assessments.

Rough approximations of nutrient inputs to water bodies can be made if the nutrient loading rate to a treatment system and the fraction of the nutrient that can be found at different distances from it are known. For example, Gold and Sims (2001) calculate an annual total loading rate of phosphorus from unsewered suburban developments at about 15 kg/ha/yr, based on

- 170 L of wastewater generated per capita per day (US EPA 1980)
- Phosphorus concentration of 16 mg/L in effluent (Reneau *et al.* 1989)
- Density of three people per household (Valiela *et al.* 1997)
- Five houses per hectare (ha)

About 60 to 95 percent of phosphorus from effluents typically is found in soils within a few meters of the drainfield (Gold and Sims 2001). As stated previously, very little phosphorus travels from onsite wastewater treatment systems to surface water bodies, unless the water is near the drainfield (Gold and Sims 2001).

In a study of 18 samples of groundwater adjacent to a lake in the Puget Sound watershed, only four showed the likely transport of more than one percent of the phosphorus released in septic tank effluent 9 to 50 meters to the lake (Gilliom and Patmont 1983). Similarly, phosphorus from septic tank effluents that is found in shallow groundwater decreases logarithmically with distance (Reneau 1979).

Even if nutrient inputs are known, an assessor still may have to utilize a water quality model to estimate concentrations in particular parts of lakes, lagoons, or streams. Gilliom and Patmont (1983) note that phosphorus in septic system effluent usually is diluted about 1,000 times before entering lake waters, though it is unclear if that relationship is still valid twenty years later.

Jones and Bachmann (1976) calculate phosphorus concentrations in lakes based on an equation of Vollenweider (1969) that requires basic information about nutrient inputs, flushing rates, and basin morphometry. Vollenweider assumes:

- The rate of input of phosphorus, the flushing rate, and the sedimentation rate are constant through time
- The lake is a continuously stirred, single compartment, open system
- The concentration of phosphorus in the outflow is identical to that in the lake
- Sedimentation of phosphorus is proportional to the phosphorus concentration in the lake

Therefore, the steady-state solution is given by:

$$TP = L/z(\sigma + p) \qquad \textbf{(Eq. E-1)}$$

where:

TP = concentration of total phosphorus (P) in lake water (mg/m^3)

L = annual P loading per unit area of lake surface (mg/m^2)

z = mean depth of lake (m)

σ = sedimentation rate, yr^{-1}

p = hydraulic flushing rate, yr^{-1}

Vollenweider's mass balance model approach to estimating nutrient concentrations in lakes is also summarized in US EPA (2000b).

Although most exposure-response models for ecological receptors in surface water require concentrations of nutrients as measures of exposure, nutrient loading rates to surface water bodies are not always easily converted to concentrations. Nixon *et al.* (2001) note that in phytoplankton-based mesocosms, such as those at the University of Rhode Island's Marine Ecological Research Laboratory, there is a good relationship between rate of nitrogen input and concentration of nitrate. However, they found that in lagoon mesocosms that may contain seagrasses, epiphytic algae, macroalgae, and benthic microflora, "there is virtually no relationship between the average concentration of inorganic nitrogen during summer in the lagoon mesocosms and the rate of nitrogen input." In summertime mesocosm experiments with a water residence time of 10 days, they observed inorganic nitrogen enrichments of 8 mmole/m^3 declining to undetectable levels within eight hours.

Moreover, Valiela and Cole (2002) demonstrate that wetlands that border lagoons can intercept (denitrify and bury) land-derived nitrogen. Thus, assessors should use caution when normalizing volumetric nitrogen loading for residence time to yield an expected or potential concentration.

Although low oxygen levels and light attenuation are potential effects of nutrient-enhanced production, we treat them as exposure parameters here because they are not assessment endpoint properties, but rather, they affect assessment endpoint properties. Temperature, reaeration, and rates of photosynthesis and decomposition of nutrient-stimulated phytoplankton and periphyton are also predictors of dissolved oxygen concentrations. Viessman and Hammer (1985) provide an example of the formulation of a water quality model to predict dissolved oxygen at a downstream location, given biochemical oxygen demand of waste discharged at an upstream location. Also, Nürnberg (1996) provides a regression of areal hypolimnetic oxygen depletion rates of North American lakes versus total phosphorus, and Kelly (2001) presents Boynton and Kemp's (2000) regression of rates of dissolved oxygen decline in the Chesapeake Bay against rates of total chlorophyll *a* deposition. However, risk assessors will be most confident in dissolved oxygen concentrations in surface water if these concentrations are measured directly.

