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A Review of Ecological Assessment Case Studies from a Risk Assessment Perspective

RISK ASSESSMENT FORUM

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A REVIEW OF ECOLOGICAL ASSESSMENT CASE STUDIES FROM A RISK ASSESSMENT PERSPECTIVE

Assembled by:

Eastern Research Group, Inc.
110 Hartwell Avenue
Lexington, MA 02173-3198
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Risk Assessment Forum
U.S. Environmental Protection Agency
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FOREWORD

To gain insight into the process of ecological risk assessment, scientists from the U.S. Environmental Protection Agency (EPA) have analyzed a cross-section of case studies representing the "state-of-the-practice" in ecological assessment. Each case was evaluated by scientific experts in a series of EPA-sponsored workshops between May 29 and June 20, 1991 (56 *Federal Register* 22869, 17 May 1991). These peer review workshops were chaired by Dr. Charles Menzie and included reviewers from universities, private organizations, and other federal agencies.

The case study workshops, along with two other workshops held during the spring of 1991 (57 *Federal Register* 22236, 27 May 1992), were part of a new EPA program to develop guidelines for ecological risk assessment. Each workshop was designed to open a dialogue among experts on issues pertaining to the development of such guidelines. This report brings together 12 case studies that illustrate important ecological risk assessment practices.

The case studies are wide-ranging in scope, representing a variety of ecosystems, ecological endpoints, chemical and nonchemical stressors, and programmatic requirements within the Agency. As a result, workshop participants were presented with a broad diversity of risk assessment and scientific issues, and many useful principles emerged from the resulting discussions.

The case studies report provides a useful first look at some common approaches to ecological assessment in relationship to a general ecological risk process. The cases selected were evaluated at the workshops as to whether they (1) effectively addressed generally accepted components of an ecological risk assessment, or (2) addressed some but not all of these components or, instead, (3) provided an alternative approach to assessing ecological effects. The analyses and discussions in this report provide useful information about ecological risk processes.

Dorothy E. Patton, Ph.D.
Chair
Risk Assessment Forum

REPORT AUTHORS AND CASE STUDY WORK GROUP LEADERS

This report was prepared by Dr. William van der Schalie and Dr. Ronald Landy of the U.S. Environmental Protection Agency, and Dr. Charles Menzie, a consultant for this activity. The case studies were initiated by EPA work groups chaired by the individuals listed below, with Dr. Landy as overall chair. The workshops were organized by Dr. Landy and Dr. van der Schalie, with the assistance of Dr. Menzie. Eastern Research Group, Inc. (ERG), EPA's contractor for this activity, provided organizational support and assembled this report. Case study authors and peer reviewers are listed at the beginning of each case study (part II).

REPORT AUTHORS

William van der Schalie
Risk Assessment Forum Staff
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

Ronald Landy
Office of Technology Transfer
and Regulatory Support
U.S. Environmental Protection Agency
Washington, DC

Charles Menzie
Menzie-Cura & Associates, Inc.
Chelmford, MA

CASE STUDY WORK GROUP LEADERS

Steven Broderius
Environmental Research
Laboratory—Duluth
U.S. Environmental Protection Agency
Duluth, MN

Randall Bruins
Region 10
U.S. Environmental Protection Agency
Seattle, WA

Patricia Cirone
Region 10
U.S. Environmental Protection Agency
Seattle, WA

James Clark
Environmental Research
Laboratory—Gulf Breeze
U.S. Environmental Protection Agency
Gulf Breeze, FL

Milton Clark
Region 5
U.S. Environmental Protection Agency
Chicago, IL

Clyde Houseknecht
Office of Water
U.S. Environmental Protection Agency
Washington, DC

Jay Messer
Atmospheric Research and
Exposure Assessment Laboratory
Research Triangle Park, NC

Ronald Preston
Region 3
U.S. Environmental Protection Agency
Wheeling, WV

CASE STUDY WORK GROUP LEADERS (continued)

Jon Rauscher
Region 6
U.S. Environmental Protection Agency
Dallas, TX

Harvey Simon
Region 2
U.S. Environmental Protection Agency
New York, NY

Quentin Stober
Region 4
U.S. Environmental Protection Agency
Atlanta, GA

Greg Susanke
Office of Pesticide Programs
U.S. Environmental Protection Agency
Washington, DC

Daniel Vallero
Atmospheric Research and
Exposure Assessment Laboratory
U.S. Environmental Protection Agency
Research Triangle Park, NC

Bill Williams
Ecological Planning and Toxicology, Inc.
Corvallis, OR

SUMMARY

This report uses case studies to explore the relationship between the process of ecological risk assessment and common approaches used by EPA (and others) to evaluate adverse ecological effects. The case studies are wide-ranging in scope, representing a variety of ecosystems, ecological endpoints, chemical and nonchemical stressors, and programmatic requirements within the Agency. The case studies were evaluated at peer review workshops as to whether they (1) effectively addressed general components of an ecological risk assessment—problem formulation, analysis, and risk characterization—or (2) addressed some but not all of these components or, instead, (3) provided an alternative approach to assessing ecological effects. Case study strengths and limitations noted by the reviewers and authors are included in the comment boxes contained in each case study.

Some of the themes that emerged from these diverse case studies are highlighted below.

- Discussions between the risk assessor, risk manager, and relevant experts are critical both at the beginning and end of an ecological risk assessment. Discussion at the beginning will help ensure that the final assessment will contribute to a management decision as well as address important ecological concerns. When the risk assessment has been completed, the conclusions, assumptions, and uncertainties of the risk assessment must be clearly conveyed.

- Some difficulties encountered in the case studies might have been avoided had more attention been paid to the initial planning stages of the assessment (problem formulation). Some case studies were too narrowly focused and did not consider all relevant stressors (both chemical and nonchemical). The ecological values to be protected should be carefully considered and clearly identified at the outset of an assessment.

- While ecological exposure and effects models were useful, sensitivity analyses and validation studies were frequently insufficient to evaluate the relevance of the models to "real world" situations.

- Field studies provided a level of realism not readily attainable in laboratory studies, but multiple stressors frequently made it difficult to identify a particular stressor as the cause of observed ecological effects. Finding a suitable reference site for comparison against a potentially affected area also was difficult.

- The case studies varied widely in their approaches to presenting the results of an assessment, although for chemical stressors relatively simple comparisons between point estimates of exposure and effect levels were common.

The varied approaches to ecological risk assessment used in the case studies are generally consistent with some, but not all, of the principles in EPA's recently published *Framework for Ecological Risk Assessment* (Framework Report) (U.S. EPA, 1992). **However, EPA notes that the cases and peer review comments were developed using "pre-framework" terminology and**

concepts. Thus it is important to understand that while the cases are useful examples of the state of the practice, they should not be regarded as models to be followed. EPA will continue to study concepts found in the Framework Report in future case studies. These case studies and others now being prepared will be used along with the Framework Report to provide a foundation for future Agency-wide guidelines for ecological risk assessment.

This report has two main parts. Part I includes general information on the development and the use of the case studies and listings of key terms and references. Part II includes the 12 case studies that were developed and reviewed for this report.

PART I. CASE STUDIES OVERVIEW

1. INTRODUCTION

The U.S. Environmental Protection Agency's (EPA) Risk Assessment Forum is developing Agency-wide guidance for conducting ecological risk assessments. The Forum initiated this activity in 1990 with a series of meetings to explore significant issues with experts in the field and to meet with individuals from state and other federal agencies to discuss their approaches to ecological risk assessment (U.S. EPA, 1991). Based on these and other discussions, the Forum's initial goals were to: (1) develop the *Framework for Ecological Risk Assessment* (Framework Report) that describes basic principles of ecological risk assessment (U.S. EPA, 1992); (2) prepare a long-range plan for developing future ecological risk assessment guidelines; and (3) develop a set of ecological assessment case studies. The impetus for developing the case studies came from EPA scientists and managers, who felt that such studies could be useful "real world" examples of the ecological risk assessment process.

This case studies report should be useful to EPA regional, laboratory, and headquarters personnel conducting ecological risk assessments, as well as to interested individuals from other federal and state agencies and the general public. In addition, this document is a useful companion volume to EPA's Framework Report.

Case studies are integral to the development and evaluation of ecological risk assessment guidelines. While this report examines other cases from a general risk assessment perspective, future activities will evaluate existing case studies using principles described in the Framework Report. In this way, case studies can serve both as examples of ecological risk assessment and as a means to evaluate recommended procedures.

2. GUIDE TO THE CASE STUDIES

2.1. Background

The case studies presented in part II of this report illustrate several types of ecological assessments. As summarized in table 1, these cases involve:

- studies done under several different federal environmental laws;
- spatial scales ranging from local impacts to regional effects to national impacts;
- different types of stressors, including single chemicals and chemical mixtures, as well as physical stressors such as hydrologic change or sedimentation;
- a variety of ecosystems, including aquatic (freshwater and marine), wetlands, and terrestrial; and
- measurement endpoints reflecting different levels of biological organization, ranging from effects on individual organisms up to and including effects on communities. (See part I, section 3, for definitions of measurement and assessment endpoints.)

This set of case studies is not comprehensive. In particular, it does not include ecological assessments done with very limited time or dollar resources. On the contrary, most of the studies covered in this report cost hundreds of thousands to millions of dollars and took months or years to complete. This point is significant because many evaluations are done quickly based on limited data. The reviewers of these case studies, therefore, recommended that future case studies include examples of ecological risk assessments done with minimal resources.

2.2. Case Study Highlights

This section highlights some common themes and principles gleaned through development and review of these case studies. The section is organized according to the framework for ecological risk assessment provided in the recently published Framework Report (U.S. EPA, 1992) (see figure 1):

- **Problem formulation**, which is a preliminary scoping process;
- **Analysis**, which includes characterization of both ecological effects and exposure; and
- **Risk characterization**, which highlights qualitative and quantitative conclusions, with special emphasis on data limitations and other uncertainties.

Table 1. Case Study Characteristics

No.[a]	Short Title	Relevant Federal Legislation[b]	Spatial Scale of Assessment	Stressor Type[c]	Ecosystem Type[d]	Level of Biological Organization[e]
1	Acidic Deposition	NAPAP	National	SC	A/F	Community
2	Bay Drums	CERCLA/SARA	Local	MC	W	Community
3	Carbofuran	FIFRA	National	SC	T	Individual
4	Coke Plant	CERCLA/ SARA	Local	MC	A/F	Community
5	Commencement Bay	CERCLA/ SARA	Local	MC	A/M	Community
6	Crop Loss	CAA	National	SC	T	Population
7	Quartz Hill	NEPA, CWA	Local	MC, P	A/M	Community
8	Rocky Mountain Arsenal	CERCLA/ SARA	Local	MC	A/F, W, T	Community
9	Kesterson	FMBTA	Local	SC	W	Population
10	Synthetic Pyrethroids	FIFRA	National	SC	A/F	Community
11	Water Quality Criteria	CWA	Local	SC	A/F	Individual
12	Wetlands Loss	NEPA, CWA	Regional	P	W	Community

[a]Numbers 1-12 refer to the sections in part II of this report.

(Notes continued on next page)

[b]Legislation

CAA:	Clean Air Act (1970)
CERCLA/ SARA:	Comprehensive Environmental Response, Compensation, and Liability Act (1980)/Superfund Amendments and Reauthorization Act (1987)
CWA:	Clean Water Act (1977)
FIFRA:	Federal Insecticide, Fungicide, and Rodenticide Act (1972)
FMBTA:	Federal Migratory Bird Treaty Act
NAPAP:	National Acid Precipitation Assessment Program (under the Acid Precipitation Act of 1980)
NEPA:	National Environmental Policy Act (1969)

[c]Stressor types

MC:	Mixture of chemicals
P:	Physical stressor (e.g., suspended solids deposition, hydrologic change)
SC:	Single chemical (case study 10 addresses a group of closely related chemicals)

[d]Ecosystem types

A/F:	Aquatic--freshwater
A/M:	Aquatic--marine or estuarine
T:	Terrestrial
W:	Wetlands

[e]Highest level of biological organization for the measurement endpoints used.

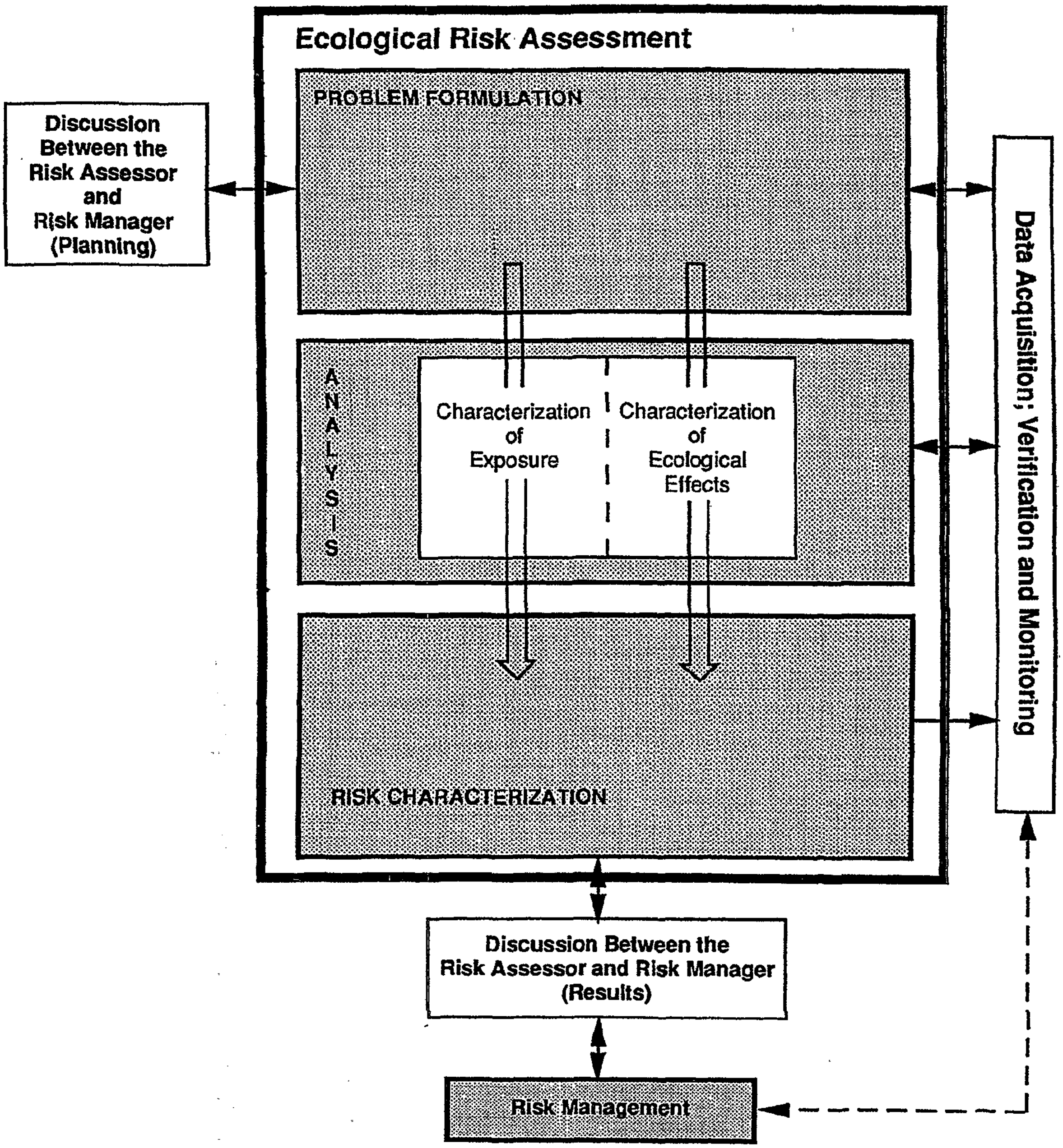

Figure 1. **The framework for ecological risk assessment (U.S. EPA, 1992). The ecological risk assessment framework is the product of a series of workshops and reviews that involved both EPA and outside scientists. While the Framework Report has been a critical first step in developing ecological risk assessment concepts, evolution of the framework concepts is expected and encouraged. One current topic of discussion is the use of the term "exposure." Some scientists feel that "exposure" is associated primarily with chemical stressors and, therefore, does not adequately encompass physical and biological stressors that have great ecological significance. This and other framework issues will be addressed in future substantive guidance.**

2.2.1. Problem Formulation

Problem formulation is an initial planning and scoping process to define the feasibility, scope, and objectives for the ecological risk assessment. This process includes preliminary evaluation of exposure and effects, as well as examination of scientific data and data needs, regulatory issues, and site-specific factors. Problem formulation defines the ecosystems potentially at risk, the stressors, and the measurement and assessment endpoints. This information may then be summarized in a conceptual model, which hypothesizes how the stressor might affect the ecological components (i.e., the individuals, populations, communities, or ecosystems of concern). Although the conceptual model idea was not available to the writers or reviewers of the case studies in this document, elements of the conceptual model were present in most of the case studies.

The Risk Assessment Framework Is Applicable to Physical Stressors

Although most of the case studies deal with exposures to chemicals, a few involve assessments of physical modifications to habitats. These include the change in water elevations in the Wetlands Loss case study and the disposal of mine tailings in the Quartz Hill case study. The stressors, ecological components, and endpoints in these assessments can be discussed within the ecological risk assessment framework.

Thorough Formulation of the Problem and Development of the Scope Are Essential First Steps for a Successful Risk Assessment

The case studies illustrate the importance of clearly defining the goals of the assessment and of developing a scope that is appropriate for achieving those goals within the constraints of available resources and the overall uncertainties of the analyses. Reviewers of the case studies generally indicated that a good assessment is one that provides the information needed to address the hypothesis, question, or management decision at a level appropriate to the decision. To accomplish this, the problem formulation should ensure that the assessment focuses on the stressors, ecological components, and endpoints that are most appropriate for the problem and for making ultimate management decisions. Reviewers observed that this was especially critical when resources are limited by fiscal constraints. The strengths and weaknesses of the case studies seem to originate, in large part, from decisions made during the preliminary planning stages.

It Is Important to Clearly Articulate Management Issues at the Beginning of an Assessment

The Crop Loss case study is a good example of an assessment where the ultimate management issue was clear from the onset; where the stressor, ecological components, and endpoints were clearly defined; and where the design of the study was structured around a clear set of hypotheses amenable to scientific inquiry. This level of clarity was achieved, in part, through meetings and interactions among researchers and others involved with the risk assessment/risk management process. The author and reviewers of this case study stressed the importance of this type of communication for clarifying goals.

The Risk Assessor Should State the Hypotheses Being Evaluated in the Risk Assessment

Many reviewers noted that the problem formulation stage could benefit from clearly stated hypotheses. Such an approach is consistent with the scientific method and would ensure that the risk assessors and managers understand the intent of the analysis. Most of the case studies did not have explicit hypotheses.

The Possibility That Multiple Stressors May Confound the Interpretation of Risks Should Be Considered

Multiple stressors, including combinations of chemical and physical stressors, required consideration in a number of the case studies. Reviewers suggested two important questions to consider when identifying stressor(s), as follows:

- Have all the relevant (or at least the most important) anthropogenic stressors been identified? Were naturally occurring stressors considered?

- Were the criteria used to identify the stressors appropriate and defensible?

Typically, it is easiest to identify appropriate stressor(s) for risk assessments leading to an explicit management decision associated with an individual stressor. Examples include the Synthetic Pyrethroids (pesticides), Acidic Deposition (hydrogen ions), and Crop Loss (ozone) case studies. However, even in such cases the relationship between observed effects in the field and the explicit cause of these effects may be incompletely understood, suggesting that other stressors may have influenced the observed effect.

It is more challenging to select the most relevant or important stressors for assessments involving multiple stressors. The kinds of information typically available to the assessor may include observational data on effects (e.g., fish or birds have died or chemicals are present in environmental media). These observations do not always lead to clear cause-and-effect relationships. In aquatic systems, for example, hypoxic events can lead to fish kills or alterations of benthic habitat unrelated to the presence of toxic chemicals in the sediments. Such effects might also occur in sediment bioassays if elevated levels of naturally occurring ammonia are present. These complications may have been present in the Bay Drums case study but were not explicitly considered as stressors.

In some cases, stressors known to be present may be ignored because of lack of information about the effects of the stressor. For example, the Kesterson case study correctly focused on the element selenium but chose not to look at boron because information on effects was lacking. The reviewers felt that boron could have been important in the overall management decision and that because the scope of this case study was extensive, it would have been appropriate to develop the information necessary to include boron in the assessment.

Selection of Ecological Components Should Be Based on Ecological Principles and Human Values

The case studies illustrate that several different ecological components may be selected, depending on the focus of the study. These components could range from one or more species (e.g., Rocky Mountain Arsenal case study) to different communities (e.g., Wetlands Loss case study) or ecosystems. Selection of components is typically based on several criteria, although these criteria were not often explicitly stated in the case studies. Factors that influenced the selection of ecological components include:

- the nature of the stressor and the potential for the stressor to interact with the ecological component (as illustrated by use of both aquatic biota and waterfowl in the Synthetic Pyrethroids case study);
- the value of the ecological component from an ecological or ecosystem perspective (as illustrated by the focus on trees in the Wetlands Loss case study); and
- the value of the ecological component from a human perspective. Examples from the case studies include:

  (1) rare, threatened, or endangered species such as the bald eagle in the Rocky Mountain Arsenal case study and the San Joaquin Kit Fox in the Kesterson case study;

  (2) species of commercial importance such as crop species in the Crop Loss case study, English sole in the Commencement Bay case study, and salmon in the Quartz Hill case study; and

  (3) species of recreational importance such as freshwater fish in the Acidic Deposition and Water Quality Criteria case studies and waterfowl in the Synthetic Pyrethroids case study.

The Possibility of Indirect Effects Should Be Considered When Ecological Components Are Selected

The Synthetic Pyrethroids case study illustrates the importance of considering ecological components that are *indirectly* affected by the stressor. In this case study, waterfowl were included as an ecological component, not because of any direct risk but because they might be affected indirectly by a reduced prey base. Indirect or secondary effects also were considered in the Wetlands Loss case study. This case study evaluated the direct effects of an increase in water level (the stressor) on trees and also included wildlife species dependent on the vegetation as ecological components of concern.

Some ecological components are considered especially important because other components depend on them for habitat or food. These include foundation or keystone species. In some cases, the presence of such components is easily recognized. In others, however, the relationships among species may not be understood. Reviewers of the case studies felt it was important to identify these key ecological components. Examples of such components include specific trees in the Wetlands Loss case study, wetlands in the Bay Drums case study, and benthic invertebrate communities in several of the case studies.

Reviewers of the Acidic Deposition case study felt that this assessment may have focused too narrowly on survival of fish species; the prey base (e.g., zooplankton) was not specifically identified as a receptor.