Chlorophyll *a* is treated as a measure of exposure in one model involving seagrasses in estuaries. Although chlorophyll *a* is best measured, concentrations may be modeled based on nutrient concentrations. For example, between 1 μM and 20 μM dissolved inorganic nitrogen in shallow estuaries, chlorophyll *a* tends to increase at slightly less than 1 μg/L with every 1 μM increase in dissolved inorganic nitrogen or approximately about 0.75 μg chlorophyll per μM dissolved inorganic nitrogen (Figure 3-2b in US EPA 2001b). Also, Nixon (2001) provides a relationship between mean chlorophyll *a* and nitrogen input from an experiment in which nitrogen, phosphorus and silicon were added in molar ratio of 12:1:1 at Marine Ecosystems Research Laboratory in Rhode Island.

Effects Assessment

Additional information that may help the user develop an effects assessment for the ecological risk assessment is provided in this section.

In choosing among exposure-response models, the following principles are worthy of consideration:

- The use of more than one type of evidence or model should give the assessor more confidence in the result and aid in the characterization of uncertainty
- Models derived from data collected in ecosystems that are similar to the ecosystem of concern (for example, oligotrophic versus eutrophic conditions, epiphyte versus macroalgae versus phytoplankton dominance, nitrogen versus phosphorus-limited conditions) and with species that are related to assessment endpoint entities are recommended
- If a site-specific model is not available, general models are usually preferable to site-specific models for other types of sites
- Laboratory or mesocosm-derived values are more reliable if they have been verified in the field

- Models that use measures of exposure that are available to the assessor are most useful
- The most direct estimate (involving a model with few parameters) usually has the lowest uncertainty
- Thresholds are useful for evaluating if there is an effect, but not for quantifying its magnitude

When field observations are used, it may not be possible to attribute causation, if multiple stressors are present or if multiple sources of one stressor are present.

Thresholds for effects and other exposure-response relationships for the focal assessment endpoint entities are presented in the following section. No specific level of protection is assumed. For example, because the level-of-effect component of the assessment endpoint is selected by the risk manager, this framework does not arbitrarily present concentrations of nitrate that are associated with a specific percentage loss of a seagrass bed. Examples of many different levels of effect are provided.

Water Quality Criteria for Nutrients

In the 1986 document *Quality Criteria for Water*, US EPA recommended no water quality criterion for nitrate, noting that concentrations of nitrate-nitrogen at or below 90 mg/L would have no adverse effects on warm-water fishes, and that this concentration would rarely occur in nature. Water quality criteria for ambient dissolved oxygen concentrations were 6.5 mg/L for the protection of larval stages of coldwater fish and invertebrates (9.5 mg/L for those embedded in sediments) and 6.0 mg/L for the protection of larval stages of warm-water organisms (US EPA 1986).

More recently, US EPA (1998a) developed a *National Strategy for the Development of Regional Nutrient Criteria*, in which a two-phase process for the development of water quality standards for nutrients is described. First, “nutrient criteria guidance” for nitrogen, phosphorus, and related parameters such as chlorophyll *a*, Secchi disk depth, and algal biomass are under development. The guidance will be expressed as numerical ranges that vary based on the type of water body (streams and rivers, coastal waters and estuaries, lakes and reservoirs, and wetlands) and region of the country (using hierarchical Level III ecoregions developed by James Omernik of the US EPA Corvallis, OR laboratory).

US EPA expects states and tribes to develop nutrient water quality criteria “to support designated uses of waters” by the end of 2003 (US EPA 1998a). These criteria may be values within US EPA’s published ranges, other values derived from US EPA and state databases, or alternative values derived using US EPA’s methodology or other scientifically defensible methods.

Because US EPA did not recommend specific models for developing nutrient criteria (such as US EPA 2001a), it is not clear to what extent these criteria will represent effects levels for aquatic biota. Thresholds for human toxicity would not be useful for ecological risk assessments. However, it is obvious that criteria for estuarine and marine coastal waters will not reflect drinking water use.

Estuary/Lagoon Exposure-Response Relationships

Estuary/lagoon exposure-response relationships are described in this section for

- Seagrass
- Benthic invertabrate and fish communities

Seagrass

Exposure-response relationships for seagrasses are presented in Table E-1. These range from thresholds to continuous relationships. In toxicology jargon, thresholds may be called lowest observed adverse effects levels (LOAELs) or lowest observed adverse effects concentrations (LOAECs). Measures of exposure include nutrients, many of the secondary stressors that result from nutrients (light attenuation, chlorophyll *a*, epiphyte load), and direct measures of seagrass.

Among nutrients, nitrogen loading rates and nitrate concentrations are emphasized, but phosphate is also considered. A significant relationship is found between the number of houses in a watershed and the percent cover of eelgrass. The measures of epiphyte and seagrass parameters are the most direct measures of endpoint properties, but these measurements are only useful in ecological risk assessments of current conditions where field measurements are feasible.

Many of the relationships in Table E-1 involve single nutrients, although it is clear that combinations of nutrients or other factors can alter effects thresholds for seagrasses. For example, Stevenson *et al.* (1993) provide dissolved inorganic nitrate concentrations that are associated with regrowth of four species of submerged aquatic plants. However, vegetation survival can also occur if the concentration of one nutrient (N or P) is low enough to limit algae that do not have access to sediment pools of nutrients. In addition, Stevenson *et al.* (1993) acknowledge that their thresholds may not apply if one of the factors changes independently of others.

Table E-1
Exposure-Response Relationships for Seagrasses

Measure of Exposure (X)	Measure of Effect (Y)	Type of Model	Value or Relationship[1]	Reference
Number of houses in watershed	% sediment area covered with eelgrass	Empirical, Waquoit Bay, MA	Log(Y) = 1.666–0.0004(X)	Short and Burdick (1996)
N loading ($kg/km^2/yr$)	% sediment area covered with eelgrass	Empirical, Waquoit Bay, MA	Log (Y) = 1.648–0.000044(X)	Short and Burdick (1996)

Table E-1
Exposure-Response Relationships for Seagrasses (Cont.)

Measure of Exposure (X)	Measure of Effect (Y)	Type of Model	Value or Relationship[1]	Reference
N loading (kg/ha/yr)	Eelgrass loss	Threshold, loss of 80–96% of bed area in 1990s, Waquoit Bay, MA	30	Hauxwell *et al.* (2003)
N loading (kg/ha/yr)	Total disappearance of eelgrass	Threshold, Waquoit Bay, MA	60	Hauxwell *et al.* (2003)
N loading (kg/ha/yr)	% seagrass production/total production	Empirical, numerous estuaries	$Y = 145.653(X^{-0.550})$	Valiela and Cole (2002)
N loading (kg/ha/yr)	% seagrass area lost (10–30 yr)	Empirical, numerous estuaries	$Y = 0.693(x) +14.211$	Valiela and Cole (2002)
N loading (kg/ha/yr)	Seagrass cover, production, extent of meadows	Threshold, numerous estuaries	20–100	Valiela and Cole (2002)
N loading (kg/ha/yr)	Seagrass cover, production, extent of meadows	Threshold, Cape Cod	20–30	Valiela and Cole (2002)
Nitrate-N loading (μM/d)	Eelgrass growth and survival	Experimental threshold, North Carolina mesocosm	3.5	Burkholder *et al.* (1992)
Total N concentration	Uninhibited eelgrass growth	Threshold (maximum), Chesapeake Bay	10 mmol/m^3	References cited in Nixon *et al.* (2001)
Total N concentration	Uninhibited eelgrass growth	Threshold (maximum), coastal Denmark	70 mmol/m^3	References cited in Nixon *et al.* (2001)
Input of total N	Decline of seagrass bed	Threshold, mesocosms, Rhode Island	2 mmol N/m^2/d	Nixon *et al.* (2001)
Dissolved Inorganic N concentration	Regrowth of submerged aquatic vegetation (Ruppia maritime, Potomogeton perfoliatus, Potomogeton pectinatus)	Threshold, estuarine gradient, Chesapeake Bay	<10 μM (or N:P ratio >100 or ~1)	Stevenson *et al.* (1993)

Table E-1
Exposure-Response Relationships for Seagrasses (Cont.)

Measure of Exposure (X)	Measure of Effect (Y)	Type of Model	Value or Relationship[1]	Reference
Insolation at surface of canopy compared to water surface	Decline of seagrasses or submerged aquatic macrophytes	Threshold	11% (5–20%)	US EPA 2001b
Dissolved inorganic P concentration	Regrowth of submerged aquatic vegetation (Ruppia maritime, Potomogeton perfoliatus, Potomogeton pectinatus)	Threshold, estuarine gradient, Chesapeake Bay	<0.35 μM (or N:P ratio >100 or ~1)	Stevenson *et al.* (1993)
Total suspended solids	Regrowth of submerged aquatic vegetation (Ruppia maritime, Potomogeton perfoliatus, Potomogeton pectinatus)	Threshold, estuarine gradient, Chesapeake Bay	<20 mg/L	Stevenson *et al.* (1993)
Chlorophyll *a*	Regrowth of submerged aquatic vegetation (Ruppia maritime, Potomogeton perfoliatus, Potomogeton pectinatus)	Threshold, estuarine gradient, Chesapeake Bay	<15 μg/L	Stevenson *et al.* (1993)
Epiphyte biomass (μg C/cm^2)	Photosynthetically available radiation-based photosynthesis of eelgrass and turtle grass, normalized to maximum rate	Empirical, Monterey Bay and Bahamas	$Y = -0.0025(x)+1$	Drake *et al.* (2003)
Epiphyte biomass (μg C/cm^2)	Spectral photosynthesis of eelgrass and turtle grass, normalized to maximum rate	Empirical, Monterey Bay and Bahamas	$Y = -0.0055(x)+1$	Drake *et al.* (2003)

Table E-1
Exposure-Response Relationships for Seagrasses (Cont.)