Defining Assessment and Measurement Endpoints and Their Interrelationships Is Essential

There was some confusion among the case study authors (and reviewers) over the meaning of the terms "assessment endpoint" and "measurement endpoint" (see part I, section 3 of this report for definitions). The case studies illustrated that defining the assessment endpoint and selecting appropriate measurement endpoints are critical. Selection of assessment endpoints is another area that requires discussions with the risk manager and others. The Acidic Deposition and Crop Loss case studies provide good examples of well-defined assessment endpoints that are clearly related to measurement endpoints.

Multiple Measurement Endpoints Help Evaluate "Overall System Integrity"

When the assessment endpoint was the overall health or integrity of the system, reviewers found that case studies using multiple measurement endpoints were better able to assess risks than those using only single measurement endpoints. One example is the Commencement Bay case study, which includes field and laboratory observations of fish and invertebrate species as well as chemical measurements. In addition, this case study uses comparisons between reference and affected areas. The use of several laboratory toxicity test and field observation methods was critical to providing a more nearly complete picture of the nature of exposure and effects in this case study.

2.2.2. Analysis

Analysis includes the technical evaluation of data on both potential exposure to stressors (characterization of exposure) and the effects of stressors (characterization of ecological effects).

Characterization of exposure involves predicting or measuring the spatial and temporal distribution of a stressor and its co-occurrence or contact with the ecological components of concern, while characterization of ecological effects involves identifying and quantifying the effects elicited by a stressor and, to the extent possible, evaluating cause-and-effect relationships.

Characterization of Exposure

Selection of Exposure Models Depends on the Purpose of the Risk Assessment and Available Resources

Estimating or representing the exposure regime appears to be one of the most technically challenging tasks in an ecological risk assessment. Most of the case studies rely on simple models or measurements, and the reviewers frequently commented on the apparent oversimplifications in characterizing exposure.

A few of the case studies use simulation or complex fate-and-transport models in an effort to represent the complex processes in real-world exposure regimes. The selection of appropriate models is a matter of scientific judgment combined with an appreciation of the overall question to be addressed, and a recognition of the available resources.

Fate-and-transport models range from simple representations to complex numerical models. For example, in the Synthetic Pyrethroids case study, simple algebraic models are used to calculate migration of the pesticide in runoff and in drift. In contrast, the Acidic Deposition case study uses a combination of sophisticated fate-and-transport models along with field measurements to estimate exposure. Reviewers observed that fate-and-transport models should be appropriate for the goals and scope of the study.

Simulation models have been developed to represent exposure regimes and to relate these to certain systemwide effects. The FORFLO model used in the Wetlands Loss case study is an example of a simulation model that attempts to represent an exposure regime for a physical stressor (change in water level). Hydrologic characteristics of the environment are incorporated into the model.

Validation of Models Is Important for Reducing the Uncertainty of Exposure Estimates

Reviewers of the case studies that used models noted the importance of verification and "reality checks" on the exposure estimates. The Synthetic Pyrethroids case study uses simple models to project the transport of the chemicals and resultant concentrations in surface waters. However, no field data are provided to support these estimates. Additional field studies are being planned so these data may be generated in the future.

The Acidic Deposition case study includes verification of model estimates. The procedures include making hindcasts of historical changes in surface water chemistry as a function of historical changes in acidic deposition rates. These hindcasts are compared with paleolimnological reconstructions using algal fossils. A watershed chemistry model is calibrated using several watersheds, and then confirmed through blind simulations using only input data. Analyses are also conducted to identify those input variables and parameters to which the models are sensitive. Reviewers felt that such sensitivity analyses should be more widely used.

When Available, Life History and Behavioral Information Can Be Useful in Characterizing Exposure

The Kesterson case study illustrates how life history and behavioral information can be utilized as part of exposure characterization. For each species in the Kesterson case study, the scientific literature was reviewed to quantify food habits, and dietary preferences were summarized by life stage, sex, and season, as appropriate. The home range of each species was estimated from literature values. Diet factors were used to model the fraction of the whole diet of each organism contributed by each compartment of the simplified selenium transfer diagram. Diet factors were based on species' food preferences as described in the literature, on their home range relative to the size of the Kesterson Reservoir area, and on the relative abundance of different types of prey species. For example, for species that consumed aquatic invertebrates, the relative abundance of herbivorous and carnivorous aquatic invertebrates at the Kesterson Reservoir was used to specify the composition of these organisms in the diet.

Food Chain and Pathway Analyses Can Be Useful for Evaluating Indirect Effects

Good examples of food chain and pathway analyses may be found in the Rocky Mountain Arsenal and Kesterson case studies. In addition to providing exposure-relevant information, evaluating food chain relationships can reveal potential indirect or secondary effects on higher trophic levels due to reduction in the prey base. The Synthetic Pyrethroids case study considers such secondary effects.

Food chain and pathway analyses have been used in case studies where wildlife species (including endangered species) have been selected as the ecological components of concern. For example, a major feature of the Rocky Mountain Arsenal risk assessment is the development and use of a pathways model to establish a quantitative relationship between concentrations of bioaccumulative contaminants in abiotic media and concentrations at different trophic levels in aquatic and terrestrial food webs.

For the Kesterson case study, seven representative wildlife species in the area were selected. Food chain exposure was considered the most important pathway for exposure of fish and wildlife to selenium, and detailed food chain exposure diagrams for each of the selected species were developed into simplified selenium transfer models. These models were used with transfer factors derived from chemical measurements in media and tissues and a Monte Carlo simulation technique to estimate the probability distribution of selenium concentrations in the diets of the key species under each of the three remedial alternatives.

Similar Exposure Principles May Be Applicable to Physical and Chemical Stressors

The Wetlands Loss case study illustrates that the exposure associated with a physical stressor (changes in water elevation) can be characterized using principles similar to those employed in more familiar cases involving chemical stressors. In this case study, exposure-effects relationships were developed between the physical stressor and habitat alterations associated with changes in forest vegetation.

Characterization of Ecological Effects

Information Needs Should Be Determined by the Purpose and Scope of the Risk Assessment

The case studies illustrate that ecological effects may be characterized using the various kinds of information given below.

- Literature Values, Criteria, or Guidelines. A few case studies (e.g., Quartz Hill) rely almost exclusively on existing literature values, criteria, or guidelines, while others (Kesterson, Bay Drums, and Wetlands Loss) incorporate such information into the assessment along with the results of laboratory and field tests.

- Laboratory and In-Field Exposure-Response Studies. Laboratory studies include acute and chronic toxicity tests in which organisms are exposed to individual or multiple stressors (e.g., complex mixtures). Data from both laboratory and field studies are included in the Bay Drums and Synthetic Pyrethroids case studies. The Crop Loss case study uses field enclosures of crop species to develop concentration-response data for ozone exposures.

- Field Studies and Surveys. The Coke Plant and Commencement Bay case studies include examples of these approaches.

The reviewers commented on the advantages and disadvantages of using different methodologies and information sources. A key point is that the information or methods used should be appropriate for the assessment. In general, the reviewers found that using a suite of methods (literature values, bioassays, field studies) provided a more complete characterization of ecological effects than relying on a single measure or literature value.

Site-Specific Criteria Are Useful Tools

Many case studies used criteria or benchmarks, i.e., chemical concentrations in environmental media (food, water, soil, sediment), below which minor or no effects are anticipated for a particular measurement endpoint. These measures can be improved by taking into account important site-specific factors related to the toxicity or other effects of the stressor upon the selected ecological components. Site-specific factors may not be accounted for in national criteria.

The Water Quality Criteria case study explains how site-specific cadmium criteria for protecting aquatic organisms in the St. Louis River were derived. Another example is the Apparent Effects Threshold (AET) values developed for chemicals in sediments for the Commencement Bay case study. AET values are site-specific concentration limits below which no effects are expected. They are derived by combining information from field studies and laboratory sediment bioassays.

Criteria Should Be Established for Selecting Appropriate Reference Areas for Field Studies

Field studies sometimes compare potentially affected areas with reference sites as a basis for characterizing effects. This approach is used in the Bay Drums and Coke Plant case studies, but the comparison between reference and test sites was difficult to interpret because the reference areas themselves had potential stressor impacts. An additional reference area had to be included in the Commencement Bay case study because of difficulties with the originally selected reference site.

Reviewers of case studies that included comparisons between affected and reference areas noted the importance of developing synoptic information on the observed ecological effects and on the presence and magnitude of the stressors. A clear set of selection criteria should be developed for reference areas, and selection should be based on these criteria. When available reference areas have limitations, these limitations should be recognized at the onset. In such cases, it may be necessary to use more than one reference area.

Establishing Causality Is Complicated by Multiple Stressors

In some case studies, it is difficult to link specific stressors with observed ecological effects because of confounding factors such as the presence of mixtures of chemicals, or a combination of chemical and nonchemical stressors. For example, it was difficult to link stressors and effects in the Commencement Bay case study because the sediments contained numerous chemicals. In the Bay Drums case study, physical factors (wetland alterations) and natural biochemical factors (anaerobic sediments) confounded the interpretation of some of the field and laboratory effects data.

Ecological Effects Models, if Applied Correctly, Are Useful for Relating Effects to Stressors

Several case studies illustrate how models can be used effectively to relate ecological effects and stressors.

- The Wetlands Loss case study links the FORFLO model, which predicts changes in forest vegetation caused by changes in water level, to Habitat Suitability Indices (HSI), which relate changes in wildlife to alterations in forest vegetation.
- The Crop Loss case study develops a model that relates plant yield to ozone exposure.
- The Acidic Deposition case study models the relationship between water chemistry variables (i.e., pH and concentrations of aluminum and calcium ions) and the probability of the presence or absence of fish for selected species.

2.2.3. Risk Characterization

Risk characterization uses the results of the exposure and ecological effects analyses to evaluate the likelihood that adverse ecological effects are occurring or will occur in association with exposure to a stressor. A risk characterization highlights summaries of the assumptions, scientific uncertainties, and strengths and weaknesses of the analyses. Finally, a risk characterization discusses the ecological significance of the risks with consideration of the types and magnitudes of the effects, their spatial and temporal patterns, and the likelihood of recovery.

Risks Can Be Characterized Both Qualitatively and Quantitatively

The Framework Report (U.S. EPA, 1992) notes that risk characterization may be qualitative or quantitative and that it frequently relies heavily on scientific judgment. Opinions of the case study reviewers about what risk characterization should be varied. Some felt that risk characterization applies only to situations where predictive, probabilistic statements can be made about future events. Others held a broader view, in which risk characterization includes either qualitative or quantitative statements of risk and involves evaluating the causal relationship between stressors and effects for existing situations as well as predicting the risk of future events.

Qualitative analyses are used in the Quartz Hill and Synthetic Pyrethroids case studies. The Quartz Hill case study compares the potential effects of mine tailings disposal in two fjords to determine which fjord would be more at risk. Reviewers were concerned that the absolute risks of disposal were not considered, but did agree that the comparative analysis was useful for assessing the relative risk of the mine tailings to the two fjords.

The Synthetic Pyrethroids case study includes a qualitative assessment of the potential secondary effects of a reduced prey base on waterfowl. Synthetic pyrethroids are known to be toxic to aquatic life that waterfowl rely upon for food, and the reviewers agreed that secondary effects on waterfowl were possible. However, a quantitative analysis was not conducted, and the reviewers felt that the qualitative analysis of the effects related to prey base was one of the weaker parts of the case study.

The Toxicity Quotient Method Is Frequently Used

The Toxicity Quotient Method is a simple method used in several case studies to compare exposure levels with benchmark or criteria effect levels. If the ratio exceeds "1," some potential for risk is presumed. In addition to the results of Quotient Method comparisons, a complete risk characterization includes the scientific uncertainties and assumptions underlying the assessment.

The Bay Drums, Coke Plant, and Water Quality Criteria case studies all compare chemical concentrations in environmental media with benchmark concentrations. For water samples, the benchmark concentrations were often the ambient water quality criteria (AWQC) or a screening value obtained from the relevant EPA AWQC document. In the Bay Drums case study, chemical concentrations in sediments are compared with toxic sediment concentrations reported in the literature.

The Carbofuran case study uses the Quotient Method in a somewhat different manner. Risks are evaluated, in part, by comparing the estimated levels of carbofuran granules per square foot of surface soil with the granule dose estimated to kill 50 percent of exposed birds (LD_{50}).

The Rocky Mountain Arsenal case study also uses the Quotient Method to evaluate risks associated with dietary exposure. This case study uses an exposure pathway analysis to estimate numerical criteria for potential exposure pathways, including dietary exposure (bioaccumulation) and surface water ingestion. (A calibration and validation process is being developed to reduce uncertainties in this approach.) The Kesterson case study uses a similar pathway analysis to determine levels of selenium in sediments that would cause adverse effects in selected fish and wildlife species.

Exposure-Response Models Offer Certain Advantages

The major advantage of exposure-response models is that risks and associated uncertainties can be quantified and the results presented to risk managers in a form that allows comparison of alternatives. Forecasted conditions can give more meaningful information to managers than the simple numerical comparisons of the Quotient

Method. For example, in the Crop Loss case study, reductions (losses) in commercial crop yields can be related to specific ozone levels which, in turn, can be considered in formulating air quality standards. The major disadvantage of using exposure-response models is that adequate validation and sensitivity analyses are frequently not available.

Risks May Be Characterized at Various Levels of Ecological Organization

The complexity of risk characterization increases with increasing levels of ecological organization. For individuals of a species, it is possible to determine if *some* will be at risk as a result of a particular exposure. To estimate population-level risks, some combination of field studies or population models is needed. Although population models are available, they were generally not used in the case studies. Instead, most of the case studies inferred that population-level effects could occur if there were risks to individuals. In the Carbofuran case study, for example, evidence of mortality of individual birds was considered sufficient to demonstrate adverse effects even without a formal assessment of population-level risks.

Risks to communities were assessed by considering species representative of various trophic groups, taxa, or habitats. While several case studies (e.g., Bay Drums, Commencement Bay, Coke Plant) evaluate changes in the structure of benthic macroinvertebrate assemblages, interactions among the organisms are not determined as frequently. The Synthetic Pyrethroids case study does consider reduction of the prey base as a possible risk to waterfowl, and the Wetlands Loss case study uses models to relate predicted changes in vegetation to risks to birds and mammals.

Ecosystem-level risks, which might involve changes in functional processes such as productivity, nutrient cycling, or decomposition, are seldom considered. The Acidic Deposition case study comes closest to ecosystem-level risk characterization. The ecological components for this case study include freshwater streams and lakes.

Some reviewers felt that certain case studies did not consider all the interactions that may be present within an ecosystem and that the case studies were too narrowly focused. Discussion of this point led to the question "What is important?" It was recognized that this question must be addressed from two sides:

1. What is important from a management standpoint? What information is needed to reach a sound decision?

2. What is important from an ecological standpoint? What information must be brought to the attention of the risk manager to aid in an informed decision?

It was clear from the discussions that ecological risk assessment does not always mean ecosystem risk assessment. The selected ecological components and methodologies must be appropriate to the ultimate risk management decision.

Risk Assessors Should Provide a Complete Picture of the Risk Assessment

The manner in which results are presented to the risk manager or to those affected by the risk management decision can affect the perception of the risk. For example, the Acidic Deposition case study presents the percentages of lakes and streams at risk. Reviewers noted that this has a different impact on the reader than if the absolute number of affected lakes and streams are presented. The risk assessor should know who the users of the risk assessment will be and should present the results in a manner that is appropriate for them. To provide a complete picture and accommodate users with varying perspectives, it may be necessary to present results in a variety of formats.

3. KEY TERMS (U.S. EPA, 1992)

assessment endpoint—An explicit expression of the environmental value that is to be protected.

characterization of ecological effects—A portion of the analysis phase of ecological risk assessment that evaluates the ability of a stressor to cause adverse effects under a particular set of circumstances.

characterization of exposure—A portion of the analysis phase of ecological risk assessment that evaluates the interaction of the stressor with one or more ecological components. Exposure can be expressed as co-occurrence, or contact, depending on the stressor and ecological component involved.

conceptual model—The conceptual model describes a series of working hypotheses of how the stressor might affect ecological components. The conceptual model also describes the ecosystem potentially at risk, the relationship between measurement and assessment endpoints, and exposure scenarios.

ecological component—Any part of an ecological system, including individuals, populations, communities, and the ecosystem itself.

ecological risk assessment—The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors.

exposure—Co-occurrence of or contact between a stressor and an ecological component.

measurement endpoint—A measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint. Measurement endpoints are often expressed as the statistical or arithmetic summaries of the observations that comprise the measurement.

risk characterization—A phase of ecological risk assessment that integrates the results of the exposure and ecological effects analyses to evaluate the likelihood of adverse ecological effects associated with exposure to a stressor. The ecological significance of the adverse effects is discussed, including consideration of the types and magnitudes of the effects, their spatial and temporal patterns, and the likelihood of recovery.

stressor—Any physical, chemical, or biological entity that can induce an adverse response.

4. REFERENCES

U.S. Environmental Protection Agency. (1991) *Summary report on issues in ecological risk assessment*. Risk Assessment Forum, Washington, DC. EPA 625/3-91/018.

U.S. Environmental Protection Agency. (1992) *Framework for ecological risk assessment*. Risk Assessment Forum, Washington, DC. EPA 630/R-92/001.

PART II. THE CASE STUDIES

The case studies included in this section follow the format shown in the box on the right. When reading the case studies, it is important to keep several points in mind:

> **Case Study Format**
>
> - Abstract. The abstract summarizes the major conclusions, strengths, and limitations of the case study.
> - Risk Assessment Approach. This section clarifies any differences between the ecological risk assessment approach used in the case study and the general process described in the Framework Report.
> - Statutory and Regulatory Background. The statutory requirements for the study are described along with any pertinent regulatory background information.
> - Case Study Description. This contains the technical description of the case study, organized according to the phases of ecological risk assessment described in the Framework Report: problem formulation, analysis (characterization of exposure and characterization of ecological effects), and risk characterization. A comment box is included at the end of each major section.
> - References.

- **The original case studies were not developed as risk assessments as defined in the Framework Report.** The Framework Report was not available when the case studies were conducted, written, or reviewed. Fortunately, the overall concepts of ecological risk assessment applied by the authors and reviewers were compatible with the broad principles (if not the details) described in the Framework Report. EPA notes that the case studies are often partial risk assessments that focus on available information without discussing other relevant considerations such as the uncertainties defined by a limited data base.

  At the workshops, each case study was evaluated as to whether it (1) effectively addressed the generally accepted components of an ecological risk assessment, or (2) addressed some but not all of these components or, instead, (3) provided an alternative approach to assessing ecological effects.

- **The strengths and limitations of each case study are highlighted in the comment boxes.** Comments made by both the peer reviewers and the case study authors are included.

- **The authors who compiled the case studies did not necessarily conduct the research upon which the case studies are based.** References to the original research are provided in each case study.

General characteristics of the case studies are summarized in table 1 (in part I), and a list of selected ecological risk assessment techniques that were used in the case studies is provided in table 2. The list is not meant to be comprehensive. Rather, it provides examples of different methods and models that were used in the various phases of the ecological risk assessment process. The application of these techniques may be reviewed by referring to the case studies in which they are used. Case studies are referenced by the section of this report in which they appear. (The corresponding titles of the case studies are given in table 1.)

Table 2. Selected Case Study Methods and Models

METHOD	DESCRIPTION	REFERENCE	CASE STUDY REPORT SECTION (PART II)
CHARACTERIZATION OF EXPOSURE			
Estimating stressor levels in the environment			
Aquatic			
SWRRB	Simulator for Water Resources in Rural Basins	Arnold et al., 1990	10
EXAMS	Exposure Analysis Modelling System	Burns, 1990	10
Steady-State Oceanographic Model	Based on distribution of natural and anthropogenic conditions using Monte Carlo simulations		7
MAGIC, ILWAS	Dynamic acidification models	Thornton et al., 1990	1
Mass Balance Study	Prediction of sediment concentrations in relation to source loading, sedimentation rates and mixing, biodegradation and diffusion		5
Hydrological Assessment	Surface and ground-water flow models	Standard EPA Protocols	2
Acid Deposition Models			
RADM	Regional Acid Deposition Model	Thornton et al., 1990	1
NADP/NTN	National Acid Deposition Program/ National Trends Network		1
Steady-State Models	Relate deposition to lake acid neutralizing capacity and pH	Thornton et al., 1990	1
Empirical Acidification Models	Relate deposition to lake acid neutralizing capacity and pH	Thornton et al., 1990	1
Sediment Chemistry			5
Terrestrial			
Kriging	National Ozone Data Base--300 sites (SAROAD)	Lefohn et al., 1987	6
	Deposition surface for loading to individual sites	NAPAP, 1990	1
Normograph	Estimation of pesticide residues found in wildlife	Urban and Cook, 1986	10

Table 2. Selected Case Study Methods and Models (continued)

METHOD	DESCRIPTION	REFERENCE	CASE STUDY REPORT SECTION (PART II)
Multimedia			
Exposure Algorithms	Determination of estimated environmental concentrations (EEC) due to pesticide runoff and drift	Unpublished	10
Estimating stressor levels in biota			
Aquatic			
Bioaccumulation	Organics	U.S. EPA, 1985	5
	Metals from mine tailings	U.S. EPA, 1988a	5
	Selenium	Saiki, 1986 Ohlendorf et al., 1986b	9
	Organics and inorganics in fish and invertebrates	Standard EPA Protocols	2
Terrestrial			
Normograph	Calculation of pesticide residues in terrestrial food webs	Urban and Cook, 1986	10
Multimedia			
Food Chain Models	Calculated selenium transfer factors from experimental data and Monte Carlo simulations	USBR, 1986a	9
CHARACTERIZATION OF EFFECTS			
Aquatic			
Organism/Population Level			
Laboratory Toxicity Tests	Pesticide assessment protocols	Preston and Hitch, 1982	10
	Sediment bioassays with amphipods, oyster larvae, and Microtox	Williams et al., 1986	5
	Chemical mixtures	Standard EPA Protocols	2
	Clam burrowing behavior	U.S. EPA, 1988a	7
	Phytoplankton growth	U.S. EPA, 1988a	7
	Critical pH levels	Baker et al., 1990 Turner et al., 1990	1
	Survival, growth, and reproduction of resident fish species	Spehar et al., 1985 Carlson et al., 1984	11

Table 2. Selected Case Study Methods and Models (continued)