Measure of Exposure (X)	Measure of Effect (Y)	Type of Model	Value or Relationship[1]	Reference
Seagrass leaf elongation	Unlikely bed survival	Environmental indicator	>1 cm/d	Nixon *et al.* (2001)
Density of seagrass	Unlikely bed survival	Environmental indicator	<100–150 shoots/m^2	Nixon *et al.* (2001)
Seagrass shoot to root ratio at midsummer	Unlikely bed survival	Environmental indicator	>1 or 2	Nixon *et al.* (2001)

[1]Caution—consult each study to determine the applicability of relationships before using them for a particular risk assessment.

Measures of effect in Table E-1 include photosynthetic rates, growth, and cover. Aerial photography and remote sensing can be used to measure seagrass habitat loss directly, but these methods are not sensitive to small changes in biomass density and cannot be used to attribute causation to particular nutrients or sources unless ground measurements are taken.

Reading the original references cited in Table E-1 is important for the risk assessor to determine if the relationships are valid for the range of exposure concentrations observed or predicted at the site of concern. For example, seagrass biomass is somewhat predictable at given levels of nitrogen input, but above certain levels of nitrogen, factors other than depth, water residence time, and nitrogen input are necessary to predict the dominant plant type in very shallow marine systems (Nixon *et al.* 2001).

Benthic Invertebrate and Fish Communities

As stated in the description of the conceptual model for shallow estuaries, effects on benthic and water column populations are possible due to trophic level interactions caused by changing vegetation (for example, relative dominance of seagrass and algae). However, these effects are unlikely to occur as a result of releases of nutrients from one OWT system, so these exposure-response relationships are not presented here.

A more direct effect would be mortality due to low oxygen levels. Low oxygen tends to affect sessile benthic organisms first, because they are exposed to the lowest concentrations of oxygen. Hypoxia is generally defined as a water column oxygen concentration less than 2 mg/L (Kelly 2001).

Rosenberg *et al.* (1991) recommend an exposure limit of 1.4 mg/L oxygen for several days to weeks for coastal benthic communities. Standards for US states are sometimes higher (for example, 5 to 6 mg/L), but are not necessarily intended to protect the most sensitive species (NRC 2000).

Numerous data are available on the effects of hypoxia on various species. Many of these data are summarized in US EPA (2000a) and are not repeated here. However, because many of these data are expressed as consistent, standard test endpoints, these values are used to illustrate the utility of a species sensitivity distribution. A fraction or percentile of the distribution of test endpoint concentrations for various species can be used to identify a concentration to which that fraction of the community would be affected. For example, half of saltwater fish populations would be expected to have LC50s at 1.12 mg/L or higher concentrations of dissolved oxygen (Figure E-1). An untested species may be assumed to be a random draw from the distribution, or the distribution may represent the proportion of species in a fish community that is likely to be affected by a particular concentration of dissolved oxygen. For distributions of most nutrients, the X-axis would be expected to have a greater range of values than this curve related to dissolved oxygen. The uses and forms of species sensitivity distributions are described in Postuma *et al.* (2002).

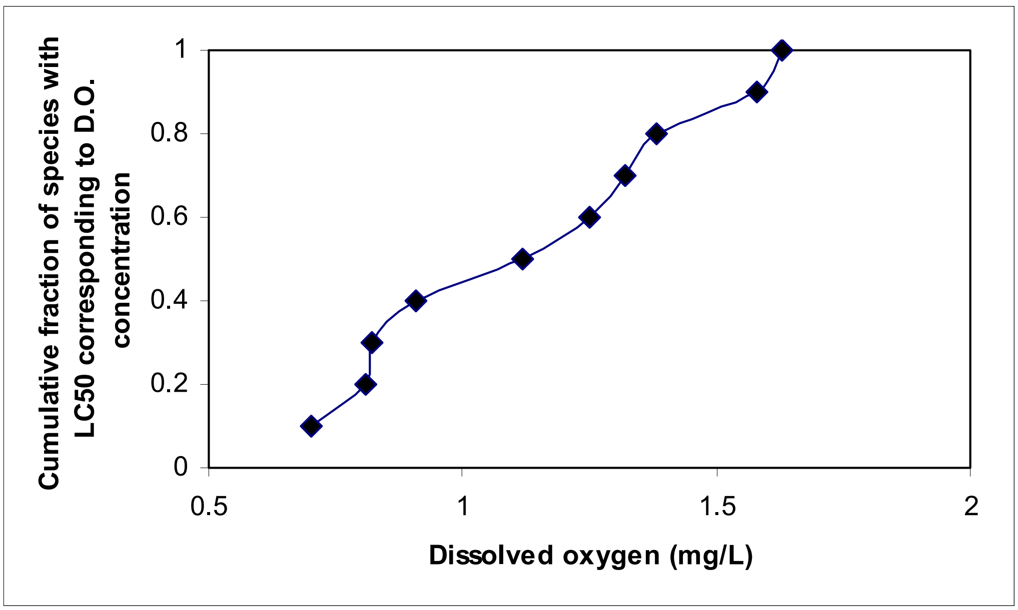

Figure E-1
Species Sensitivity Distribution of LC50s for Saltwater Fish Exposed to Low Concentrations of Dissolved Oxygen