METHOD	DESCRIPTION	REFERENCE	CASE STUDY REPORT SECTION (PART II)
Condition Factors	Based on weight and length of individual fish		4
Fish Histopathology	Examination of livers	U.S. EPA, 1985	5
	Examination for tumors, lesions, and other anomalies		4
Fish Population Model	Logistic regression equation for presence/absence of fish	Baker et al., 1990	1
Community/Ecosystem Level			
Aquatic Mesocosms	Impacts on invertebrate communities	U.S. EPA, 1990	10
Pond Studies	Impacts on fish and invertebrates	U.S. EPA, 1990, 1991	10
WET Model	Wetland Evaluation Technique for assessing function and values of wetlands	Adamus et al., 1987	2
Macroinvertebrate Community	Species diversity index: Sorrenson's quotient and similarity		2
	Abundance, species richness, Bray-Curtis similarity	U.S. EPA, 1985	5
Fish, Macroinvertebrate, and Plankton Communities		Davis and Lathrop, 1991	4
Recolonization	Effects of mine tailings	U.S. EPA, 1988a	7
Terrestrial			
Organism/Population Level			
Avian Laboratory Toxicity Tests	Effects of carbofuran	Urban and Cook, 1986	3
	Pesticide assessment protocols	Preston and Hitch, 1982	10
	Hatchability, deformities, and mortality	Ohlendorf et al., 1986	9
Pathological Examinations			9
Field Observations	Rare and endangered species	Not included	2
Open-Top Field Chambers	Foliar injury, growth, reproduction, yield, and physiological responses	Heck et al., 1991 Heagle and Heck, 1980	6
Acetylcholinesterase Activity		Robinson et al., 1988	8

Table 2. Selected Case Study Methods and Models (continued)

METHOD	DESCRIPTION	REFERENCE	CASE STUDY REPORT SECTION (PART II)
Samples of Chance	Necropsies on dead or dying organisms		8
Earthworm Population Studies			8
Avian Reproductive Success	Percent nests hatched and fledged, egg weight, volume, dimensions, and thickness	ESE, 1989	8
Population Studies	Species occurrence, population density, and age-class structure: snails, earthworms, vegetation, ducks and coots, and prairie dogs		8
Multimedia			
Organism/Population Level			
Habitat Suitability Index	HSI model for present and future capability of a site to provide basic habitat requirements for wildlife indicator species	USFWS, 1981	12
Community/Ecosystem Level			
FORFLO	Bottomland forest succession model that simulates the growth, reproduction, and competition of a mixed-tree species forest stand	Pearlstine et al., 1985	12
RISK CHARACTERIZATION			
Quotient Method	Effects concentration/expected exposure concentration	Dewitt, 1966	10 3
Ambient Water Quality Criteria	Defining extent of area exceeding AWQC		4 7
	Recalculation, indicator species, and resident species procedures were used to modify national cadmium criteria	Spehar and Carlson, 1984a, b	11
Reference Indices	Elevation above reference indices	U.S. EPA, 1989b USBR, 1986	5 9
Apparent Effects Threshold	Based on amphipod mortality, oyster larvae abnormalities, and benthic macroinvertebrate taxa abundance	U.S. EPA, 1989b USBR, 1988	5 9
Weibull Model	Predicted yield losses based on ozone exposure	Somerville et al., 1989, 1990	6

Table 2. Selected Case Study Methods and Models (continued)

METHOD	DESCRIPTION	REFERENCE	CASE STUDY REPORT SECTION (PART II)
Pathways Analysis	Criteria developed based on characterization of effects and observed environmental concentrations using a modified food chain model	Fordham and Reagan, 1991	8
Wasteload Allocations	Setting limits on wastewater loads and nonpoint source allocations	U.S. EPA, 1991	8
Indicator Species Procedure	Takes into account factors that affect the bioavailability and/or toxicity of stressors in characterizing risk		8
Weight-of-Evidence Approach	Multiplicity of evidence supporting hypothesis at a site where many potential stressors existed		2
	Process of elimination of alternative stressors and corroborative support for acid deposition	NAPAP, 1990	1

SECTION ONE

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

EFFECTS OF ACIDIC DEPOSITION ON AQUATIC ECOSYSTEMS— A REGIONAL PROBLEM

AUTHOR AND REVIEWERS

AUTHOR

Kent W. Thornton
FTN Associates, Ltd.
Little Rock, AR

REVIEWERS

Thomas M. Frost (Lead Reviewer)
Center for Limnology
University of Wisconsin
Madison, WI

Nancy A. Bryant
ENSR Consulting & Engineering
Acton, MA

Kenneth L. Dickson
Institute of Applied Sciences
University of North Texas
Denton, TX

Judy L. Meyer
Institute of Ecology
University of Georgia
Athens, GA

Douglas P. Ormrod
Office of Graduate Studies
University of Guelph
Guelph, Ontario, Canada

Randall S. Wentsel
Toxicology Division
U.S. Army Chemical Research,
Development, and Engineering Center
Aberdeen Proving Grounds, MD

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LIST OF ACRONYMS

AERP	Aquatic Effects Research Program
ANC	acid-neutralizing capacity
ASI	acidic stress index
CAAA	Clean Air Act Amendments
DOC	dissolved organic carbon
EPA	U.S. Environmental Protection Agency
ILWAS	Integrated Lake/Watershed Acidification Study
MAGIC	Model of Acidification of Ground Water in Catchments
NADP/NTN	National Acid Deposition Program/National Trends Network
NAPAP	National Acid Precipitation Assessment Program
NSWS	National Surface Water Survey
RADM	Regional Acid Deposition Model
SBRP	Southern Blue Ridge Province
SOS/T	State-of-Science/Technology
USGS	U.S. Geological Survey

ABSTRACT

The National Acid Precipitation Assessment Program (NAPAP) is a 10-year congressionally mandated research program to assess the effects of acidic deposition on the environment. The U.S. Environmental Protection Agency (EPA) directed the Aquatic Effects Research Program (AERP) to assess the regional effects of acidic deposition on lakes and streams in the United States. The AERP designed and implemented a strategic assessment plan that implicitly followed the steps in risk assessment. A National Surface Water Survey (NSWS) determined the percentage and extent of lakes and streams that were acidic or potentially susceptible to acidic deposition (problem formulation). It indicated that not all aquatic systems in the surveyed regions were susceptible to acidic deposition. For regions with subpopulations of aquatic systems at risk, total acidic deposition estimates were determined from field measurements and model simulations (e.g., wet deposition estimates from the National Acid Deposition Program [NADP] monitoring network and dry deposition estimates from the Regional Acid Deposition Model [RADM]) (characterization of exposure). Watershed chemistry models simulated the effects of acidic deposition on watershed-lake/stream chemistry, and fish-response models determined the effects of surface-water acidification on fish loss (characterization of ecological effects). The final step was to assess risk to aquatic systems (both changes in water chemistry and in fish response) projected to occur under alternative sulfur dioxide emission-control and no-emission-control scenarios (risk characterization). These scenarios involved coupling emission models, the RADM, watershed-lake/stream chemistry models, and fish-response models with associated error bounds. This effort represents one approach to addressing a regional problem in an ecological risk assessment framework.

1.1. RISK ASSESSMENT APPROACH

Although the risk assessment framework was not used in designing the National Acid Precipitation Assessment Program (NAPAP), all framework elements are included in the program's various phases (figure 1-1). Therefore, this case study can serve as a fairly complete example of the framework's application. In this study, the ecological components are lakes and streams that may be acidified. The stressor is acidic deposition.

1.2. STATUTORY AND REGULATORY BACKGROUND

Congress authorized the NAPAP under the Acid Precipitation Act of 1980 (P.L. 96-294, Title VII) from concern that acidic deposition might have adverse effects on aquatic systems, forests, agricultural crops, construction materials, cultural resources, atmospheric visibility, and human health. NAPAP was given the statutory responsibility to prepare comprehensive scientific, technological, and economic information to assist legislators and other decision-makers in developing policies to control acidic deposition. NAPAP, charged with conducting a 10-year program of research and assessment, investigated the causes and effects of acidic deposition and analyzed alternative strategies to control or mitigate these effects. The program was responsible for coordinating and collaborating with other relevant foreign and domestic research activities. The act required the NAPAP to provide the President, Congress, and the public with annual reports and a final 1990 Integrated Assessment. NAPAP was not to recommend specific emission-control levels or targets; rather, it was to provide Congress, the President, and federal and state policy officials with relevant information to use in formulating policy, legislation, and regulations.

1.3. CASE STUDY DESCRIPTION

This case study is based largely on analyses performed in conjunction with the NAPAP 1990 Integrated Assessment to evaluate the effects of acidic deposition on aquatic systems. The research was conducted by the Aquatic Effects Research Program (AERP). See appendix A for a list of scientific contributors. The U.S. Environmental Protection Agency (EPA) was the lead agency responsible for the Aquatic Effects Task Group in NAPAP. Four primary policy questions guided the research and assessment efforts: (1) How extensive is the damage to aquatic resources due to current and historical levels of acidic deposition; (2) what is the anticipated future damage to these resources; (3) what levels of damage to sensitive surface waters are associated with various rates of acidic deposition; and (4) what is the rate of change or recovery of affected systems if acidic deposition rates decrease?

The general approach used in this assessment is shown schematically in figures 1-1 and 1-2 and described in the following sections.

1.3.1. Problem Formulation

In the 1960s and 1970s, Scandinavian scientists collected evidence of acidity in precipitation and watershed runoff that was contributing to the acidification of lakes and streams (Almer et al., 1974; Gjessing et al., 1976; Hultberg, 1977; Henriksen, 1979; Okland, 1979; Drablos and Tollan, 1980). Parallel studies in the United States, Canada, the United Kingdom, and Central Europe

Figure 1-1. Structure of Analysis for Effects of Acid Deposition

PROBLEM FORMULATION

Stressors: hydrogen ion derived from nitric and sulfuric acids.

Ecological Components: sensitive aquatic ecosystems including lakes and streams.

Endpoints: assessment endpoint is fish survival. Measurement endpoint includes water chemistry (e.g., ANC, pH, DOC, Ca, Al).

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
Models of acid deposition and watershed chemistry were used to predict surface water conditions for alternative emission scenarios.	Laboratory and field studies were used to examine fish response to acidity. An acidic stress index (ASI) was developed.

RISK CHARACTERIZATION

Emission, atmospheric, watershed, and fish response models were coupled to provide estimates of the proportion of water bodies that may be adversely affected by acidic deposition under various scenarios. Uncertainty was quantitatively evaluated using Monte Carlo and other methods.

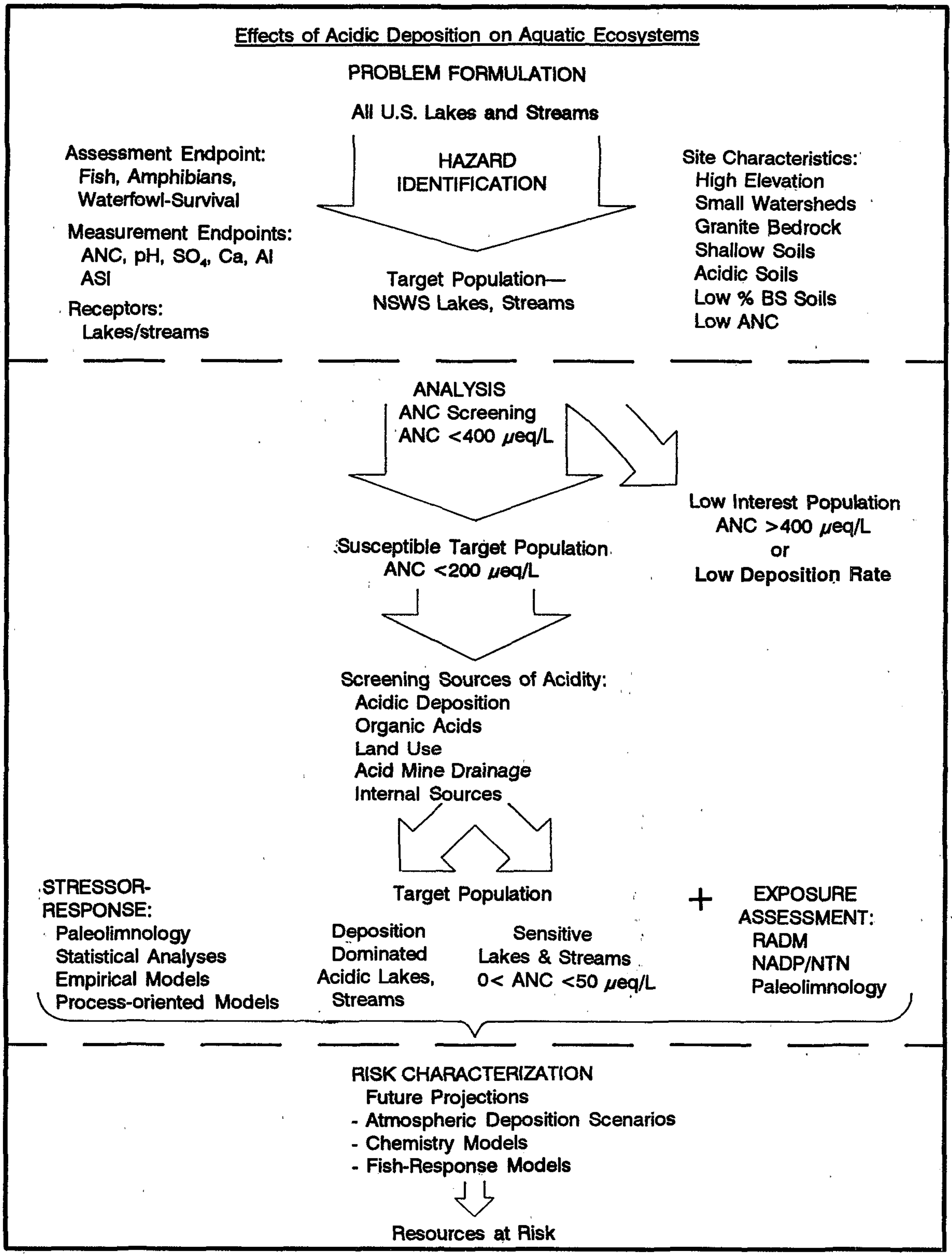


Figure 1-2. Steps in a regional ecological risk assessment

during this period confirmed the existence of acidic deposition and its potential link to surface-water acidification (Beamish and Harvey, 1972; Cogbill and Likens, 1974; Beamish et al., 1975; Davis et al., 1980; Schindler et al., 1980; Burns et al., 1981; Altshuller and Linthurst, 1984). The apparent loss of sport fish populations in lakes in areas of high acidic deposition, such as the Adirondacks, added to mounting concerns (Schofield, 1976a, b).

Stressors. The primary components of acidic deposition that affect surface-water acidification are sulfuric and nitric acids. The largest sources of sulfur compounds linked to acidic deposition are coal and oil combustion in electric power-generating and industrial facilities. Sulfate is the dominant acidic anion in wet deposition and, therefore, its effects received the greatest study (National Academy of Sciences, 1984; National Research Council, 1986). Sulfuric and nitric acids are strong acids that can cause changes in the hydrogen ion concentration (pH) in surface waters, with direct and indirect deleterious effects on aquatic organisms. Low pH (i.e., high hydrogen ion) can be directly toxic to aquatic species, but also can indirectly affect the ecosystem by causing shifts in the food web (e.g., the loss of a prey species) (Schindler, 1988). Low pH also can mobilize metals, such as aluminum, that are toxic to aquatic species, especially fish.

Ecological Components. The ecological components examined in this study include "sensitive aquatic ecosystems." A measure used in the study to identify and characterize such systems is acid-neutralizing capacity (ANC), the capacity of a system to buffer itself against changes in pH. The term alkalinity refers primarily to the neutralization of acids by the carbonate/bicarbonate system in freshwaters and was the common measure prior to NAPAP. ANC is the more appropriate measure because it includes other proton acceptors besides carbonate/bicarbonate. In general, alkalinity was measured before NAPAP, and ANC was measured during NAPAP. Acidic lakes or streams, by definition, have ANC chronically less than or equal to zero. Chronically acidic means that the annual average ANC concentration is less than zero; the term does not include aquatic systems that have acidic episodes during storm or snow-melt events. Acidification refers to the loss of ANC or the ability to decrease pH, and acidified refers to surface waters with previously higher ANC or pH values that declined because of acidic inputs. A lake or stream might be acidified but not be acidic. The effects of acidic deposition occur over decades rather than years and over large geographic areas such as New England.

General attributes of sensitive aquatic ecosystems include granitic or noncalcareous bedrock, shallow acidic soils with low base saturation, seepage lakes and small lakes or streams located in the upper portions of the watershed, and systems with low ANC. These characteristics are common to the European, Scandinavian, and North American lakes and streams cited earlier in the literature. Various geologic and soil indices used these characteristics in describing regions potentially sensitive to acidic deposition in the United States (Norton, 1980; McFee, 1980).

Regions with low ANC/alkalinity lakes and streams were mapped based on existing surface water alkalinity data (Omernik and Powers, 1983). The boundaries between regions were drawn using measured alkalinity and receptor characteristics such as geologic and soil sensitivity indices. These alkalinity maps identified regions with low alkalinity lakes and streams that might be susceptible to acidic deposition. Alkalinity was selected to characterize aquatic systems because (1) it is, by definition, a measure of the capacity of the system to neutralize acids; (2) it is an integrated measure of many processes occurring in the watershed/aquatic system controlling surface water acidification; and (3) alkalinity measurements are available for many U.S. lakes and streams. The alkalinity maps served as the basic framework for designing the National Surface Water Survey (NSWS).

The first step in characterizing the ecological components was to determine the proportion and extent of aquatic resources that were acidic or potentially susceptible to acidic deposition. In 1984, EPA initiated the NSWS to quantify the extent and distribution of such aquatic systems. The NSWS, conducted between 1984 and 1986, was a statistically designed survey that provided unbiased estimates of the number, length, area, and location of acidic and low-pH lakes and streams in the United States, based on samples from areas of the country known to contain surface waters with little capacity for neutralizing acids. For quantitative estimates, a specific target population of lakes and streams was defined.

Target Population. The NSWS lake target population of interest was composed of lakes with surface areas >4 ha (1 acre = 0.40 ha) in the East or >1 ha and $<2{,}000$ ha in the West. Lakes identified on 1:250,000-scale U.S. Geological Survey (USGS) topographic maps were given a number, the lakes were stratified by alkalinity or ANC class (e.g., <100, 100–200, >200 microequivalents/liter, or μeq/L), and a systematic random sampling process was used to select sample lakes. The NSWS target stream population contained stream reach segments with drainage areas less than 150 km^2 that were large enough to be represented as blue lines on 1:250,000-scale USGS topographic maps. For streams, a regularly spaced dot-grid, with about 13 km between dots, was randomly overlaid on the regions of interest, and an association rule was used to select the appropriate stream associated with each dot for inclusion in the first-stage sample. These sample reaches were characterized by site name, watershed area, and other geographic information. A probability subsample of these stream reaches was selected because there were too many first-stage streams to field sample for water chemistry. The target populations for evaluation were lakes in the Northeast, Upper Midwest, Southeast, Florida, and mountainous West, and streams in the Mid-Atlantic Highlands and Coastal Plain, Southeastern Highlands, and Florida Regions. Complete descriptions of the National Lake Survey and National Stream Survey can be found in Linthurst et al. (1986), Landers et al. (1987), and Kaufmann et al. (1988).

Endpoints. The assessment endpoint was the vulnerability of lakes and streams to acidification at levels that would endanger the survival of fish species (i.e., based on laboratory mortality data or field data of fish presence or absence as a function of water chemistry). The measurement endpoints were chemical measures related to survival of fish, definition of acidic versus nonacidic systems, or the source of acidity. These included ANC, pH, sulfate, dissolved organic carbon (DOC), calcium, and aluminum. Secondary measurements were important in refining the sources of acidity, assessing potential impacts on aquatic organisms, and ensuring the quality of the data.

Comments on Problem Formulation

Strengths of the case study include:

- *The types and extent of aquatic systems at risk received a comprehensive examination in this program. The study established criteria for screening lakes and streams throughout the country to identify those with low resistance to changes in their acid status. These criteria were then applied to information on the distribution of aquatic systems throughout the country to identify the distribution of those at risk.*

- *This program benefited from the measurement of alkalinity as an integrative limnological variable by early limnologists. The results from this program also provided an extensive background of information on which decisions could be based. The availability of this information illustrates the value of long-term data-gathering efforts.*

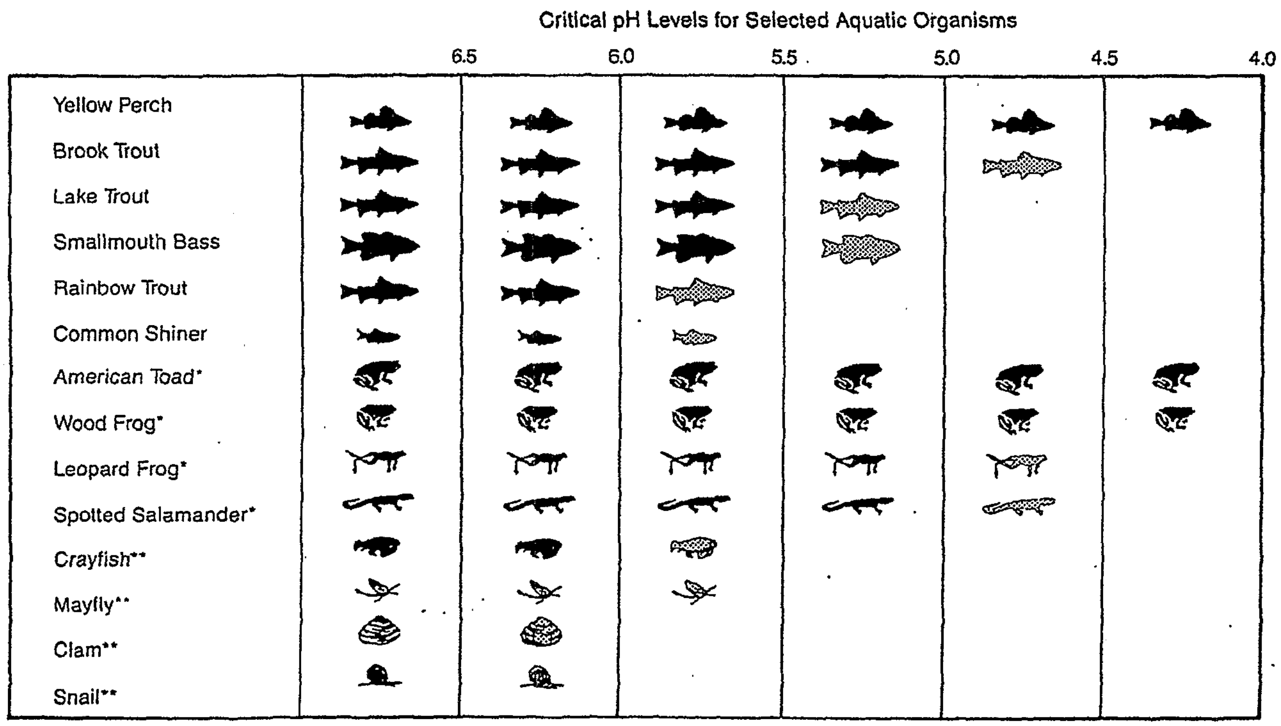
Limitations include:

- *Identification of ecological components at risk should be treated more rigorously. The case study focuses on fish survival as indicated from the acid stress index. Other components of the system may have been at risk but are not evaluated (e.g., zooplankton). As part of the problem formulation stage of a risk assessment, it would be helpful to have a clear rationale for selecting particular components and endpoints.*

1.3.2. Analysis: Characterization of Ecological Effects

Developing the Relationship Between Acidity and Effects on Fish. The relationship between surface-water acidity and acute effects on aquatic life could be demonstrated readily in laboratory and field experiments. (Figure 1-3 presents critical pH levels for selected aquatic organisms.) This information was synthesized and integrated in a series of State-of-Science/Technology (SOS/T) reports that supported the NAPAP 1990 Integrated Assessment (J.P. Baker et al., 1990; Turner et al., 1990).