Fresh Water Exposure-Response Relationships

Fresh water exposure-response relationships are described in this section for

- Phytoplankton
- Periphyton
- Aquatic macrophytes
- Benthic invertabrate and fish communities
- Amphibian populations

Phytoplankton

Chlorophyll is a useful measure of phytoplankton biomass, and several broad relationships between phosphorus concentrations in lakes and chlorophyll *a* are available to the risk assessor (Table E-2). Most of the regressions presented in Table E-2 are linear, but van Nieuwenhuyse and Jones (1996), based on a few references, note that the total phosphorus-chlorophyll relationship for lakes may be curvilinear across broad ranges of phosphorus concentrations. They provide relationships between total phosphorus and chlorophyll in streams, with one model including stream catchment area. The relationships for streams tend not to be as tight as those for lakes, because of the importance of flow generally, and flooding intervals, specifically.

The relationship between total phosphorus and primary productivity in Wetzel (1983) is also nonlinear because of the self-shading effects of dense algal populations. In some systems, the predictions of chlorophyll may be affected by grazing pressure (such as a high number of filter-feeding bivalves, Nixon *et al.* 2001), high turbidity, and nitrogen limitation.

Algal species dominance is more difficult to predict than total production. Wetzel (1983) provides a table of minimum phosphorus requirements per unit cell volume of algal genera that are common to lakes of progressively increasing productivity: *Asterionella*, *Fragilaria*, *Tavellaria*, *Scenedesmus*, *Oscillatoria*, and *Microcystis*. Blooms of cyanobacteria, a subset of the phytoplankton community, would be expected to be assessment endpoint entities in many risk assessments for onsite wastewater treatment. In an analysis of 17 world lakes, Smith (1983) found that water bodies having an epilimnetic total nitrogen to total phosphorus ratio greater than 29, by weight, usually had low proportions of cyanobacteria. Scheffer *et al.* (1997) did not find a significant relationship between abundance of cyanobacteria and either the total nitrogen to phosphorus ratio or the concentration of phosphorus in 55 Dutch lakes, but did find a significant relationship with Secchi disk depth and another multivariate shade indicator.

Table E-2
Exposure-Response Relationships for Phytoplankton in Fresh Water

Measure of Exposure (X)	Effect (Y)	Type of Model	Value or Relationship	Reference
Mean annual concentration, total P (mg/m^3)	Mean annual chlorophyll *a* (mg/m^3)	Empirical, lakes in Experimental Lakes Area, Ontario	$Y = 0.987 X – 6.520$	Schindler (1977)
Input of P, normalized for mean depth, water residence time, P sedimentation	Mean annual chlorophyll *a* (mg/m^3)	Empirical, several oligotrophic lakes, mesotrophic lakes, eutrophic lakes	See paper; graph reprinted in Nixon *et al.* (2001)	Vollenweider (1976)
Total P concentration	Summer mean chlorophyll *a* (mg/m^3)	Empirical, 143 lakes	$\log Y = 1.46 \log X – 1.09$	Jones and Bachmann (1976)

Table E-2
Exposure-Response Relationships for Phytoplankton in Fresh Water (Cont.)

Measure of Exposure (X)	Effect (Y)	Type of Model	Value or Relationship	Reference
Total P concentration at spring overturn (mg/m^3)	Summer mean chlorophyll *a* (mg/m^3)	Empirical, 19 lakes in southern Ontario and 27 other North American lakes	$\log Y = 1.449 \log X - 1.136$	Dillon and Rigler (1974)
Summer mean total P concentration (mg/m^3)	Summer mean chlorophyll *a* (mg/m^3)	Empirical, 292 stream samples, worldwide	$\log Y = -1.65 + 1.99 \log X - 0.28 (\log X)^2$	van Nieuwenhuyse and Jones (1996)
Summer mean total P concentration (mg/m^3); stream catchment area	Summer mean chlorophyll *a* (mg/m^3)	Empirical, 292 stream samples, worldwide	$\log Y = -1.92 + 1.96 \log X_1 - 0.30 (\log X_1)^2 + 0.12 + 0.12 \log X_2$	van Nieuwenhuyse and Jones (1996)
Predicted total P concentration (mg/m^3)	Annual primary productivity ($g\ C/m^2/yr$)	Empirical, Laurentian Great Lakes, other American Lakes, European Lakes	See Wetzel (1983), Figure 13-10	Wetzel 1983, from Vollenweider 1979