Changes in surface-water chemistry were related to changes in fish response through a generic acidic stress index (ASI) and through logistic regression models based on statistical analyses of historical fish records in susceptible regions with documented fish loss. The ASI was developed to reflect the combined effects of pH, aluminum, and calcium on selected fish species (J.P. Baker et al., 1990). The model output (i.e., ASI) ranges from 0 to 100, corresponding to between 0 and 100 percent fish mortality in the laboratory experiments used to develop the model. The specific ASI threshold value above which effects might occur, however, varied by species. For example, an ASI > 10 might be appropriate for sensitive species, while an ASI > 30 might be representative for more tolerant species. Similar reference values were established for pH, with reference values of 6.0, 5.5, and 5.0 established for possible effects on biotic populations. The exposure-effects



*Embryonic life stages.
**Selected species.

Figure 1-3. Critical pH levels for selected organisms (J.P. Baker et al., 1990)[a]

[a]Solid symbols reflect favorable pH ranges. Shaded symbols indicate less-favorable pH ranges. pH ranges that generally do not support populations of a particular organism have no symbol.

relationships, therefore, were established between water chemistry variables (i.e., pH, Al, and Ca) and the probability of fish presence/absence for selected species.

By comparing the ASI with other sources of information on fish response, approximate reference levels were defined, above which fish populations might be lost as a result of the high levels of acid stress (J.P. Baker et al., 1990). Surface waters with ASI values exceeding these response thresholds are identified as unsuitable due to acidic stresses. These threshold values differ between lakes and streams because fish responses differ in streams and lakes. Significant uncertainty can be associated with extrapolating laboratory results to field conditions, however, because of uncontrolled, natural variability in field systems, matrix effects on chemical reactions, biotic interactions among species, and other factors.

In the Adirondack subregion, sufficient historical field data were available to formulate logistic regressions for estimating the probability of fish presence/absence based on pH (J.P. Baker et al., 1990). These formulations were used to refine estimates of unsuitable fish habitat in the Adirondack subregion; they could not be developed for other subregions because of insufficient historical data.

Developing the Relationship Between Acid Deposition and Acidification. Multiple analyses were performed to quantify the relationship between acidic deposition and acidification of freshwater lakes and streams. These analyses included statistical associations and correlations between stressor variables and physical/chemical response variables or measurement endpoints (Church et al., 1989); empirical relationships between sulfur deposition and changes in surface-water chemistry (Church et al., 1989; Thornton et al., 1990); process-oriented models describing the quantitative relationships between acidic deposition and watershed processes and interactions with the receiving aquatic system to result in changes in surface-water chemistry (Church et al., 1989; Thornton et al., 1990); paleolimnological analyses to quantify the relationship between historical changes in surface-water chemistry and changes in aquatic life (Sullivan et al., 1990); field manipulation studies to quantify the exposure-effects relation between deposition rate and surface-water acidification (Church et al., 1989); and studies documenting the change in surface-water chemistry following reduction in acidic deposition and quantifying this exposure-effects relationship (Dillon et al., 1987).

The Environmental and Social Systems Analysts/Department of Fisheries and Oceans model is an empirical steady-state model used to assess changes in surface-water chemistry as a function of alternative deposition scenarios. A steady-state model predicts surface-water chemistry values that eventually will be achieved if the system approaches equilibrium with a constant set of inputs (Thornton et al., 1990). Three process-oriented, dynamic models were used to determine the change in water chemistry through time (Model of Acidification of Ground Water in Catchments [MAGIC]), Regional MAGIC, and Integrated Lake/Watershed Acidification Study [ILWAS] [Thornton et al., 1990]). Dynamic model projections are critical for systems that slowly approach steady-state conditions. MAGIC, Regional MAGIC, and ILWAS were used for projections of future changes in surface-water chemistry as a function of alternative deposition scenarios. MAGIC also was used for hindcasts of historical changes in surface-water chemistry as a function of historical changes in acidic deposition rates. These hindcasts were compared with paleolimnological reconstructions using algae fossils. Evidence from paleolimnological studies, which estimate historic ANC based on algae fossils in lake sediments, suggests that lakes with ANC less than 50 μeq/L are most responsive to changes in acidic deposition (Sullivan et al., 1990).

Calibration of Predictive Exposure-Effects Models. The watershed chemistry models were calibrated on three Northeast watersheds and two watersheds in the Southern Blue Ridge Province (SBRP) and then confirmed through blind simulations using only input data, without further calibration, and comparing the model output with additional stream and lake data for the confirmation period. The fish-response models were evaluated using field bioassay data to compare projected with observed responses. Sensitivity analyses also were conducted to identify those input variables and parameters to which the models were sensitive. These variables and parameters received additional attention during the calibration process. These confirmation exercises, however, were conducted on short periods of record (i.e., 3–5 years) while the model simulations made projections for 50 years. Long-term data were not available for model confirmation.

Additional information on the relationship between exposure and effects was obtained by examining paleolimnological data on historical changes in chemical (e.g., pH, aluminum levels) conditions and biotic species assemblages (Sullivan et al., 1990). These were judged to be particularly useful inasmuch as the effects of acidic deposition on aquatic ecosystems occur over long time periods. These data were obtained for a target population of lakes in the Adirondack subregion of New York and provided additional evidence not only of historical exposure levels, but also that lakes in regions receiving acidic deposition had been acidified, and some had become acidic.

1.3.3. Analysis: Characterization of Exposure

Identifying the Target Population of Ecological Components (Lakes and Streams). Of an NSWS target population of about 28,000 lakes and 60,000 streams, 4 percent and 8 percent, respectively, were acidic (table 1-1). Almost no lakes were acidic in the West; virtually all acidic lakes and streams were in the eastern United States. About half of these lakes (~16,000) and streams (~30,000) had ANC <200 μeq/L, indicating they were potentially susceptible to acidic deposition, and 20 percent of the aquatic resources (~5,000 lakes and 12,000 streams) were considered sensitive to acidic deposition (ANC <50 μeq/L, table 1-1).

While there were a significant number of acidic systems in selected regions of the country, not all were acidic because of atmospheric deposition. To identify those lakes and streams that might be affected by acidic deposition, diagnostic procedures using NSWS data and other information were developed to eliminate systems that were acidic because of nonatmospheric sources. The screening determined that the source of acidity in about 75 percent of the acidic lakes (890 lakes) and 50 percent of the acidic streams (2,200 streams) was dominated by acidic deposition. During this screening process, the estimated target population of lakes and streams affected by, or potentially susceptible to, acidic deposition was reduced from 28,300 lakes and 59,600 streams to 15,500 lakes and 27,900 streams, respectively.

Because acidic deposition is a regional problem, the analyses focused on population attributes rather than on individual lakes or streams. Some relationships emerge at the population or regional scale that are not apparent at individual sites. For example, the linear relationship between median

Table 1-1. Percentage Estimates of Number of Lakes and Streams With ANC and pH Below Three Arbitrary Reference Values for Each Measure[a]

			Percentage of Lakes and Streams					
				ANC			pH	
Region	Lake or Stream[b]	Total Number	≤0	≤50	≤200	≤5	≤5.5	≤6
New England	L	4,330	4	20	64	2	6	11
Adirondacks	L	1,290	14	38	73	10	20	27
Mid-Atlantic	L	1,480	6	14	41	1	6	8
Highlands	S	27,700	8	22	48	7	11	17
Mid-Atlantic Coastal Plain	S	11,300	12	30	56	12	24	49
Southeastern	L	258	<1	1	34	<1	<1	<1
Highlands	S	18,900	1	8	52	1	2	9
Florida	L	2,100	23	40	50	12	21	33
	S	1,730	39	70	78	31	50	72
Upper Midwest	L	8,500	3	16	41	2	4	10
West	L	10,400	<1	16	66	<1	<1	1
All NSWS	L	28,300	4	19	56	2	5	9
	S	59,600	8	20	51	7	12	22

[a] The chemical definition of an acidic system is ANC <0. Most scientists agree that acid-sensitive fish are stressed at pH <5.0. However, some scientists believe that acidic episodes can occur if ANC <200 and that fish can be stressed if pH <6. Therefore, three reference values are given for ANC and pH (NAPAP, 1990).

[b] The stream estimates in this table are based on the chemistry measured at the upstream end of each surveyed stream reach. Because ANC and pH almost always increased with distance downstream, the percentage estimates of acidic streams in this table are higher than estimates based on downstream measurements (SOS/T 9:3). In low-ANC streams, the median downstream change was +5 μeq ANC per km of stream length (+0.06 pH units per km). On a length basis, 7,900 km (4%) of the 211,000 km of NSWS streams were acidic, and 26,400 km (13%) had ANC ≤50 μeq/L.

Estimates were also made on the basis of lake surface area (SOS/T 9:8). As acidic lakes tended to be smaller than nonacidic lakes, the percentage of acidic lake area in the NSWS was a factor of 2 smaller than the percentage of acidic lakes based on numbers. Overall, 263 km^2 (2%) of the 12,000 km^2 of lake area in the NSWS were acidic, and 1,310 km^2 (11%) had ANC ≤50 μeq/L.

wet sulfate deposition and median surface-water sulfate concentrations in NSWS subregions is apparent at a regional scale (figure 1-4) even though there was no apparent relationship in individual lakes or streams. Mid-Atlantic and Southeastern Highland streams did not follow this relationship because these regions are still retaining sulfur and are not in sulfate steady-state between atmospheric inputs and watershed outputs. Model predictions of chemical-response variables were also analyzed for the target population as a whole. Sensitivity analyses were performed to evaluate the relative changes in ANC and pH as a function of different changes in deposition (e.g., from 50 percent decrease to 30 percent increase in 10 percent increments). These sensitivity analyses indicate that there is a linear relation between the median change in sulfur deposition and the median change in ANC in 50 years (figure 1-5).

Estimating Current and Future Levels of Acid Deposition. Aquatic systems are exposed to both wet and dry acidic deposition. The wet deposition was monitored in the National Acid Deposition Program/National Trends Network (NADP/NTN), but the dry component was exceedingly difficult to measure even at research monitoring sites. Therefore, the Regional Acid Deposition Model (RADM) was used to estimate the dry deposition and to project the fate, transport, and transformation of sulfur and nitrogen emissions in various regions in the East. These estimates were modified based on watershed physiography and vegetative cover (e.g., coniferous or deciduous forest, open meadows, etc.). The dry deposition estimates were combined with NADP/NTN wet deposition estimates to project the total acidic deposition to which the watershed was exposed. These total acidic deposition estimates were used as inputs to the watershed chemistry models, which projected changes in water chemistry over 50 years and, subsequently, exposure of selected fish populations to hydrogen ion and toxic aluminum concentrations.

Field studies were conducted for RADM to confirm projections. Episodic, atmospheric measurements made over the Ohio River Valley, the Mid-Appalachian area, and the Northeast were compared with RADM projections of dry and wet acidic deposition during these same episodes. Sensitivity analyses also were conducted to identify those input variables and parameters to which the model was most sensitive. These variables and parameters were given additional attention during the calibration process. Different acidic deposition scenarios were projected over the next 50 years. Exposure estimates obtained from RADM and the NADP/NTN were used as inputs to watershed chemistry models, and outputs from these chemical models were used as inputs to the fish-response models described in section 1.3.2. The different acidic deposition scenarios that were selected did not correspond directly to legislative bills but did bound the range of emission control scenarios incorporated in the bills (figure 1-6). One scenario (S1) assumed no controls on sulfur emissions from electrical utilities beyond those already legislated. This no-new-control scenario (S1) estimated reductions in emissions over the 50-year period as new technology/equipment, more efficient and cost-effective in removing sulfur, replaced older equipment. The other scenarios assumed reductions in sulfur emissions (from 1980 levels) of 12 million tons (S3), 10 million tons (S4), and 8 million tons (S5) by the year 2000. For each scenario, estimates of the proportion of currently acidic systems that might recover and the additional proportion of systems that might become acidic were made for systems in the Northeast, Mid-Atlantic, and a portion of the Southeast, the Southern Blue Ridge Province.

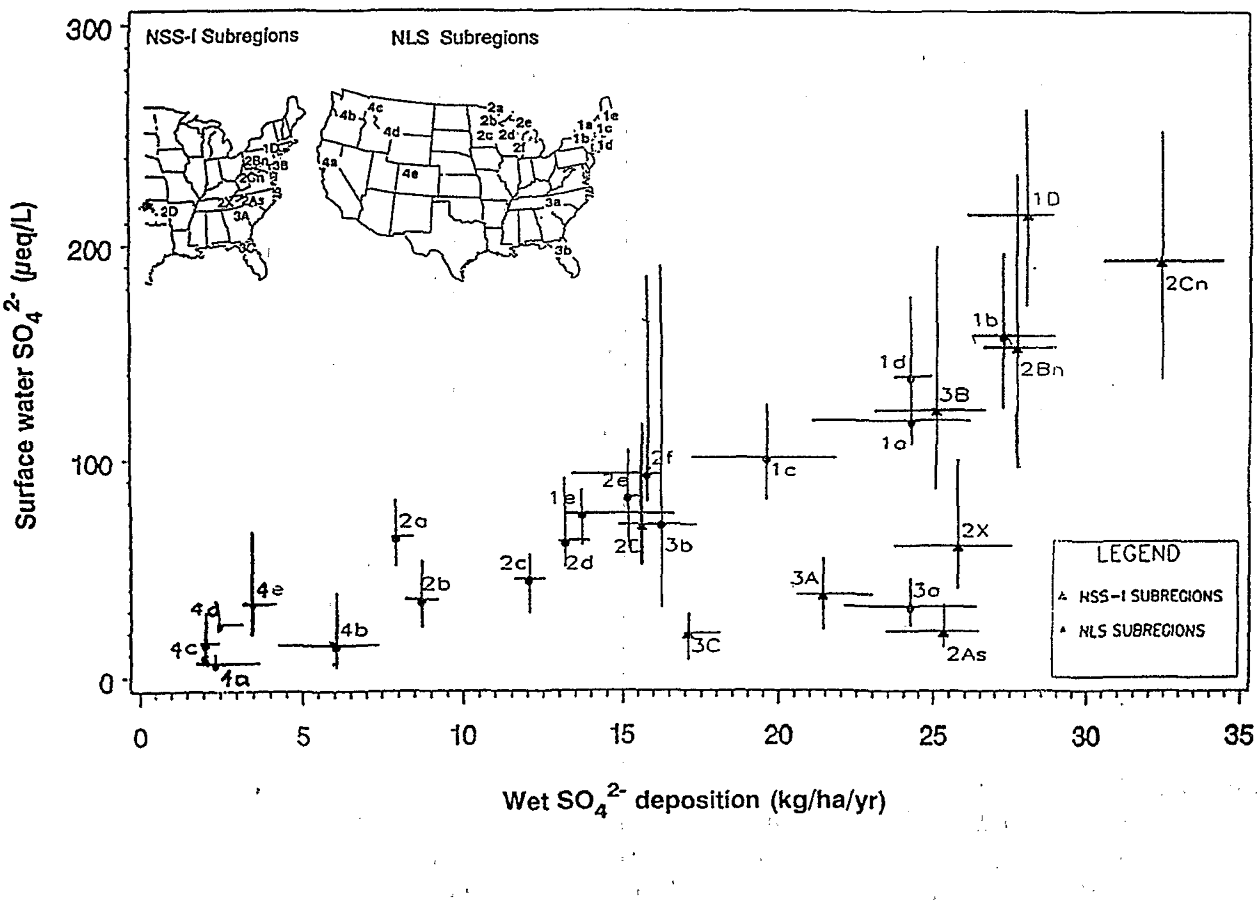


Figure 1-4. Relationship between median wet sulfate deposition and median surface-water sulfate concentrations in NSWS subregions (Sullivan et al., 1988)

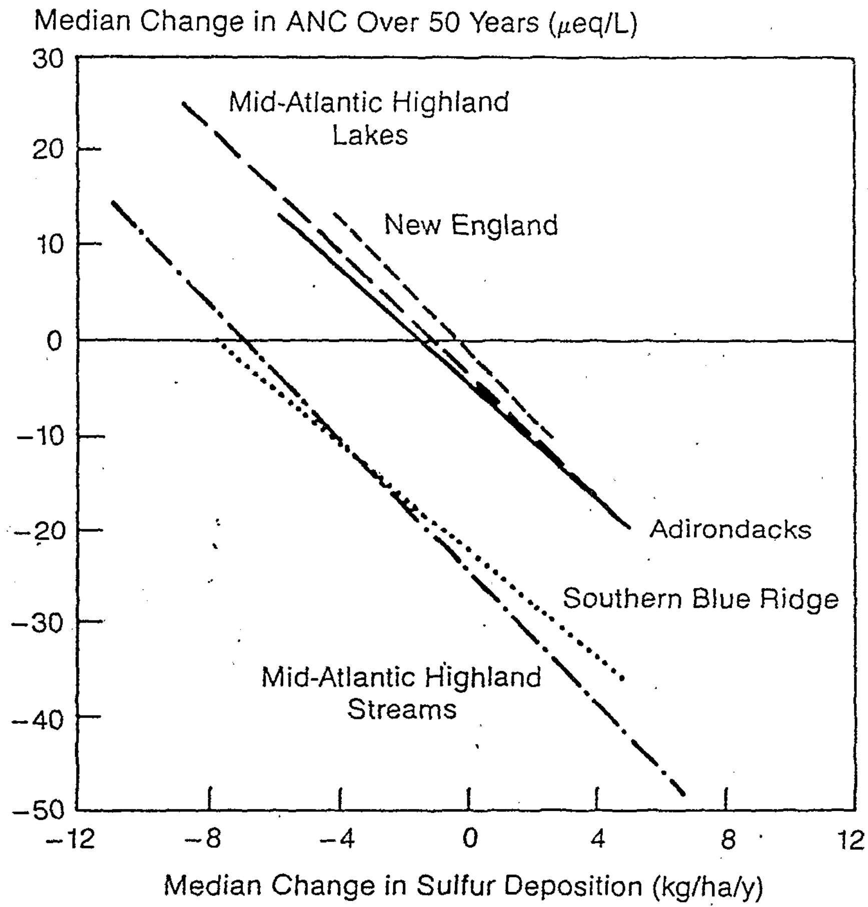
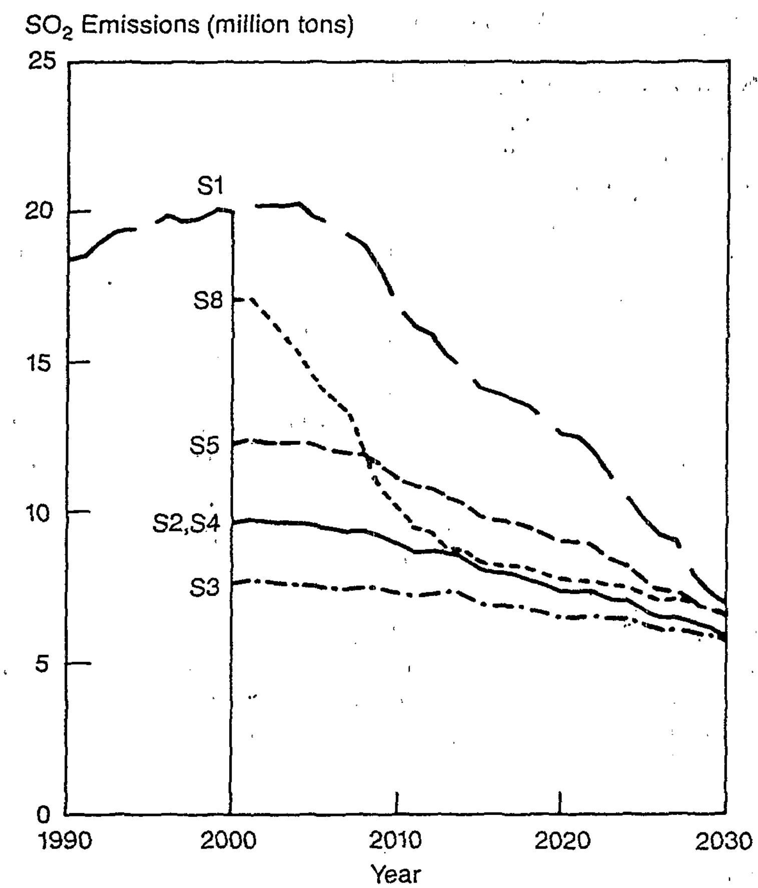


Figure 1-5. Median change in projected ANC (µeq/L) for 50-year MAGIC simulations versus median change in sulfur deposition (kg/ha/yr) for each deposition scenario and subregion. (Points on each line correspond to -50%, -30%, -20%, 0%, +20%, +30% change from current deposition.) The range of absolute changes in sulfur deposition varies from region to region due to differences in current deposition (NAPAP, 1990).



Note: These emission patterns result from the use of uniform input values for both the reference case baseline, S1, and the control scenarios. There are many other plausible baselines which show future emissions without new controls which are much higher or much lower than those in this baseline. Use of other baselines would yield different changes between the control and baseline cases but would not significantly affect the emissions of the control scenarios.