Eutrophication is not listed as an assessment endpoint entity in this risk assessment framework, but the process may be of interest to a risk manager. A risk-assessment goal may be to determine whether the lake has transitioned to a higher trophic state in the past decade, that is, from oligotrophic to mesotrophic or mesotrophic to eutrophic. Various investigators provide classification schemes relating nutrient concentrations to trophic designations. For example, Wetzel (1983, Table 13-14) modifies a scheme from Vollenweider (1979) that includes ranges of total phosphorus concentrations, total nitrogen concentrations, chlorophyll *a* concentrations of phytoplankton, chlorophyll *a* peak concentrations, and Secchi disk depth (transparency).

Periphyton

Periphyton biomass is not always related to nutrient concentration, and Bourassa and Cattaneo (1998) review several of the studies that observed significant relationships and those that did not. In one study, more than half of the periphyton biomass in 13 rivers in southern Ontario and western Quebec were explained by total phosphorus concentration (Chételat *et al.* 1999). In another investigation, almost half of the variation in mean monthly chlorophyll *a* in 25 New Zealand rivers was explained with a combination of dissolved nutrient data and days of accrual, to account for flood frequency (Biggs 2000). Bourassa and Cattaneo (1998) observed that in the range of 5 to 60 µg/L of phosphorus in twelve Laurentian streams in Quebec, grazer biomass and mean grazer size explain a majority of the variability in periphyton, with current velocity and depth also being significant, but phosphorus not being significant.

If periphyton biomass or production is an assessment endpoint property in an ecological risk assessment, the assessor should examine all available studies to determine which exposure-response relationship to use (or if nutrient inputs are likely to be significant at all).

Aquatic Macrophytes

Macrophyte abundance and production is influenced by complex processes that include nutrient availability, light penetration, and additional biotic factors. Bachmann *et al.* (2002) found that macrophytes were predictably absent in Florida lakes at phosphorus concentrations above 0.166 mg/L.

Other investigators have hypothesized that macrophytes decline when they are shaded by epiphytes and filamentous algae that are stimulated by high nutrient loads (Phillips *et al.* 1978), although Bachmann *et al.* did not observe this behavior. The pondweed *Potamogeton pectinatus* remained in a Netherlands lake at phosphorus levels above 0.6 mg/L (van den Berg *et al.* 1999). Charophytes (macroalgae) were observed to disappear from this lake at phosphorus concentrations above 0.3 mg/L, but recolonization required concentrations below 0.1 mg/L phosphorus (van den Berg *et al.* 1999).

Below the high nutrient threshold from Bachmann *et al.* (2002), there was no relationship between nutrients and densities of submerged macrophytes. The lack of a relationship may be explained by the fact that macrophytes can obtain their nutrients from sediments in addition to the water column (Bachmann *et al.* 2002). Furthermore, Scheffer *et al.* (1993) found alternative equilibria in shallow lakes at similar nutrient levels, that is, a clear state dominated by aquatic macrophytes or a turbid state with high algal biomass. Macrophytes tend to have more of a predictable effect on nutrient concentrations (because of uptake) than nutrients have on macrophytes (Bachmann *et al.* 2002).

Light penetration is also a factor in determining macrophyte biomass, but light thresholds may not be useful for indicating macrophyte dominance. Lakes in Florida showed a decrease in the biomass of macrophytes below water color values of about 150 Pt-Co (platinum-cobalt) units, but phytoplankton abundance also decreased at that color. The average Secchi disk depth in macrophyte-dominated lakes was greater than the depth in algal-dominated lakes, but there was a broad range of overlap between the two groups of lakes. For a single lake in the Netherlands, van den Berg *et al.* (1999) found a threshold Secchi depth (0.4 m) below which charophytes are not observed, but a slight negative correlation of *Potamogeton pectinatus* with Secchi depth.

High flows are a factor that is pertinent to macrophytes in streams. Wade *et al.* (2001) present a dynamic, mechanistic model for the Kennet River in southern England that represents the phosphorus cycle of reservoirs (including total and soluble reactive phosphorus) and in-stream processes that control the transfer of phosphorus between those reservoirs. Water flow, suspension and deposition of suspended sediment, and growth of epiphytes and macrophytes are modeled. The use of this model is not necessarily recommended here, because many of the parameters and processes are specific to the Kennet River.

However, the model is a useful illustration that shows

- Exposure-response relationships can be represented by mechanistic models
- The processes that regulate macrophyte growth in fresh water are complex and not necessarily amenable to simple thresholds or regressions

Therefore, risk assessors with interest in macrophyte growth in freshwater systems may need to perform site-specific research to support the characterization of effects.

Benthic Invertebrate and Fish Communities

The benthic invertebrate and fish communities are identified as assessment endpoint entities, and nitrate and dissolved oxygen are depicted as measures of exposure in Figure 5-1 and Figure 5-2.