Figure 1-6. Emissions of sulfur dioxide from electric utilities assuming no additional sulfur controls beyond those already legislated (S1) and alternative sulfur reductions (from 1980 levels) of 12 million tons (S3), 10 million tons (S4), and 8 million tons (S5) by the year 2000 (NAPAP, 1990)

1.3.4. Risk Characterization

The final step in an ecological risk assessment is to integrate information on exposure, exposure-effects relationships, and the refined target population of lakes and streams to characterize the risk from acidic deposition to that population. Coupling emission, atmospheric, watershed, and fish-response models provided estimates of the proportion of lakes and streams that might be exposed to acidic deposition at rates sufficient to affect the region's fish. These estimates were based on multiple independent criteria for assessing the effects of acidic deposition on aquatic systems. These independent criteria included the proportion of lakes and streams with pH less than 6.0, 5.5, and 5.0; ANC less than 50 and 0 μeq/L; and ASI values greater than 30 and 10. The values included in the following tables and figures are for pH <6.0, ANC <0 μeq/L, and ASI >10 for sensitive fish species to illustrate the approach. (Note: These three criteria are independent, i.e., an ANC <0 does not correspond to an ASI <10 or a pH <6. See note a, table 1-1.) Multiple criteria were used because of the uncertainty associated with the effects of acidity on different biotic species, including fish. The regions and aquatic systems projected to show the greatest change in unsuitable fish habitat were Adirondack lakes and Mid-Atlantic Highland streams (table 1-2). The Adirondack region has a large proportion of lakes with low ANC. In NSWS, 38 percent of Adirondack lakes had ANC less than 50 μeq/L and 14 percent had ANC less than 0 μeq/L.

With no-emission controls, this estimate of acidic lakes was projected to increase to 22 percent and then decrease to 8 percent by the year 2030 (assuming emission reductions occur as new technology replaces older equipment). The percentage of lakes unsuitable for sensitive fish species is projected to follow the same pattern as lake acidity, increasing from 12 percent to 15 percent and then decreasing to about 6 percent by 2050.

There was little difference in projected effects with or without controls at the end of 50 years, but the control-reduction scenarios did prevent additional acidification of lakes and loss of habitat for sensitive fish species that was projected to occur under the no-control scenario between 1990 and 2030. From the year 2000 to 2030, the average percentage of lakes with chemistry unsuitable for sensitive fish species was projected to be 12 percent without controls and 4 to 6 percent with controls.

Under the control scenarios, 6 percent of the lakes were projected to be acidic in 2030. This figure, however, was twice the number of Adirondack lakes that were likely to have been acidic in preindustrial times based on paleolimnological evidence (Sullivan et al., 1990). Delays in emission reductions under the no-control scenario relative to the reduction scenarios resulted in delays in biological recovery. The longer the period of adverse chemical conditions, the greater the probability of fish loss from lakes or streams. For this reason, extended periods of unsuitable chemistry or delays in deposition reductions might have a greater detrimental effect on fish communities than was apparent from the ASI.

Of the modeled subregions, the Mid-Atlantic Highlands received the highest levels of sulfur deposition. Median total sulfur depositions were 17.1 and 21.8 kg/ha/yr for modeled lakes and streams, respectively, in this region. Surface water would be expected to respond to changes in deposition based on the high levels of sulfur deposition and large proportion of low-ANC systems.

Table 1-2. Average Percentage (Calculated for Years 2000-2030) of Lakes and Streams With Chemistry Unsuitable for Sensitive Fish Species Over 50-Year Projections Under Illustrative Deposition Scenarios for Each Region (NAPAP, 1990)

	Scenario			
Regions	S1	S5	S4	S3
Adirondacks	12	6	4	4
New England	0.4	0	0	0
Mid-Atlantic Highland Streams				
Sensitive Species	25	22	22	20
Brook Trout	16	6	5	5
Mid-Atlantic Highland Lakes	2	0	0	0

(Twenty-two percent of the streams and 14 percent of the lakes have ANC <50 $\mu eq/L$ based on the NSWS; 8 percent of the streams and 6 percent of the lakes have ANC <0 $\mu eq/L$.)

The percentage of acidic streams was projected to increase from the current 5 percent to 10 percent by the year 2000 with no new controls (S1) (figure 1-7). The proportion of acidic streams is then projected to decline to 3 percent by 2030. The number of streams with pH less than 6 was projected to increase from the current 5 percent to 12 percent in the year 2000 and then to decline to 9 percent by 2030 (figure 1-7). The proportion of streams with projected chemistry unsuitable for sensitive fish species would increase from the current 22 percent to 25 percent in 1990 through 2020, then decline to 22 percent by 2030.

Under the deposition-reduction or control scenarios, the proportion of acidic streams (i.e., ANC <0) was projected to decline and range from 0 to 3 percent by the year 2000 and hold nearly constant until 2030 (figure 1-7). The number of systems with pH less than 6 would first increase from 5 percent to 11 percent in 1990, then decrease from 4 percent to 6 percent in 2010 through 2020, and increase again from 7 percent to 9 percent by 2030. This projected increase reflects continuing, long-term acidification in some Mid-Atlantic Highland streams. The percentage of systems with chemistry unsuitable for sensitive fish species would first increase from 22 percent to 25 percent in 1990, then decrease to fluctuate around 17 to 22 percent from 2000 to 2030. The emission-control scenarios (S3-S5) prevented the doubling of the number of acidic streams that had been projected for the no-new-control scenario in the Mid-Atlantic Highlands.

The proportion of streams unsuitable for sensitive fish species would be held roughly constant by the S3-S5 scenarios at about 3 to 5 percent less than the proportion under S1. The average percentage of streams with chemistry unsuitable for sensitive fish species would be 25 percent under S1 and 20 to 22 percent under the S3-S5 scenarios.

Uncertainty was quantitatively evaluated for sampling variability, model input, and calibration error. Sample variability is common to all data-collection efforts and depends on the proportion of watersheds sampled. For example, the NSWS determined that 14 percent of the lakes in the Adirondacks were acidic, based on the results of sampling 155 of the estimated 1,290 Adirondack lakes in the population of interest. In this case, one can calculate that if the NSWS were repeated many times, randomly choosing a set of lakes each time, the estimated percentage of acidic Adirondack lakes would fall between 9 percent and 19 percent in 95 of 100 surveys. Only about one-fifth as many Adirondack lakes (35) were surveyed in the Direct/ Delayed Response Project, a soil survey and modeling study; this smaller set of lakes was both simulated by MAGIC, a process-oriented model, and used for paleolimnological studies. An estimate of the percentage of acidic lakes in the Adirondacks made from the smaller sample is slightly different (16 percent instead of 14 percent) and has somewhat greater uncertainty (e.g., range from 3 to 29 percent). Although sampling uncertainty affects the starting point of simulations (i.e., what proportion of lakes in the region are currently acidic), it does not affect the projected changes in the chemical composition of these lakes.

Another view of sample uncertainty is illustrated in figure 1-8, which shows the variability of projected ANC change from watershed to watershed within the Adirondacks. Individual watersheds may behave differently, and there is often large variance about the median. The median ANC

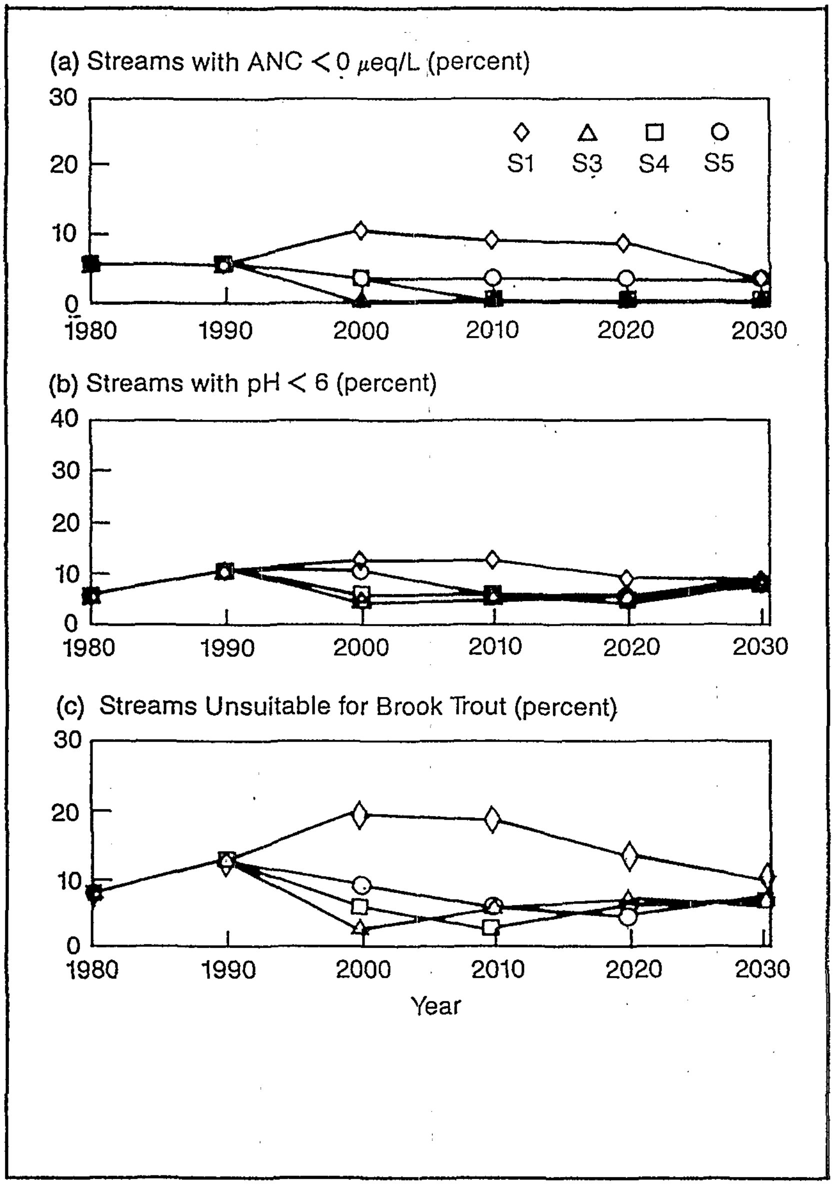


Figure 1-7. Percentage of Mid-Atlantic Highland streams with (a) ANC <0 μeq/L, (b) pH <6, and (c) chemistry unsuitable for brook trout based on MAGIC projections for 50 years under illustrative deposition scenarios (NAPAP, 1990)

Because the true uncertainty of these assessment techniques is unknown, qualitative comparisons must suffice. The numerical model estimates remain our only means of comparing effects of different deposition scenarios. Model results are presented in this assessment as median changes and percentage of systems falling within given criteria. These presentations are useful ways to summarize the data so that differences among scenarios and regions can be seen. These estimates have a great deal of uncertainty about them and are intended for relative comparisons only. Based on all lines of evidence, we have confidence in the direction and relative amounts of projected change. Projections of absolute future conditions and timing of changes are highly uncertain. The numerical model results may represent an upper range for median change in systems with ANC less than 50 μeq/L.

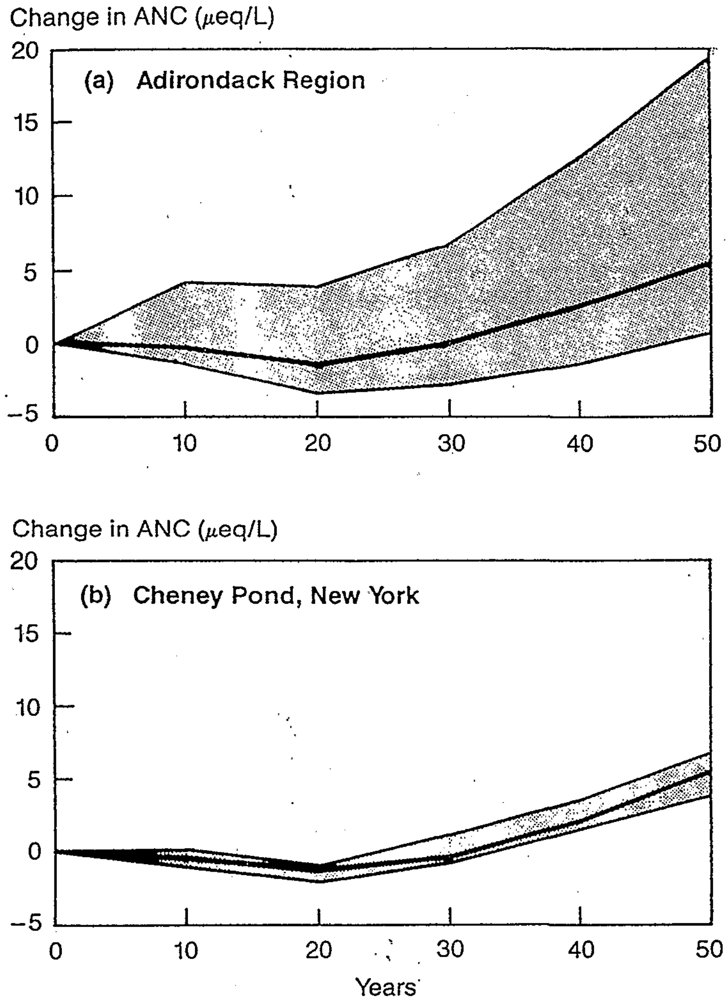


Figure 1-8. Median and range of change in ANC over 50 years for MAGIC projections for (a) 35 lakes in the Adirondacks illustrating variability among watersheds and (b) 10 MAGIC projections for Cheney Pond, NY, illustrating uncertainty in model inputs and calibration (NAPAP, 1990)

change for the Adirondack lakes under the 30 percent reduction scenario is about 5 μeq/L, but the ANC change over 50 years ranges from 1 to 19 μeq/L (figure 1-8).

The uncertainty generated by possible error in model inputs is generally smaller than the watershed-to-watershed sampling variability. The effects of this second source of uncertainty can be quantitatively estimated by various methods, such as using Monte Carlo simulations or running a model many times with different input values that reflect the range of uncertainty in each input. The Monte Carlo approach was used in the MAGIC projections. Each watershed modeled with MAGIC was calibrated up to 10 times with slightly different sets of input parameters. Each input parameter was randomly chosen from within the uncertainty range for that parameter. The range of model projections resulting from the set of calibrations for each watershed represents the parameter and calibration uncertainty in the model. The model input uncertainty for watersheds in the Adirondacks ranges from 2 to 14 μeq/L with a median of 7 μeq/L.

This regional ecological risk assessment provides a quantitative estimate of the target population and quantitative estimates of the proportion of the population that might change at different levels of acidic deposition. The state of science, however, does not currently permit quantitative estimates of the probability that these changes will occur.

Comments on Risk Characterization

Strengths of the case study include:

- *This case study illustrates how models may be used as part of an integrated approach for linking source, fate, and effects of a stressor. The models employed include (a) deposition models that estimated rates of acid deposition within a region or at a site; (b) acidification models that estimated levels of acidification associated with deposition, taking into account hydrologic regimes, in-watershed, and in-lake processes; and (c) population-response models employing stressor-response relationships.*

- *Overall, this case study provides a good example of a risk assessment. The assessment achieved its main goals of (1) quantifying the extent to which lakes and streams have suffered deleterious effects in response to acid deposition and (2) projecting how the numbers of affected lakes and streams will change under different regulatory scenarios.*

Limitations include:

- *The extensive geographic scope of this case study can be viewed as both a strength and a limitation. It illustrates how a problem can be approached from a broad regional perspective. However, it is also possible that too much emphasis is placed*

Comments on Risk Characterization (continued)

on the detail of this regional analysis. The percentage of systems at risk nationwide provides one perspective from which to view an environmental problem, but it is not the only perspective that will influence management decisions. Too much emphasis can be placed on such regionalization. For example, contrast the need to determine how many lakes and streams are at risk in terms of percentages with the identification of a single Superfund site. Some general guidelines need to be developed.

- *The format for presenting and comparing risk can introduce subjective biases. For example, 38 percent of the number of Adirondack lakes have an ANC <50 μeq/L; 12 percent of the lake area in the Adirondacks has an ANC <50 μeq/L. Most of the sensitive Adirondack lakes are small, but there are many of them. The underlying comparative base can influence the perception of risk. Risk communication considerations must be initiated at the onset of the study, not at its completion. The indicators selected, for example, contribute significantly to the comparison and communication of risk. Most decision-makers focus on fish even though zooplankton might be a more sensitive indicator. Indicators should be interpretable in terms of assessment endpoints and/or societal values. Both acidic systems (e.g., ANC <0 μeq/L) and the probability of sport fish loss are two indicators that are understood by decision-makers and the public.*

- *This study focuses on the current number and future estimates of acidic systems (i.e., ANC <0) rather than acidified systems (i.e., decreased ANC because of acidic input). This approach probably underestimates the number of systems affected by acidic deposition. However, long-term monitoring records are required to estimate the number of acidified systems, and these records do not exist (National Research Council, 1986).*

General comments:

Several key lessons learned in the NAPAP experience include:

- *Policy-relevant questions should be formulated early in the process and used to guide the research, analyses, and assessment.*

- *Long-term monitoring records are essential in assessing regional or large-scale problems because of the long time scales associated with ecological effects.*

- *Selection of interpretable indicators is essential when assessing endpoints.*

- *Regional ecological risk assessments require a weight-of-evidence approach and the integration of multiple levels of analyses, including:*

Comments on Risk Characterization (continued)

- *Site-specific process research;*
- *Regional surveys to evaluate extent, magnitude, and distribution of stressor and effects;*
- *Quality assurance programs that are an integral part of the analyses to ensure that the quality of the data is known;*
- *Integrated modeling approaches to project future conditions that incorporate regional differences; and*
- *Qualitative and quantitative estimates of error or uncertainty and clear presentations of these estimates.*

●*Multiagency, multidisciplinary efforts are crucial for regional assessments, both in reducing bias and in providing a broad-scale, policy-relevant perspective.*

●*SOS/T documents are useful for providing the synthesis of technical material and serving as a reference for the assessment document. The 1990 NAPAP Integrated Assessment is generally aimed at a different audience and should integrate and present the technical results without extensive discussion of the details that are included in the SOS/T documents.*

Some of the remaining problems that need to be addressed include:

●*Quantitative estimates of all components of uncertainty or error must be presented clearly to decision-makers.*

●*Better procedures are required for presenting uncertainties to decision-makers and policy analysts and for communicating environmental risk.*

●*Techniques for determining the likelihood or probability of impacts need to be developed. Currently, it is possible to estimate the effects of acidic deposition on aquatic systems qualitatively, but it is not possible to quantify these probabilities.*

●*Ecological monitoring networks, rather than chemical or specific resources, must be implemented nationally and maintained for long periods of time.*

●*A set of decision models or tools is needed in addition to the scientific models or tools for ecological risk assessment.*

●*Technical information from the regional perspective should be transferred to regional and program offices and states for their use.*

The aquatic effects portion of the NAPAP involved substantial resources (probably more than $100M over 10 years). Few assessments are likely to have such resources

Comments on Risk Characterization (continued)

at their disposal. Thus, the extent to which this case study can serve as a model for other assessments must be scaled to differences in resources available for those assessments. At the same time, it should also be possible to evaluate where cuts in the resources available for the NAPAP could have been made with a minimal effect on the final product.

A major criticism of NAPAP is that it did not contribute to legislation on acidic deposition because its results were (1) presented in scientific reports and (2) published after the Clean Air Act Amendments (CAAA) were passed. The NAPAP results were used during the CAAA debates in Congress and within the federal agencies but were presented as briefing packages, congressional testimony, and fact sheets. Better procedures, however, need to be developed to peer review results quickly during congressional debates and to present information so it can contribute to the decision-making process.

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APPENDIX A

SCIENTIFIC CONTRIBUTORS TO THE 1990 NAPAP AQUATIC ASSESSMENT

CONTRIBUTORS

Dr. Joan Baker, Western Aquatics, Inc., Raleigh, NC

Dr. Lawrence Baker, U.S. Environmental Protection Agency, Corvallis, OR

Dr. David Brown, NAPAP, Washington, DC

Dr. Sigurd Christensen, Oak Ridge National Laboratories, Oak Ridge, TN

Dr. Alan Herlihy, U.S. Environmental Protection Agency, Corvallis, OR

Dr. Philip Kaufmann, Utah State University, Logan, UT

Dr. Dixon Landers, U.S. Environmental Protection Agency, Corvallis, OR

Mr. David Mamorek, Environmental and Social System Analysts, Vancouver, British Columbia, Canada

Dr. Mike Sale, Oak Ridge National Laboratories, Oak Ridge, TN

Dr. Timothy Sullivan, E&S Environmental Chemistry, Inc., Corvallis, OR

Dr. Kent Thornton, FTN Associates, Ltd., Little Rock, AR

Dr. Robert Turner, Oak Ridge National Laboratories, Oak Ridge, TN

Dr. Parker Wigington, U.S. Environmental Protection Agency, Corvallis, OR

SECTION TWO

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

THE BAY DRUMS, PEAK OIL, AND REEVES SOUTHEASTERN AREAWIDE WETLAND IMPACT STUDY

AUTHORS AND REVIEWERS

AUTHORS

Quentin J. Stober
Region 4
U.S. Environmental Protection Agency
Athens, GA

Alan G. Auwarter
Region 4
U.S. Environmental Protection Agency
Athens, GA

REVIEWERS

Judy L. Meyer (Lead Reviewer)
Institute of Ecology
University of Georgia
Athens, GA

Nancy A. Bryant
ENSR Consulting & Engineering
Acton, MA

Kenneth L. Dickson
Institute of Applied Sciences
University of North Texas
Denton, TX

Thomas M. Frost
Center for Limnology
University of Wisconsin
Madison, WI

Douglas P. Ormrod
Office of Graduate Studies
University of Guelph
Guelph, Ontario, Canada

Randall S. Wentsel
Toxicology Division
U.S. Army Chemical Research,
Development, and Engineering
Center
Aberdeen Proving Grounds, MD

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LIST OF ACRONYMS

AET	apparent effects threshold
AWIS	areawide wetland impact study
AWQC	ambient water quality criteria
AWRI/FS	areawide remedial investigation/feasibility study
BDRI	Bay Drums remedial investigation
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
CLW	Central Wetland
CPW	Cypress Pond Wetland
EPA	U.S. Environmental Protection Agency
ER-L	Effects Range-Low
ER-M	Effects Range-Median
MSL	mean sea level
NOAA	National Oceanic and Atmospheric Administration
NOW	North Wetland
NPL	National Priority List
OCP	organochlorine pesticides
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PRP	potentially responsible party
SEG	Southeastern Galvanizing
SEW	Southeastern Wire
SFW	Spray Field Wetland

LIST OF ACRONYMS (continued)

SOW	South Wetland
SVOC	semivolatile organic compound
TCL	target compound list
UNC	Unnamed Creek
VOC	volatile organic compound
WET	Wetland Evaluation Technique

ABSTRACT

The areawide wetland impact study that is the basis of this case study had two objectives. The first was to evaluate the ecological status of wetlands associated with the Bay Drums, Peak Oil, and Reeves Southeastern hazardous waste sites. Field surveys of the wetlands and laboratory investigations of sampled material evaluated the existing flora and fauna, ecological functions, water quality, bioaccumulation, and toxicity of the surface waters and sediments. The second objective was to test the possible sources of toxicity; the soil, surface water, and sediments from each of four industrial sites; and contaminant pathways to the wetlands. This provided input to feasibility study design options and baseline information to evaluate the effectiveness of specific remedial actions.