Nitrate is generally the least toxic of the three forms of nitrogen (ammonia, nitrate, and nitrite) to fish and amphibians (Rouse *et al.* 1999). Hecnar (1995) notes that LC50s for fish in several studies range from 800 to 12,000 mg/L nitrate (180 to 2,700 mg/L nitrate-N), high values that are consistent with US EPA's 1986 decision not to set a water quality criterion for nitrate (see previous discussion). However, significant mortality of eggs and/or fry of salmonid species have been recorded at 5 (steelhead trout, *Salmo gairdneri*), 10 (rainbow trout, *Salmo gairdneri*; cutthroat trout, *Salmo clarki*), and 20 (chinook salmon, *Oncorhynchus tsawtscha*) mg/L nitrate (1.1, 2.3, and 4.5 mg/L nitrate-N) in waters of low hardness (Kincheloe *et al.* 1979). Thus, it is recommended that the risk assessor review existing effects data for toxicity to fish of different species and life stages, if the fish community is an assessment endpoint entity.

A species sensitivity distribution of toxicity values for effects levels of hypoxia may be plotted that would be analogous to the distribution in Figure E-1, derived from US EPA (2000a). Effects of low oxygen in fresh water would be expected to be similar to those in salt water and to display similarly low variability.

Amphibian Populations

Recent research has indicated that amphibians may be at risk from nitrate in vernal ponds, lakes, and streams. Effects levels for toxicity of nitrate to amphibians are presented in Table E-3. Water quality criteria intended to protect human health (10 mg/L) may not be protective of some amphibians. An assessor may choose one or more of these effects levels for use in a risk assessment (depending on the amphibian species of concern), or similar effects levels may be plotted in a species sensitivity distribution analogous to that in Figure E-1, and the sensitivity of an untested species may be assumed to be a random variate in that distribution. This plot would facilitate comparisons with distributions of nitrate concentrations from the characterization of exposure. However, the assessor should note that nitrate tolerance of amphibians such as the common frog may vary, based on the level of adaptation in a particular region (Johansson *et al.* 2001).

Effects on reproduction, growth, and survival are assumed to relate most directly to the endpoint properties of abundance and production. Deformities are usually related to reproduction as well, and may be considered important by themselves. Therefore, tests that focus on these effects are more pertinent to and useful in the risk assessment than tests of behavior.

Table E-3
Effects Levels for Toxicity of Nitrate-N to Amphibians

Species	Common Name	Stage	Chemical	Toxicity Test Endpoint	Concentration (mg/L)	Reference
Bufo americanus	American toad	Tadpole	Ammonium nitrate	96-hr LC50	13.6, 39.3	Hecnar (1995)
Pseudacris triseriata	Chorus frog	Tadpole	Ammonium nitrate	96-hr LC50	17.0	Hecnar (1995)
Rana pipiens	Leopard frog	Tadpole	Ammonium nitrate	96-hr LC50	22.6	Hecnar (1995)
Rana clamitans	Green frog	Tadpole	Ammonium nitrate	96-hr LC50	32.4	Hecnar (1995)
Pseudacris triseriata	Chorus frog	Tadpole	Ammonium nitrate	Chronic: Development, behavior or mortality	2.5–10	Hecnar (1995)
Rana pipiens	Leopard frog	Tadpole	Ammonium nitrate	Development, behavior or mortality	2.5–10	Hecnar (1995)
Rana clamitans	Green frog	Tadpole	Ammonium nitrate	Development, behavior	2.5–10	Hecnar (1995)
Bufo bufo	Common toad	Tadpole	Ammonium nitrate	96 hr LC50	385	Xu and Oldham (1997)
Bufo bufo	Common toad	Tadpole	Ammonium nitrate	168 hr LC50	338	Xu and Oldham (1997)
Bufo bufo	Common toad	Tadpole/ meta-morph	Ammonium nitrate	30-d Subchronic, 21% lethality and 17% failure to resorb tails during metamorphosis	23	Xu and Oldham (1997)

Table E-3
Effects Levels for Toxicity of Nitrate-N to Amphibians (Cont.)