Three wetlands—identified as the North, Central, and South Wetlands—adjacent to the hazardous waste sites were the subject of an impact investigation. Each wetland was potentially subject to receiving ground- or surface-water flow contaminated with materials originating from the hazardous waste sites. The South and Central Wetlands provided hydrologic connections between surface and ground water. The same was assumed for the North Wetland.

In ecological functions, all wetlands received a moderate to high rating. With a diversity of aquatic habitat, all wetlands supported a balanced community of aquatic life: wading birds, fish, aquatic invertebrates, and reptiles.

The sediments and surface water of the three wetlands were found to be relatively contaminated with an array of inorganic and organic chemicals. Florida water quality standards for Class III waters and U.S. Environmental Protection Agency ambient water quality criteria for aquatic life were exceeded by orders of magnitude for aluminum, arsenic, barium, cadmium, copper, iron, and zinc. The excesses occurred primarily at stations on the Unnamed Creek and at a single station in the South Wetland.

Surface waters of the three wetlands were free of acute or chronic toxicity with the exception of a single station. Toxicity, however, was common to the sediments collected from the wetlands, including the reference wetlands. Toxicity was most pronounced in the sediments from the Central Wetland and one of the two reference wetlands, where virtually all test species were affected. The source of the sediment toxicity remains to be further investigated.

The apparent toxicity of the sediment does not appear to have impaired wetland functions—balanced communities of plants and animals remain. However, the Unnamed Creek, an outlet from the North Wetland but also associated with direct drainage from the Reeves Galvanizing facility, was severely affected by heavy metal contamination. The diversity of benthic macroinvertebrates associated with the creek was severely reduced. Sediments and surface water were significantly toxic to virtually all species bioassayed. Fish and benthic macroinvertebrates collected from the creek revealed excessive tissue concentrations of zinc and iron. The levels of zinc found in the biological tissues could constitute an environmental threat to predators of these species.

A large proportion of site source materials tested toxic. The toxicity site source evaluation helped localize the areas of each site where environmental hazards are the most severe. Site source data should also guide remedial decisions in order to correct existing or potential hazards most efficiently. Background toxicological data are now in place from which to monitor later remedial success. Some test organisms were found to be more sensitive than others to source materials; this information can be used to select a subset of tests for future monitoring.

2.1. RISK ASSESSMENT APPROACH

This case study was not initially designed as a risk assessment as defined in the framework (figure 2-1). The Bay Drums, Peak Oil, and Reeves Southeastern areawide wetland impact study (AWIS) is an impact assessment rewritten as a risk assessment. The AWIS, as designed, places the wetlands associated with four Superfund sites at the center of concern. The wetlands are fully characterized as habitats; in effect, they and their viability as habitats, rather than individual species, are treated as the ecological components of concern. Sampling of surface water, sediments, and benthic invertebrates in the study wetlands and in two reference sites is described at length, but no adverse effects are named until they are empirically identified through macrobenthic analyses, tissue analyses, and media-based toxicological testing.

This case study offers a number of valuable lessons on the application of risk assessment: (1) the necessity for multiple measurement endpoints at several levels of biological organization, (2) the difficulty in choosing clean reference sites in a heavily industrialized area, (3) the need for better benchmarks for risk assessments in wetlands, and (4) the value of identifying measurement endpoints to use in assessing the efficacy of remediation efforts.

2.2. STATUTORY AND REGULATORY BACKGROUND

The Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA or Superfund) directs that for designated hazardous waste sites, "The President shall select a remedial action that is protective of human health and the environment . . . at a minimum [shall] take into account: . . . (c) the persistence, toxicity, mobility and the propensity to bioaccumulate of such hazardous materials and their constituents . . . and conduct an assessment of permanent solutions . . . that will result in a permanent and significant decrease in the toxicity, mobility or volume of the hazardous substance, pollutant or contaminant" (U.S. Senate Committee on Environment and Public Works, 1987, Sec. 121[b][1]).

This assessment focuses on the environmental component of that directive and attempts to provide the basis for understanding the environmental impact of four National Priority List (NPL) Superfund sites on the wetlands that surround them.

Section 404 of the Clean Water Act requires protection of wetlands. Under the authority of the Clean Water Act, the U.S. Environmental Protection Agency (EPA) has developed federal water quality criteria, which are used by the states in establishing water quality standards for surface water. Florida state water quality standards thus make up an additional regulatory justification for the AWIS.

2.3. CASE STUDY DESCRIPTION

2.3.1. Problem Formulation

<u>Site Description</u>. The wetlands impact study area is located on State Road 574, a quarter mile west of Faulkenburg Road, Hillsborough County, near Tampa, Florida. The area under assessment includes four Superfund sites—Peak Oil, Bay Drums, Reeves Southeastern Galvanizing

Figure 2-1. Structure of Analysis for Bay Drums Areawide Impact Study

PROBLEM FORMULATION

Stressors: chemical contamination of soil and sediments; physical alteration of wetland habitat.

Ecological Components: wetlands, invertebrates, birds, fish, and endangered species.

Endpoints: assessment endpoint was ecological integrity of wetlands. Measurement endpoints include surveys of wetland function, sediment toxicity, and tissue residues.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
This study did not focus on characterizing exposure. Data are available on levels of contaminants but are not used quantitatively.	A suite of measures including toxicity tests, field surveys, and residue levels were used to compare "impacted and reference" areas.

RISK CHARACTERIZATION

Risks were characterized by:

- comparisons of concentrations to benchmarks (e.g., water quality criteria) using the Quotient Method.
- field and laboratory studies of biological conditions. This involved comparisons between "impacted and reference" areas.

The use of a suite of measures provided a basis for identifying "impacts," but the causative agents were not easily discerned.

(Reeves SEG), and Reeves Southeastern Wire (Reeves SEW)—and three freshwater wetlands—designated North, Central, and South Wetlands—that surround them (see figure 2-2). The Spray Field and Cypress Pond Wetlands are offsite and up gradient. These reference wetlands were selected from a wide array of wetlands for comparison based on their similarities in hydrology, vegetation, and sediment to the three site-related wetlands.

Stressors. Existence of multiple contaminants at these sites was determined by numerous physical studies. A wide array of volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs), organochlorine pesticides (OCPs), and polychlorinated biphenyls (PCBs) were found in the soils around Peak Oil, Bay Drums, and Reeves SEW (table 2-1). Metals were found at all sites, but the Reeves sites were primarily contaminated with iron and zinc. Table 2-1 summarizes the number of contaminants in each group by media. The sites and the surrounding media are contaminated with a mixture of inorganic elements and organic compounds. Representative sampling results for zinc and toluene from the surficial aquifer are presented in figures 2-3 and 2-4. The isopleths indicate contamination gradients onsite, and similar data from emergency response and reconnaissance information for remedial work onsite were used in planning the ecological studies.

In addition to the chemicals, physical effects also have stressed the habitat. Alteration of physical wetland habitat is obvious because of intense industrial activities resulting from drum recycling and storage. During the early stages of the investigation, 70,000 cubic yards of shingles deposited on the site were removed before field site investigations were initiated. Construction of drainage control berms, backfilling of wetlands, construction of wastewater holding ponds, and other activities have reduced or eliminated wetland habitat at the site.

These investigations focused on offsite wetlands and the movement of contaminants to those areas. In addition to contamination originating from the study sites, the Central and South Wetlands receive surface-water runoff from the land application of treated domestic wastewater from a nearby wastewater treatment plant. One comparison site, the Spray Field Wetland, also located in the land application spray fields, was available to address the issue of water quality impact from the treated waste. Comparison sites are limited in this area because of other surrounding industrial activity with similar associated contaminants and widespread residential development in this north Tampa area.

Ecological Components. ***Wetlands***. The focal environmental resources in the area are three wetlands associated with Superfund sites. A classification method developed by the U.S. Fish and Wildlife Service (Cowardin et al., 1979) was used to evaluate all wetlands that had the following vegetative classification similarities. All wetlands were classified as having palustrine vegetative systems in the emergent/aquatic bed class. In general, all five wetlands are dominated with emergent vascular vegetation, such as sedges, rushes, cattails, and Peruvian seedbox. Secondary vegetation comprises mainly floating plants including duckweed, water ferns, and water hyacinth (table 2-2). Distinctions among the study wetlands are:

- The North Wetland (1.75 acres) is the only one of the five wetlands having a surface-water inlet and outlet, the former through a ditch north of the Bay Drums and Peak Oil sites and the latter an unnamed creek through which surface water

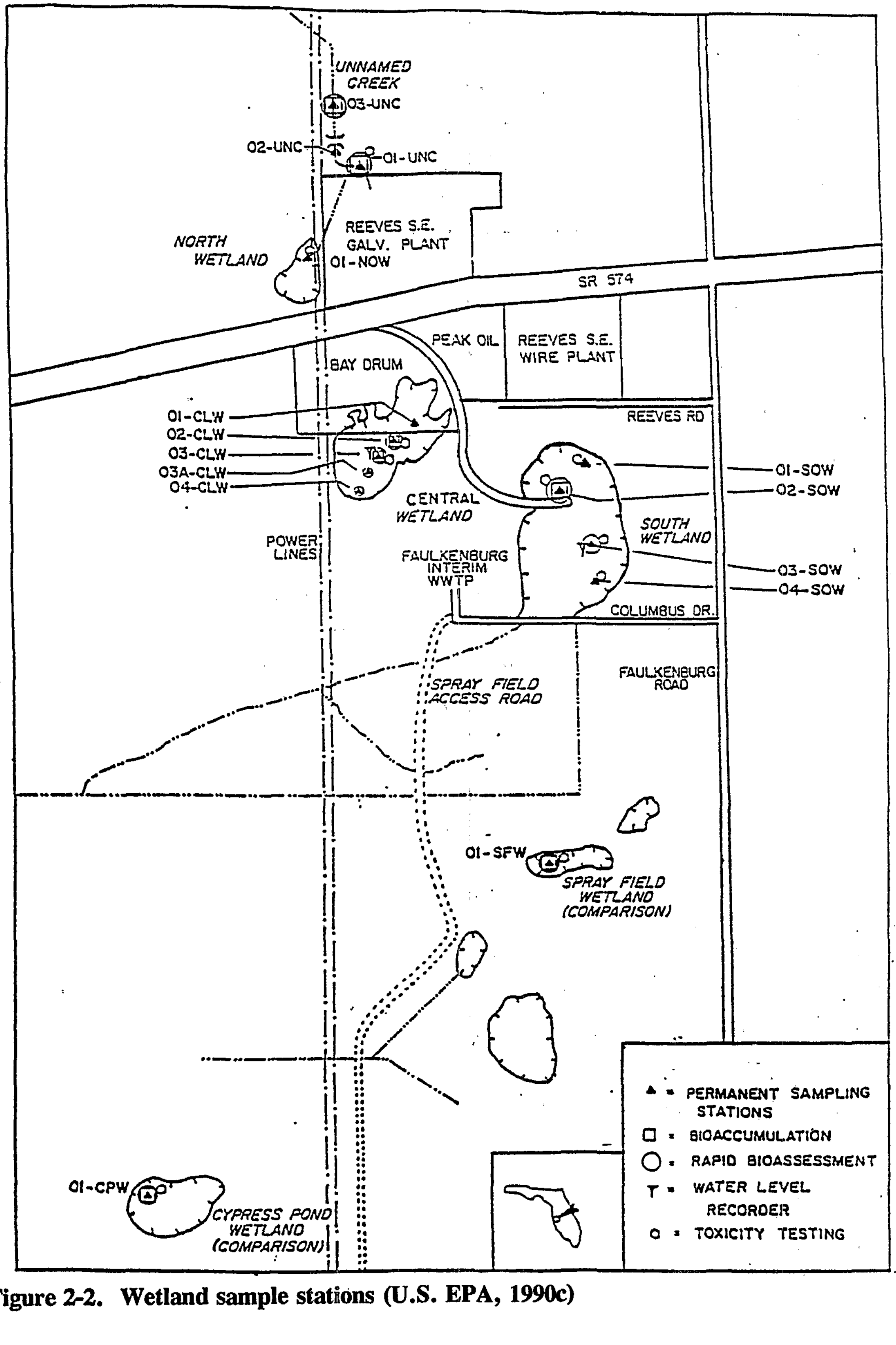

Figure 2-2. Wetland sample stations (U.S. EPA, 1990c)

Table 2-1. Type and Number of Contaminants in Each Group by Site and Media

Contaminants[a]	Peak Oil	Bay Drums	Reeves SEG	Reeves SEW
VOCs	x	x		
SVOCs	x	x		x
OCPs	x	x		x
PCBs	x	x		x
Inorganics and Metals	x	x	x	x

Contaminants	Surficial Aquifer	Upper Floridian Aquifer	Surface Water	Sediments
VOCs	23	14	4	7
SVOCs	17	10	3	17
OCPs	4	1	-	1
PCBs	-	-	1	1
Inorganics and Metals	23	13	20	23

[a]VOC = volatile organic compound
SVOC = semivolatile organic compound
OCP = organochlorine pesticides
PCB = polychlorinated biphenyl

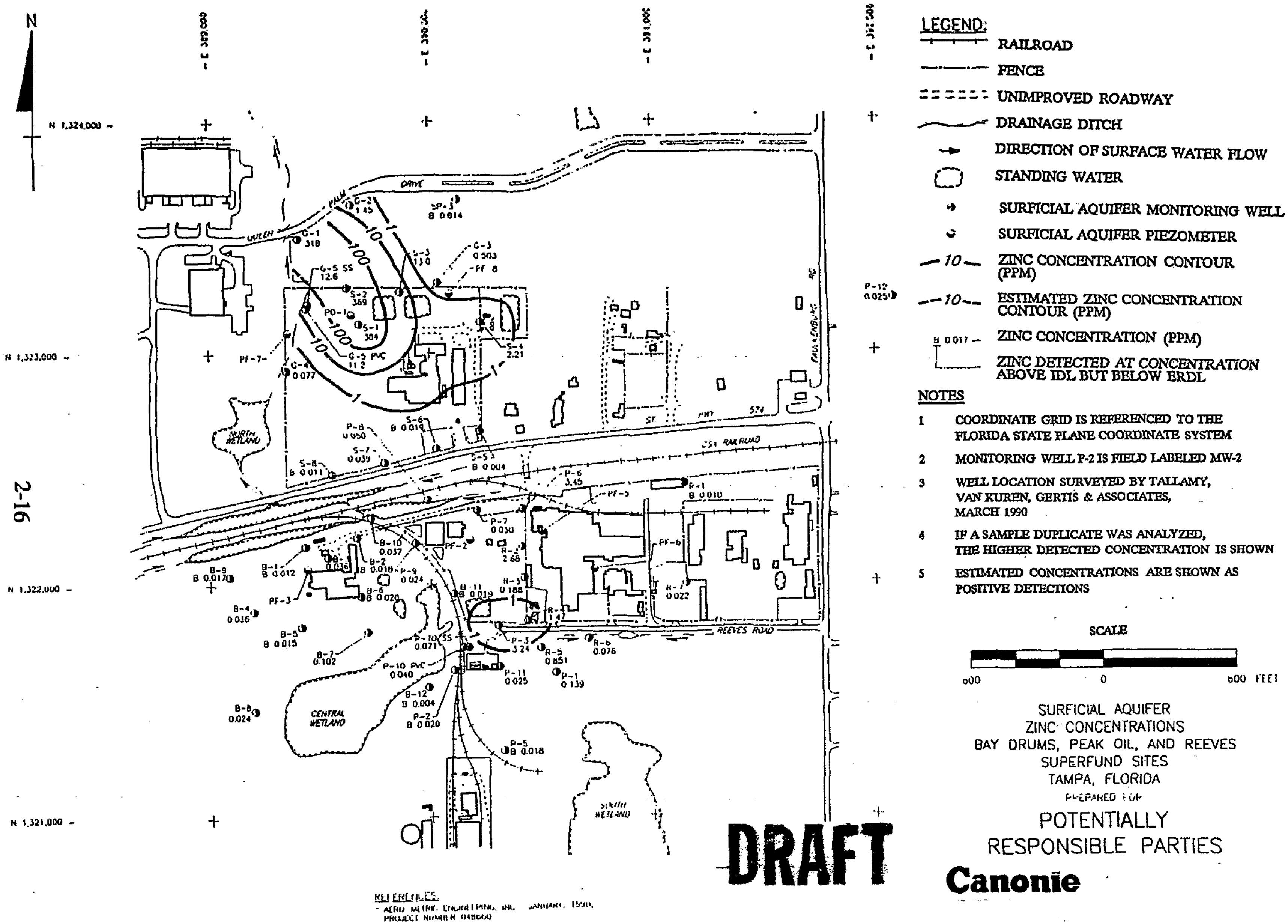

Figure 2-3. Surficial aquifer zinc concentrations, Bay Drums, Peak Oil, and Reeves Superfund sites, Tampa, Florida (Canonie Environmental, 1991)

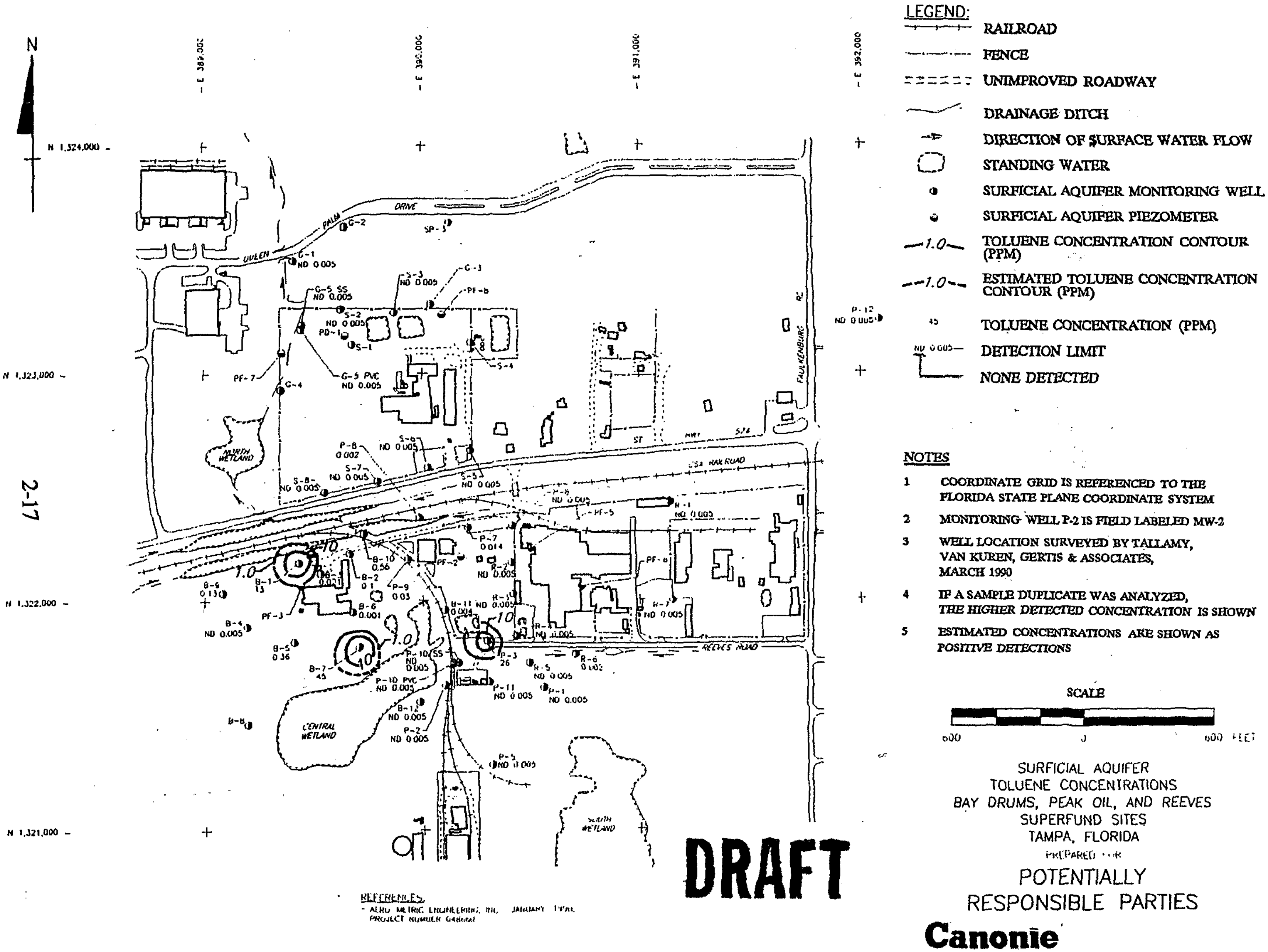

Figure 2-4. Surficial aquifer toluene concentrations, Bay Drums, Peak Oil, and Reeves Superfund sites, Tampa, Florida (Canonie Environmental, 1991)

Table 2-2. Vegetation List: North (NOW), Central (CLW), South (SOW), Spray Field (SFW), and Cypress Pond (CPW) Wetlands (U.S. EPA, 1990c)

Common Name	Species	Relative Abundance[a]	Indicator Status[b]
North Wetland:			
Peruvina seedbox	*Ludwigia peruviana*	Dominant	OBL
Mosquito fern	*Azolla caroliniana*	Frequent	OBL
Mud-midget	*Wolffiella floridana*	Infrequent	OBL
Duckweed	*Lemna* spp.	Frequent	OBL
Water-hyssops	*Bacopa* spp.	Frequent	OBL
Soft rush	*Juncus ffusus*	Frequent	FACW+
Central Wetland:			
Peruvian seedbox	*Ludwigia peruviana*	Dominant	OBL
Pickerelweed	*Pontederia cordata*	Frequent	OBL
Duckweed	*Lemna* spp.	Frequent	OBL
Soft rush	*Juncus effusus*	Frequent	FACW+
South Wetland:			
Cattail	*Typha latifolia*	Dom. emergent	OBL
Water spangles	*Salvinia rotundifolia*	Dom. aqua. bed	OBL
Water hyacinth	*Eichhornia crassipes*	Frequent	OBL
Sedge	*Cyperus haspan*	Frequent	OBL
Water penny	*Hydrocotyle umbellata*	Frequent	OBL
Smartweed	*Polygonum* spp.	Frequent	
Spray Field Wetland:			
Peruvian seedbox	*Ludwigia peruviana*	Dominant	OBL
Soft rush	*Juncus effusus*	Frequent	FACW+
Cattail	*Typha latifolia*	Frequent	OBL
Cypress Pond Wetland:			
Cattail	*Typha latifolia*	Codominant	OBL
Soft rush	*Juncus effusus*	Codominant	FACW+
Arrowroot	*Thalia geniculata*	Infrequent	OBL
Bald cypress	*Taxodium distichum*	Infrequent	OBL
Pickerelweed	*Pontederia cordata*	Frequent	OBL
Peruvian seedbox	*Ludwigia peruviana*	Dominant	OBL
Common to All Study Wetlands:			
Peruvian seedbox	*Ludwigia peruviana*		
Cattail	*Typha latifolia*		
Soft rush	*Juncus effusus*		

[a]Relative abundance by estimated coverage.
[b]OBL = obligate wetland species, FACW+ = facultative wetland species.

leaves the area after crossing a corner of Reeves SEG. The North Wetland receives, or may have received, runoff from any of the four Superfund sites.

- The Central Wetland (6.25 acres) is at present connected above ground with the Bay Drums pond on the Bay Drums property, but it has no defined surface-water inlet or outlet. Surficial water inflows are from direct rainfall or runoff. Its catchment basin includes the Bay Drums and Peak Oil sites and land application (spray) of treated domestic wastewater. The northern boundary of the Central Wetland has been physically altered by activities such as stacking roofing shingles on the Bay Drums site. The remainder is maintained pasture for land application of wastewater, powerline right-of-way, and a rail spur.

- The South Wetland (9.7 acres) has no defined surface-water inlet or outlet. Its catchment basin includes the Reeves SEW and Peak Oil sites and the land application (spray) of treated domestic wastewater.

- The Spray Field Wetland (reference site, 0.8 acres) is located within the land application spray field. It is surrounded by grassed fields maintained by routine mowing. The water quality and hydrology of the wetland are influenced by the discharge of treated domestic wastewater. The wetland has no defined surface-water inlet, but a small outlet could be active during high water after a storm.

- The Cypress Pond Wetland (reference site, 1.5 acres) is located southwest of the Spray Field Wetland. Like the Central and South Wetlands, it has no defined surface-water inlets or outlets. Stands of bald cypress are found along the edges and several trees are located close to the middle of the wetland. Contamination of both reference sites is likely from adjacent industrial and urban activities.

The seasonal quantity and quality of surface water are critical in the formation and maintenance of wetland habitats. Water quantity was monitored via surface-water elevations in all wetlands except the North from April 1988 to November 1989 by simultaneous staff gauge readings. Seasonal variation was approximately 2 ft. Comparison of water levels at the South Wetland with official rainfall records indicates a direct response. Since the site has insignificant surface-water relief, water-level recession rates appear to be attributable to evapotranspiration. Comparison of the four wetland systems shows that the seasonal shift in the water level of the Central Wetland is significantly greater than that of the others.

Surficial flow patterns for the period 1948 to 1985 were provided by a photohistory from EPA's Environmental Monitoring Systems Laboratory (U.S. EPA, 1985). Four of the five wetlands have no permanent surface-water inlets and depend on rainfall and potential ground-water recharge for surface-water inflow. These surficial flow patterns have been physically altered by industrial activities at the site.

Monitoring of classical water quality parameters focused on dissolved oxygen. Low concentrations of dissolved oxygen in the wetlands are illustrated by diel monitoring records. For the Spray Field and Central Wetlands, dissolved oxygen peaked during the day with minimum concentrations of 0 to 2 mg/L at night. Dissolved oxygen levels in the Cypress Pond and South Wetland were consistently near zero.

The potential for interaction of ground and surface waters in these wetlands is pronounced. Due to the shallow soil lithology, no significant confining layer was found that would prevent vertical migration of site contaminants, representing a direct pathway for exchange of contaminants between surface and ground water.

The North, Central, and South Wetlands and their associated ecotones provide food and cover for several species of birds, reptiles, fish, and mammals. The study wetlands provide a short-hydroperiod flooding regime that concentrates fish and macroinvertebrates for easy food gathering by wading birds.

Macroinvertebrate communities. Benthic macroinvertebrate communities were sampled qualitatively with a multihabitat rapid bioassessment protocol. The benthic macroinvertebrate community was identified as an important measurement endpoint to evaluate one aspect of the wetlands. The Cypress Pond Wetland was used as a reference site to compare benthic macroinvertebrate communities in the other wetland sites. All sampling was conducted in January 1989 when 53 taxa were found (table 2-3).

Birds and fish. In April 1988, several wading birds, including white ibis and snowy egret, were observed actively feeding within receding surface waters of the North Wetland. Great blue herons, great egrets, roseate spoonbills, white ibis, and black skimmers have all been sighted in the Central Wetlands. Several year-round residents such as the common marsh hen and red-winged blackbirds have been sighted several times in the study wetlands. Brown thrashers, mockingbirds, and quail are prevalent in the thickets, trees, and open grassed fields within the ecotones adjacent to the study wetlands.

Four species of fish common to Florida wetlands were collected for identification or for tissue analysis: *Gambusia affinis* (mosquito fish), *Mollienisia latipinna* (sailfin molly), *Fundulus* sp. (killifish), and *Jordanella floridae* (flagfish). They can survive low levels of dissolved oxygen and are common forage fish for birds and other predators.

Endangered species. Several species of endangered plants and animals have a range that includes Hillsborough County. The American alligator, listed as threatened (*Federal Register*, 4 June 1987), was sighted on several occasions. The South Wetland, relatively large, secluded, and semipermanently to permanently flooded, probably supports their year-round presence, but given the proximity and seclusion of the study wetlands, alligators are likely to migrate freely among them to feed.

The endangered wood stork (*Federal Register*, 28 February 1984) also may feed in the study wetlands, especially when fish and macroinvertebrates are concentrated in small isolated pools. A wood stork was observed near the site. Suitable nesting habitat is not available, however, except for the tree communities that fringe the South Wetland. No signs of previous or current nesting sites have been observed in the South Wetland.

Table 2-3. Checklist of Benthic Macroinvertebrates (Bay Drums Wetlands Study, January 1989) (U.S. EPA, 1989)

	Sta. 01-CPW	Sta. 01-SFW	Sta. 02-SOW	Sta. 03-SOW	Sta. 02-CLW	Sta. 04-CLW	Sta. 01-UNC	Sta. 03-UNC	Sta. 01-NOW[a]
DIPTERA									
Culicidae									
Anopheles sp.	x	x					x		
Hansonia sp.	x					x			
Culex sp.	x					x			
Uranotaenia sp.	x								
Ceratopogonidae									
und. sp.	x		x			x			
Chaoboridae									
Chaoborus sp.					x				
Tabanidae									
Chrysops sp.									x
Tipulidae									
Limonia sp.									x
Stratiomyidae									
Eulalia sp.					x				
Chironomidae									
Chironomus sp.					x[b]				
C. sp. 1		x							
C. sp. 2		x							
Goeldichironomus sp.	x					x		x	x
Kiefferullus sp.	x				x				
Parachironomus sp.					x				
Polypedilum sp.	x								
P. trigonum		x							

Table 2-3. **Checklist of Benthic Macroinvertebrates (continued)**

	Sta. 01-CPW	Sta. 01-SFW	Sta. 02-SOW	Sta. 03-SOW	Sta. 02-CLW	Sta. 04-CLW	Sta. 01-UNC	Sta. 03–UNC	Sta. 01-NOW[a]
EPHEMEROPTERA									
Callibaetis sp.	x				x[b]	x			
Caenis sp.		x	x						
ODONATA									
Argia sp.			x	x	x	x		x	
Ischnura sp.	x	x	x		x	x		x	x
Enallagma sp.	x			x		x			
Nehalennia sp.		x							
Anax sp.	x		x		x	x			
Coryphaeschna sp.			x						
Pachydiplax longipennis	x		x	x	x	x		x	x
Erythemis sp.		x		x	x				
LEPIDOPTERA									
Nymphula sp.									x
HEMIPTERA									
Nepidae									
Ranatra sp.		x			x				
Naucoridae									
Pelocoris sp.	x	x	x	x	x				x
Corixidae									
Sigara sp.		x			x				
Mesoveliidae									
Mesovelia sp.		x		x					
Belostomatidae									
Belostoma sp.	x	x				x	x		x

Table 2-3. Checklist of Benthic Macroinvertebrates (continued)

	Sta. 01-CPW	Sta. 01-SFW	Sta. 02-SOW	Sta. 03-SOW	Sta. 02-CLW	Sta. 04-CLW	Sta. 01-UNC	Sta. 03–UNC	Sta. 01-NOW[a]
Gerridae									
Gerris sp.			x						
Notonectidae									
Notonecta sp.									x
COLEOPTERA									
Enochrus sp.			x					x	
Hydrocanthus sp.		x	x			x	x		x
Hygrotus sp.		x							
Peitodytes sp.	x				x	x	x		
Berosus sp.					x		x		
Tropisternus sp.	x	x	x	x		x	x	x	x
Coptotomus sp.				x			x	x	x
Hydrophilus sp.			x						
Graphoderus sp.	x					x			x
Apion sp.	x	x							x
Helodidae	x		x						
CRUSTACEA									
Hyalella azteca	x	x	x	x	x	x		x	x
Astacidae	x	x	x	x	x	x		x	x
Ostracoda		x							
OLIGOCHAETA									
Tubificidae			x						
Lumbriculidae		x						x	x
Dero sp.			x						
Peloscolex sp.									

Table 2-3. **Checklist of Benthic Macroinvertebrates (continued)**

	Sta. 01-CPW	Sta. 01-SFW	Sta. 02-SOW	Sta. 03-SOW	Sta. 02-CLW	Sta. 04-CLW	Sta. 01-UNC	Sta. 03–UNC	Sta. 01-NOW[a]
HIRUDINEA									
Helobdella lineata		x							
H. stagnalis								x	
GASTROPODA									
Lymnaea sp.						x			
Pseudosuccinea sp.					x				
Physella sp.		x			x			x	x
Loevapex sp.					x				x
TOTAL TAXA	22	23	19	11	22	18	7	14	21
[a]Quotient of Similarity		0.35	0.44	0.42	0.45	0.65			0.46

[a]Comparison to Sta. 01-CPW.
[b]Aberrant.

The range of the bald eagle, Florida scrub jay, eastern indigo snake, and Florida golden aster includes Hillsborough County, but adequate feeding or breeding opportunities do not exist within the study wetlands and their adjacent ecotones.

Endpoints. The ecological integrity of the wetlands was chosen as the assessment endpoint supported by a suite of measurement endpoints (techniques) used to measure integrity. The interactions of a suite of specific wetland measurement endpoints with environmental contaminants were assessed to determine the impacts to the wetlands. Toxicity testing of contaminant mixtures in several media was used to evaluate the extent to which toxic conditions occurred at and around the sites and to provide insight into the possible migration of toxic-causing materials from the site into adjacent wetlands.

Wetland attributes were analyzed with the Wetland Evaluation Technique (WET), Version 2.0 (Adamus et al., 1987). WET assesses functions and values of a wetland in three categories: *Social significance* assesses the value of a wetland to society in terms of its special designations, potential economic value, and strategic location; *effectiveness* rates the capability of a wetland to perform a function based on its physical, chemical, or biological characteristics; and *opportunity* rates the ability of a wetland to perform a function to its level of capability. The wetland functions and values that are assessed include ground-water recharge, ground-water discharge, floodflow alteration, sediment stabilization, sediment/toxicant retention, nutrient removal/transformation, production export, wildlife diversity/abundance, aquatic diversity/abundance, uniqueness/heritage, and recreation. Only three functions and values are rated for opportunity: (1) floodflow alteration; (2) sediment/toxicant retention; and (3) nutrient removal/transformation. A rating of high, moderate, or low is assigned to each function and value in the three categories (table 2-4). The wetlands were rated as moderate to high in ecological functions and values.

Comments on Problem Formulation

- *Because the AWIS was not designed as an ecological risk assessment, risk hypotheses were not explicitly developed in this case study, although choosing endpoints is implicitly based on hypotheses. Possible hypotheses include: (a) There is an inverse relationship between level of contamination and biological integrity as measured by various endpoints; and (b) by decreasing contaminant levels, there would be improvement in biological integrity. Had these been the risk hypotheses at the outset, the sampling design might have been different. Contaminant levels and known effects were not used to establish a causal relationship between contaminants and their adverse effects.*

- *At this site the stressors include habitat alteration, hydrologic changes, and chemical contaminants. Synergistic effects are likely to be important, although they were not assessed. The problem formulation step seems especially important at Superfund sites, where there is such a wide range of potential stressors.*

Table 2-4. Wetland Evaluation Technique (WET) Probability Rating Comparison: North (NOW), Central (CLW), South (SOW), Spray Field (SFW), and Cypress Pond (CPW) Wetlands (U.S. EPA, 1990c)

Functions/Values[1]	Wetland[2]				
	NOW	CLW	SOW	SFW	CPW
Social Significance:					
Ground Water					
Recharge	M	M	M	M	M
Discharge	H	H	H	H	H
Floodflow Alteration	M	L	M	M	M
Sediment Stabilization	M	M	M	M	M
Sed./Toxicant Retention	H	H	H	H	H
Nut. Removal/Transformation	M	M	M	M	M
Wildlife D/A	M	M	M	M	M
Aquatic D/A	M	M	M	M	M
Uniqueness/Heritage	H	H	H	H	H
Recreation	L	L	L	L	L
Effectiveness:					
Gound Water					
Recharge	L	U	U	U	U
Discharge	M	M	M	M	L
Floodflow Alteration	M	H	H	H	H
Sediment Stabilization	H	M	L	M	L
Sed./Toxicant Retention	H	H	H	H	H
Nut. Removal/Transformation	L	H	H	H	H
Production Export	M	L	L	L	L
Wildlife D/A					
Breeding	L	L	L	L	L
Migration	L	H	H	L	L
Wintering	L	H	H	L	L
Aquatic D/A	L	L	L	L	L
Opportunity:					
Floodflow Alteration	H	H	H	H	H
Sed./Toxicant Retention	H	H	H	H	H
Nut. Removal/Transformation	L	L	L	H	L

[1]D/A = diversity/abundance.
[2]H = high, M = moderate, L = low, U = uncertain.

2.3.2. Analysis: Characterization of Ecological Effects

Ecological effects were characterized as part of the wetland assessment and to provide toxicity data for samples collected at the industrial sites. The toxicity data were obtained: (a) to understand and evaluate effects seen in the wetlands, (b) to help define appropriate remedial scenarios, and (c) to provide benchmarks from which to judge remedial effectiveness.

Environmental Toxicity Testing. The toxicity assessment involved the evaluation of the three wetland areas under study, two reference wetland areas, and four industrial sites. Implementation of the assessment required cooperation and coordination among several sections within EPA Region 4, several contractors, subcontractors, and oversight contractors for both EPA and the potentially responsible parties (PRPs).

More than 400 samples of surface water, sediments, soil, and ground water were taken from the industrial sites and wetlands for chemical analysis. One hundred of these samples (excluding ground-water samples) were selected to be split for toxicity testing, targeting those samples most likely to test toxic (site-collected samples), or to establish gradients of effect (wetland samples). No attempt was made to randomize locations of samples taken for toxicity tests.

Samples were collected, handled, and distributed for analysis in accordance with the EPA and EPA-approved work plans that governed several remedial investigations, feasibility studies, and site-source characterization efforts that were under way simultaneously at the study sites (Canonie Environmental, 1988a, b; Pace Laboratories, Inc., 1988; U.S. EPA, 1989).

Prior to using soils and sediments for most tests, elutions were prepared with laboratory-pure water (80:20, Milli-Q®:Perrier®). For sediments and saturated soils this process diluted each sample with four times its volume of water. For drier soils, the ratio was the same, but its basis was wet weight to dry weight.

Water, sediment, soil samples, and eluates were divided according to the needs of individual toxicity tests (figure 2-5). Water samples were tested with the Microtox bacterial assay (*Photobacterium* nr. *phosphoreum*); a freshwater alga (*Selenastrium capricornutum*); a freshwater cladoceran (*Ceriodaphnia dubia*); a freshwater fish (*Pimephales promelas*); and a terrestrial plant, a species of lettuce, *Lactuca sativa*. Soils and sediments were eluted, and the eluates were tested with the same suite of organisms. Table 2-5 identifies the statistical analyses used in working up test data.

The toxicity tests used were those for which standard protocols existed and for which individual organisms could be cultured. Several species from various phyla were chosen to increase the confidence of correctly diagnosing the toxicity in an array of media. Although laboratory-to-field extrapolation is always a concern, the observed toxic responses correlated with other measurement endpoints at the most heavily affected sites.

Results of the toxicity testing are summarized below for each area tested.

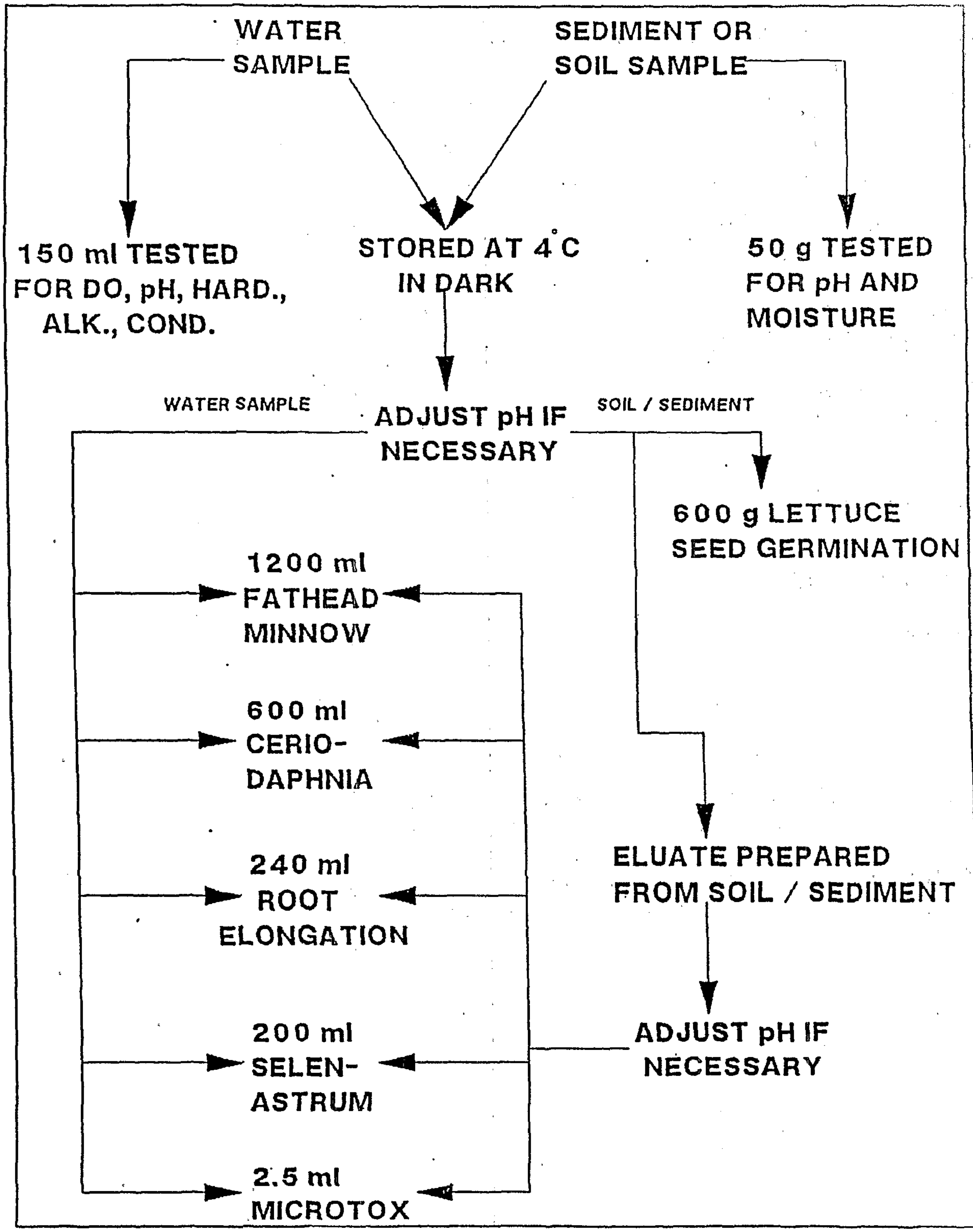

Figure 2-5. Toxicity tests flowchart (U.S. EPA, 1990c)

Table 2-5. Statistical Analysis of Data From Toxicity Tests Performed on Bay Drums Samples

Toxicity Test	Statistical Analysis[a]
Fathead Minnow Chronic Test (survival and growth)	Dunnett's procedure
Ceriodaphnia Chronic Test (survival and reproduction)	Student's t-test
Ceriodaphnia Acute Test (survival)	LC_{50} obtained by Probit analysis or graphical method (line intercept)
Lettuce Seed Germination	Student's t-test
Lettuce Root Elongation (germination and root length)	Dunnett's procedure
Selenastrum	Dunnett's procedure
Microtox	EC_{50} obtained by line intercept method

[a]Data generated by the toxicity test were analyzed by either hypothesis testing (Dunnett's or Student's t-test) or endpoint estimates (Probit analysis or line intercept methods). Detailed procedures for these statistical methods can be found in the EPA manuals EPA 600/4-85/013 and EPA 600/4-89/01, and in a computer software program supplied by the Microbics Corporation (developer of the Microtox test).

Bay Drums. The site was generally contaminated with a spectrum of inorganic elements and organic compounds, many of which were present at concentrations well above those known to cause adverse biological effects. Surface soils were contaminated with a larger number of chemicals in higher concentrations than the majority of samples. Some sample stations on the western and southern property borders appeared to be much less contaminated than the majority of samples. Water from the Bay Drums Pond was essentially nontoxic to any of the test organisms exposed to it. Water from the northern drainage ditch and the backfill pond was highly toxic to daphnids (water fleas) but not to other species (table 2-6). All four water samples stimulated algal growth.

Sediments from the northern drainage ditch and both ponds were highly toxic to daphnids and algae, and inhibited germination of lettuce seeds (table 2-6). The fish (*P. promelas*) test was sensitive to contaminants found on this site, but *Daphnia* tests were more sensitive for almost every sample. The daphnids would therefore be prime candidates for future toxicity testing at this site, both as a monitor and to evaluate the success of remediation. Tests using bacteria (*P. phosphoreum*) and algae (*S. capricornutum*) were also sensitive and may be useful in the future. Lettuce (*L. sativa*) tests were largely insensitive: development of root length was not hampered by any site sample, and seed germination was affected only by sediment samples and two soil samples.

Peak Oil. All surface waters analyzed contained many inorganic elements. Aluminum, iron, and lead exceeded the ambient water quality criteria (AWQC) most frequently. Water from the southernmost of the three onsite ponds was considerably more toxic than water from the other two. Almost all sediments and soils from there were highly toxic to most species. Soil samples split for bioassay were logged as being brown or dark brown in color and oily. The *Daphnia*, minnow, bacterial, and algal tests were all sensitive to most site samples and would be useful as future remedial monitoring tools.