Species	Common Name	Stage	Chemical	Toxicity Test Endpoint	Concentration (mg/L)	Reference
Bufo bufo	Common toad	Tadpole	Sodium nitrate	Growth	9	Baker and Waights (1993)
Bufo bufo	Common toad	Tadpole	Sodium nitrate	Mortality	22.6	Baker and Waights (1993)
Litoria caerulea	Tree frog	Tadpole	Sodium nitrate	Development	9	Baker and Waights (1994)
Litoria caerulea	Tree frog	Tadpole	Sodium nitrate	Mortality	9	Baker and Waights (1994)
Ambystoma gracile	Northwestern salamander	Larva	Potassium nitrate	15-d, LOAEC (13% lethality)	12.5	Marco and Blaustein (1999)
Rana pretiosa	Oregon spotted frog	Larva	Potassium nitrate	15-d, LOAEC (39% lethality)	12.5	Marco and Blaustein (1999)
Trituris helvetica	Palmate newt	Larva	Ammonium nitrate	Mortality, Rapid metamorphosis	11.3	Watt and Jarvis (1997)
Pseudacris regilla	Pacific tree frog	Embryo	Sodium nitrate	10-d LC50	578.0	Schuytema and Nebeker (1999)
Xenopus laevis	African clawed frog	Embryo	Sodium nitrate	4, 5-d LC50	871.6, 438.4	Schuytema and Nebeker (1999)
Pseudacris regilla	Pacific tree frog	Embryo	Sodium nitrate	10-d, LOAEC, length, weight	111.0	Schuytema and Nebeker (1999)
Xenopus laevis	African clawed frog	Embryo	Sodium nitrate	5-d, LOAEC, length	111.0	Schuytema and Nebeker (1999)

Table E-3
Effects Levels for Toxicity of Nitrate-N to Amphibians (Cont.)

Species	Common Name	Stage	Chemical	Toxicity Test Endpoint	Concentration (mg/L)	Reference
Xenopus laevis	African clawed frog	Embryo	Sodium nitrate	5-d, LOAEC, weight	56.7	Schuytema and Nebeker (1999)
Xenopus laevis	African clawed frog	Embryo	Sodium nitrate	5-d, LOAEC, deformity	230.4	Schuytema and Nebeker (1999)
Xenopus laevis	African clawed frog	Embryo	Sodium nitrate	4-d, LOAEC, frog embryo teratogenesis assay, weight, length	>470.4	Schuytema and Nebeker (1999)
Rana sylvatica	Wood frog	Egg	Sodium nitrate	Hatching success, deformity	>9	Laposata and Dunson (1998)
Ambystoma jeffersonianum	Jefferson salamander	Egg	Sodium nitrate	Hatching success, deformity	>9	Laposata and Dunson (1998)
Ambystoma maculatum	Spotted salamander	Egg	Sodium nitrate	Hatching success, deformity	>9	Laposata and Dunson (1998)
Bufo americanus	American toad	Egg	Sodium nitrate	Hatching success, deformity	>9	Laposata and Dunson (1998)
Rana cascadae	Cascades frog	Larva	Sodium nitrate, pH 5, pH 7	21-d mortality, stat signif	>4.5	Hatch and Blaustein (2000)
Rana cascadae	Cascades frog	Larva	Sodium nitrate, pH 5	21-d mortality, 31%, not stat signif	1	Hatch and Blaustein (2000)
Rana cascadae	Cascades frog	Larva	Sodium nitrate, pH 5, UV-B	21-d mortality, 50%, 25%	1, 4.5	Hatch and Blaustein (2000)
Xenopus laevis	African clawed frog	tadpole	Sodium nitrate	mortality	>65.9	Sullivan and Spence (2003)

Source: Rouse *et al.* (1999) used with permission from *Environmental Health Perspectives* and amended with additional data.

Reduced feeding and weight loss were observed in tadpoles in Hecnar (1995) and Baker and Waights (1993, 1994).

Hecnar speculates that bacteria in the gut may reduce nitrate to nitrite, oxidizing hemoglobin to methemoglobin and reducing oxygen uptake.

Ammonium nitrate was used as the source of nitrate in some tests, and toxicity of the ammonium ion may be partly responsible for the toxicity. Johansson *et al.* (2001) showed greater mortality of common frogs exposed to ammonium nitrate than of those exposed to the same concentration of sodium nitrate. Therefore, thresholds derived from bioassays of ammonium nitrate should be used with caution.

Although beyond the scope of this framework, a risk assessment for larger scale wastewater treatment at a distance from surface water bodies should consider the following additional factors:

- Potential effects of nitrate in soil on amphibians. In one study amphibian mortality was observed on recently fertilized fields (Schneeweiss and Schneeweiss 1997), as cited in Rouse 1999. Another investigation measured effects of ammonium nitrate on *Rana temporaria* (Oldham *et al.* 1997).
- Nitrite as a stressor. Nitrite has been exhibited to impede tadpole development at levels as low as 3.5 mg/L (Marco and Blaustein 1999).
- Potential indirect effects of nitrate on amphibians. Amphibian insect prey and some predators (fish) are sensitive to similar levels of nitrate as amphibians (Rouse *et al.* 1999). Also, effects of nitrate on tadpoles may be mediated by effects on their algal forage (Xu and Oldham 1997). Thus, amphibians may be impacted indirectly by aquatic community dynamics, and the direction of the expected effect would depend on the relative sensitivity of amphibians, prey, and predators to nitrate.

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WU-HT-01-18

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