Reeves Southeastern Wire. Pond water and sediments were highly toxic. Pond sediments contained both inorganics and organics, including arsenic at biologically high concentrations. Pond water exceeded AWQC for 10 inorganic chemicals: the highest rates were for copper, lead, and zinc. Site soils were generally contaminated with lead and zinc above toxic concentrations and tested chronically toxic to living organisms. Cyanide was found in soils on the west side of the site. *Daphnia* would be a preferred test species for monitoring remedial progress at this site since they were sensitive to more samples than were other species.

Reeves Southeastern Galvanizing. Surface waters from the inactive ponds were nontoxic. Their sediments were highly toxic, with high concentrations of several metals. Soil samples from the extreme eastern side of the site were essentially nontoxic. Other soils were generally toxic, contaminated with several inorganic elements. This is especially true for the former drum storage area and the drainage south of it, the area northwest of the Reeves Galvanizing property, and the high conductance area. *Daphnia* and algae were most sensitive to samples from this site and would be useful in future remedial monitoring.

Table 2-6. Wetland Bioassay Data (U.S. EPA, 1990c)

BAY DRUMS, PEAK OIL AND REEVES SE AREAWIDE WETLAND IMPACT STUDY, TAMPA, FLORIDA				CERIODAPHNIA DUBIA			PIMEPHALES PROMELAS			LACTUCA SATIVA			PHOTOBACTERIUM nr PHOSPHOREUM		S. CAPRICORNUTUM
				CHRONIC (c)		ACUTE(d)	CHRONIC (c)		ACUTE(d)						
SAMPLE TYPE	SAMPLE LOCATION	STATION ID #	DATE TAKEN	MEAN NO OFFSPRING	% SURVIVAL	48 HOUR LC50 (% CONC.)	MEAN FRY WT. (mg)	% SURVIVAL	48 HOUR LC50 (% CONC.)	MEAN ROOT LENGTH(mm)	% SURVIVAL	% GERMINATION	5 MIN. EC50 % CONC.	15 MIN. EC50 % CONC.	% CHANGE
SURFACE WATER	NORTH WETLAND	01-NOW	11/30/89	30.5	100	b	0.582	97	b	47.4	93	f	a	a	+ 259
SURFACE WATER	UNNAMED CREEK	01-UNC	11/30/89	0 *	0 *	0.22	0 *	0 *	2.4	10.7 *	100	f	21.56	4.94	+ 28
SURFACE WATER	CENTRAL WETLAND	01-CLW	11/30/89	26.9	100	b	0.715	100	b	41.8	93	f	a	a	+ 88
SURFACE WATER	CENTRAL WETLAND	02-CLW	11/30/89	32.4	100	b	0.613	100	b	44.3	87	f	a	a	- 100 *
SURFACE WATER	CENTRAL WETLAND	03-CLW	11/30/89	27.8	100	b	0.432	93	b	38.4	100	f	a	a	+ 456
SURFACE WATER	CENTRAL WETLAND	04-CLW	11/30/89	30.4	90	b	0.510	100	b	46.9	93	f	a	a	+ 40.3
SURFACE WATER	SOUTH WETLAND	01-SOW	11/28/89	2.4 *	100	b	0.460	93	b	41.4	93	f	a	a	- 100 *
SURFACE WATER	SOUTH WETLAND	02-SOW	11/28/89	24.2	100	b	0.330	97	b	51.7	87	f	a	a	- 100 *
SURFACE WATER	SOUTH WETLAND	03-SOW	11/28/89	21.6	100	b	0.463	97	b	42.7	100	f	30.44	36.53	- 100 *
SURFACE WATER	SOUTH WETLAND	04-SOW	11/28/89	20.7	100	b	0.406	97	b	41.7	100	f	a	a	- 100 *
SURFACE WATER	REFERENCE-CYPRESS POND	01-CPW	11/28/89	24.8	100	b	0.489	97	b	42.0	100	f	a	a	+ 167
SURFACE WATER	REFERENCE-SPRAY FIELD	01-SFW	11/28/89	23.0	90	b	0.499	97	b	43.3	93	f	a	a	+ 4
SEDIMENT	NORTH WETLAND	01-NOW	11/30/89	0 *	100	b	0.319 *	60 *	b	45.2	93	90	4.21	3.30	- 100 *
SEDIMENT	UNNAMED CREEK	01-UNC	11/30/89	0 *	0 *	5.3	0.499	87	b	28.6	100	33 *	a	a	- 80.4*
SEDIMENT	CENTRAL WETLAND	01-CLW	11/30/89	0 *	0 *	2.5	0.290 *	50 *	b	38.0	93	63 *	6.96	3.80	- 100 *
SEDIMENT	CENTRAL WETLAND	02-CLW	11/30/89	0 *	0 *	b	0.115 *	10 *	b	32.6	100	58 *	4.02	3.35	- 100 *
SEDIMENT	CENTRAL WETLAND	03-CLW	11/30/89	0 *	0 *	70.5	0 *	0 *	58.5	35.3	80	23 *	6.35	8.27	- 100 *
SEDIMENT	CENTRAL WETLAND	04-CLW	11/30/89	0 *	0 *	71	0.121 *	27	b	38.2	93	28 *	8.13	10.28	- 100 *
SEDIMENT	SOUTH WETLAND	01-SOW	11/28/89	0 *	0 *	69	0.352 *	43	b	35.3	100	40 *	6.54	5.79	- 100 *
SEDIMENT	SOUTH WETLAND	02-SOW	11/28/89	0 *	90	b	0.450	73	b	38.3	100	48 *	6.35	5.45	- 100 *
SEDIMENT	SOUTH WETLAND	03-SOW	11/28/89	0 *	70	b	0.473	90	b	38.7	80	58 *	10.29	11.24	- 94 *
SEDIMENT	SOUTH WETLAND	04-SOW	11/28/89	0.3 *	100	b	0.480	97	b	36.3	87	40 *	63.79	72.07	- 94 *
SEDIMENT	REFERENCE-CYPRESS POND	01-CPW	11/28/89	0 *	0 *	70.5	0 *	0 *	63	31.2	93	45 *	7.24	6.14	- 100 *
SEDIMENT	REFERENCE-SPRAY FIELD	01-SFW	11/28/89	11.8 *	100	b	0.547	90	b	38.9	87	67 *	66.13	69.68	- 79 *

* - SIGNIFICANTLY MORE TOXIC THAN CONTROL (p < 0.01)
a - NOT SIGNIFICANTLY MORE TOXIC THAN CONTROL (p < 0.01)
b - NO TEST CONDUCTED
c - SINGLE CONCENTRATION CHRONIC TEST CONDUCTED ON ALL SAMPLES
d - MULTI CONCENTRATION ACUTE TEST CONDUCTED ONLY IF CHRONIC TEST ORGANISMS DIE WITHIN 48 HOURS
e - ORGANISMS LIVED LONGER THAN 48 HRS IN CHRONIC TEST
f - GERMINATION TEST ONLY CONDUCTED ON SOIL AND SEDIMENT SAMPLES
+/- STIMULATION/RETARDATION; STIMULATED SAMPLES WERE NOT ANALYZED FOR SIGNIFICANT TOXICITY

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *Multiple endpoints are used. None of the measurement endpoints used would have been effective as a single indicator of response. Hence, a suite of metrics is necessary, not because we do not know the right one to use in a particular situation, but because no single metric will permit a risk characterization. The level of confidence increases with the number of different metrics used.*

- *A thorough description of the habitat was developed (by necessity abbreviated in this version); a thorough assessment was made of the toxicity of environmental media. By applying the full suite of toxicity tests, the study is able to identify sensitive species and fulfill its practical (and clearly stated) purpose: to provide baseline data to track remediation. Field details and documentation of methods are sufficient to make further sampling and testing replicable.*

- *In a component-centered study, reference sites were chosen that are comparable in hydrology, sediment, and vegetation to the study wetlands.*

Limitations include:

- *Little attention is given to comparing the wetlands to the designated reference sites, and the reference sites do not appear to have been fortunately chosen. One reference site (along with two study sites) is described as receiving runoff from the land application of treated domestic wastewater, a potentially confounding factor. The other reference site, Cypress Pond, chosen to be at some distance (see figure 2-2) from the Superfund sites, nevertheless shares with one study site the lowest level of dissolved oxygen and its sediment was found to be highly toxic. Further investigation of Cypress Pond oxygen levels and sediment toxicity is beyond the scope of the AWIS, to be sure, but these factors severely limit its use as a reference site.*

- *In this case study, contaminant levels and known effects are not used to establish a causal relationship between contaminants and their adverse effects.*

- *A possible limitation of the toxicity testing is the use of eluates of soil and sediments. Eluates primarily remove the water-soluble contaminants from soil and sediment, leaving potentially toxic insoluble or particle-bound chemicals in the solid phase. If the resulting eluate is toxic, that is significant, but a negative result cannot be considered conclusive. Even a positive result may not fully characterize the toxicity of the sediment, since other sublethal effects may be associated with the solid-phase contaminants.*

Comments on Analysis: Characterization of Ecological Effects (continued)

General comment:

- *In extrapolating from laboratory toxicity analyses to field situations, several issues need to be considered: (a) How do the sensitivities of native species differ from those of the test organisms? (b) Does a laboratory study offer a conservative estimate of the effect (because of differences in the exposure regime in the field) or would a field assay provide a more conservative estimate (because of the incorporation of indirect and synergistic effects)?*

2.3.3. Analysis: Characterization of Exposure

Hydrologic Assessment/Transport of Chemicals. The potential for interaction of surficial ground and surface waters in these wetlands is pronounced. The minimum bottom elevations of the Central and South Wetlands are approximately 35.0 and 33.2 ft mean sea level (MSL), respectively. The ground-water elevations in the vicinity of the Central and South Wetlands range from 33.59 to 36.80 ft MSL and 36.57 to 37.72 ft MSL, respectively (Canonie Environmental, 1990). From these data, the potential for ground water to be intercepted by the wetlands is clearly evident.

Furthermore, in the shallow soil lithology assessment (U.S. EPA, 1990a) no significant confining layer was found that would preclude vertical migration of site contaminants. Assuming the shallow soil lithology remains similar beneath the wetlands, particularly the South and Central areas, the wetlands represent a direct pathway for the exchange of contaminants between the ground- and surface-water regimes. Thus, a probable pathway is established for the movement of contaminants originating onsite to the wetlands. Equally apparent is that the same hydrologic connection provides a pathway for contaminated water to enter the ground-water system.

A ditch from Bay Drums and Peak Oil connects to the North Wetland. Surface and subsurface flow from these sites to the Central Wetland is also probable. The proximity of Reeves SEG to the North Wetland makes direct surface runoff appear probable, but some surface water from this site drains to the Unnamed Creek, which also drains the North Wetland.

Another potential source of contamination to the Central and South Wetlands is surface runoff from spray irrigation of treated effluent from a central sewage treatment facility. Surface-water drainage from this operation appears to influence the southern region of the South Wetland and the Spray Field Wetland.

Spatial Distribution of Contamination. Surface-water and sediment samples were collected from all five wetlands and the Unnamed Creek associated with the North Wetland. Each sample coincided with the location selected for toxicity testing (figure 2-2). Only grab samples were collected. Sediment samples were collected with a hand auger. All sample collection and processing in the field were conducted according to ESD standard operating procedures. The Spray Field and Cypress Wetlands served as reference sites for determining the nature and extent of chemical contamination in the North, Central, and South Wetlands. Surface-water and sediment analyses indicate that the chemical contamination of these three wetlands appears primarily related to the hazardous waste sites.

For metals, the surface water associated with the South Wetland and the Unnamed Creek was the most contaminated. Reeves SEG appears to be the principal source of metal contamination.

Maximum total phthalate ester concentrations in the surface water were associated with the four stations located in the South Wetland. The maximum concentrations were reported as potentially toxic to phytoplankton.

For aquatic sediments, the North Wetland and the Unnamed Creek were the most contaminated with metals. Lead and zinc concentrations at the Unnamed Creek were the highest levels reported and probably reflected the effects of surface drainage from Reeves SEG.

Organic contamination in the sediments was most pronounced in the Central Wetland, probably because the Central Wetland extended into the Bay Drums facility, in recent years actually connecting with onsite waste disposal ponds. Drainage from the Peak Oil site is also a probable source of contaminants.

Bioaccumulation of Contaminants. Whole-body contaminant levels in aquatic organisms living in or near the study sites were measured to provide an indication of exposure and to evaluate possible effects on higher levels of the food chain. This analysis may also reveal bioaccumulative chemicals of concern that exist in surface waters or sediments in concentrations too low to measure in the water itself. Chemicals found in elevated concentrations can be tracked during and after remedial efforts onsite as a measure of the success of those efforts.

Aquatic animals were sampled on January 18 and 19, 1989, from all wetland areas near the four industrial sites that constitute the areawide remedial investigation and from the two reference areas; samples were not taken within the property lines of the industrial sites. High water at the time of sampling made the Central Wetland extend well into the Bay Drums property, connecting the sources of toxicants; two sampling stations were located within the Central Wetland. An effort was made to collect at least 20 g of tissue for a fish and invertebrate species from each sampling area.

Samples of fish were collected at each sampling site with a small seine or a dip net. From each collection the largest individuals were retained for analysis. The fish species found throughout the wetlands were almost exclusively topwater, live-bearing fish that are not dependent on oxygenated water above the sediments for reproduction. In every case, samples for tissue analysis consisted exclusively of mosquito fish (*Gambusia affinis*). Samples were analyzed for all target compound list (TCL) metals and for TCL organics other than volatiles (table 2-7). Volatiles were not analyzed since the grinding step in sample preparation would be expected to drive off these compounds.

Table 2-7. Composite Whole Fish Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetlands Impact Study, Tampa, Florida (U.S. EPA, 1990c)

	01-CPW CYPRESS POND 01/18/89 0930 (MG/KG)	01-SFW SPRAY FIELD 01/18/89 1300 (MG/KG)	01-NOW NORTH WETLAND 01/19/89 1815 (MG/KG)	01-UNC UNNAMED CREEK 01/19/89 1630 (MG/KG)	03-UNC UNNAMED CREEK 01/19/89 1715 (MG/KG)	02-CLW CENTRAL WETLAND 01/19/89 1300 (MG/KG)	04-CLW CENTRAL WETLAND 01/19/89 0930 (MG/KG)	02-SOW SOUTH WETLAND 01/18/89 1430 (MG/KG)
INORGANIC ELEMENTS								
ALUMINUM	8.9	12	35	59	31	8.4	11	20
BARIUM	1.7	2.4	3.4	4.2	5.9	5.3	8.8	3.2
BORON	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
CALCIUM	10000	14000	10000	12000	10000	7300	10000	9300
COPPER	0.88	1.2	0.88	1.2	1.1	1.1	1.4	1.5
IRON	32	20	55	390	150	77	28	77
MAGNESIUM	380	490	400	370	370	300	390	360
MANGANESE	4.7	8.8	9.7	7.8	7.3	20	11	7.1
MERCURY	0.12	0.04	N/A	0.08	0.05	0.02	--	0.05
POTASSIUM	2500	3000	2500	2400	2500	2000	2600	2400
SODIUM	910	1000	940	1100	960	620	840	860
STRONTIUM	19	35	10	12	11	9.7	13	10
TITANIUM	1.3	1.7	1.3	1.6	1.4	1.0	1.3	1.3
ZINC	40	26	50	370	180	38	60	81
ZIRCONIUM	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
EXTRACTABLE ORGANIC COMPOUNDS								
BENZOIC ACID	--	0.97[a]	--	--	--	--	--	--

[a]Estimated value

-- MATERIAL WAS ANALYZED FOR BUT NOT DETECTED.

Crayfish were collected at all but two sampling locations—one relatively deep-water station and a second just outside the Reeves Galvanizing site where no aquatic or benthic invertebrates were found (table 2-8).

Overall, fish and crayfish sampled from the various wetland areas did not reveal a wide spectrum of contaminants at concentrations grossly over background. The primary exceptions were very high concentrations of iron and zinc found in samples from the Unnamed Creek, which receives runoff from the Reeves Galvanizing operation.

Several inorganics were widely present at concentrations moderately elevated over background. These include aluminum, barium, copper, iron, manganese, titanium, and zinc (table 2-9).

Similar overage tables were prepared for data from samples collected from each implicated industrial site to document potential hazard from these sources.

Comments on Analysis: Characterization of Exposure

- *The links between stressors (chemicals) and ecological components via specific exposure routes are not clear. The assessment would have been strengthened by illustrating how the spatial and temporal distribution of the stressors interface with the spatial distribution and life history patterns of the components. More information on the spatial distribution of contaminants and their fate and transport would have been helpful. This information was developed as part of the Remedial Investigation Study for the site but should be viewed as part of the overall risk assessment.*

2.3.4. Risk Characterization

The AWIS was conducted to document the contamination in the surface waters and sediments of the wetland areas, toxicity to plants and animals, and ecological functioning in the wetlands as a whole. This information establishes existing conditions and can be used to evaluate the efficacy of remedial actions in reducing toxicity and overall biological effects. As such, the work conducted as part of the AWIS serves as the first (i.e., baseline) step in an overall monitoring program. With regard to the characterization of risks under existing conditions, analytical data for each study site were compared with relevant criteria or other available effects-related information to link specific contaminants with adverse environmental effects (tables 2-10 and 2-11). A small portion of the original investigation results has been selected to illustrate the approach used.

Risks Based on Comparisons to Benchmarks. To characterize ecological risk, the concentration of each chemical identified in samples of environmental media was compared with a benchmark concentration known to produce an adverse biological effect. For water samples, the effective concentration chosen was the AWQC (table 2-12). In the absence of an AWQC, a

Table 2-8. Composite Whole Crayfish Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa, Florida (U.S. EPA, 1990c)

	01-CPW CYPRESS POND 01/18/89 1000 (MG/KG)	01-SFW SPRAY FIELD 01/18/89 1330 (MG/KG)	01-NOW NORTH WETLAND 01/19/89 1800 (MG/KG)	03-UNC UNNAMED CREEK 01/19/89 1730 (MG/KG)	04-CLW CENTRAL WETLAND 01/19/89 1030 (MG/KG)	02-SOW SOUTH WETLAND 01/18/89 1500 (MG/KG)
INORGANIC ELEMENTS						
ALUMINUM	10	14	45	68	15	31
BARIUM	3.1	4.2	7.2	7.5	9.2	6.5
BORON	N/A	N/A	N/A	N/A	N/A	N/A
CALCIUM	9600	16000	12000	25000	17000	16000
COPPER	4.1	7.8	5.4	6.8	7.9	8.4
IRON	48	24	140	700	25	150
MAGNESIUM	240	360	260	390	370	340
MANGANESE	2.4	3.3*	8.9	20	4.9	5.8
MERCURY	0.12	0.17*	0.04	0.02	N/A	0.03
NICKEL	1.6	--	--	--	--	--
POTASSIUM	1700	1800	2000	1900	2100	2200
SODIUM	1700	2000	1600	1700	1300	1600
STRONTIUM	37	80	23	42	41	33
TIN	2.4	--	--	--	--	--
TITANIUM	1.2	1.7	1.6	3.0	1.9	2.0
ZINC	11	20	22	180	18	21
ZIRCONIUM	N/A	N/A	N/A	N/A	N/A	N/A
EXTRACTABLE ORGANIC COMPOUNDS						
BENZOIC ACID	1.1*	--	--	--	--	--

*Estimated value
-- MATERIAL WAS ANALYZED FOR BUT NOT DETECTED.

Table 2-9. Concentration of Inorganic Elements in Wetland Tissue Samples as a Multiple of the Low Background Wetland Analysis,[a] Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa, Florida (U.S. EPA, 1990c)

	01-NOW NORTH WETLAND		01-UNC UNNAMED CREEK	03-UNC UNNAMED CREEK		02-CLW CENTRAL WETLAND	04-CLW CENTRAL WETLAND		02-SOW SOUTH WETLAND	
	FISH	CRAYFISH	FISH	FISH	CRAYFISH	FISH	FISH	CRAYFISH	FISH	CRAYFISH
ALUMINUM	3.9	4.5	6.6	3.5	6.8	1.2	2.2	1.5	-	3.1
BARIUM	2.0	2.3	2.5	3.4	2.4	1.9	5.2	3.0	1.9	2.1
CALCIUM	-	1.3	1.2	-	2.6	-	-	1.8	-	1.7
COPPER	-	1.3	1.4	1.3	1.7	1.3	1.6	1.9	1.7	2.0
IRON	2.8	5.8	19.5	7.5	29.2	3.9	1.4	-	3.9	6.3
MAGNESIUM	1.1	1.1	-	-	1.6	-	-	1.5	-	1.4
MANGANESE	2.1	3.7	1.7	1.6	8.3	4.3	2.3	2.0	1.5	2.4
MERCURY	N/A	-	2.0	1.3	-	-	-	-	1.3	-
NICKEL	-	-	-	-	-	-	-	-	-	-
POTASSIUM	-	1.2	-	-	1.1	-	-	1.2	-	1.3
SODIUM	-	-	1.2	1.1	-	-	-	-	-	-
STRONTIUM	-	-	-	-	1.1	-	-	1.1	-	-
TIN	-	-	-	-	-	-	-	-	-	-
TITANIUM	-	1.3	1.2	1.1	2.5	-	-	1.0	-	1.7
ZINC	1.9	2.0	14.2	6.9	16.4	1.5	2.3	1.6	3.1	1.9

[a]For reference values, see U.S. EPA, 1990c.

- = Concentration equal to or lower than the lower background station concentration.

Table 2-10. Wetland Surface Water Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa, Florida (U.S. EPA, 1990c)

WIS STATION ID BDRI STATION ID AWRI/FS STATION ID	01-CPW 01-RFW CYPRESS POND 11/28/89 1115	01-SFW 02-RFW SPRAY FIELD 11/28/89 1355	01-NOW A NORTH WETLAND 11/30/89	01-UNC B UNNAMED CREEK 11/30/89	01-CLW 5R CENTRAL WETLAND 1/4/90	02-CLW 6 CENTRAL WETLAND 11/30/89	03-CLW 7 CENTRAL WETLAND 11/30/89	04-CLW 8 CENTRAL WETLAND 11/30/89	01-SOW D SOUTH WETLAND 11/28/89	02-SOW E SOUTH WETLAND 11/28/89	03-SOW F SOUTH WETLAND 11/28/89	04-SOW G SOUTH WETLAND 11/28/89	02-CLW C CENTRAL WETLAND 11/30/89	01-UNC 01-SAP UNNAMED CREEK 1/9/90
INORGANIC ELEMENTS	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L	UG/L
ALUMINUM	1800	210	422	77600	625	B 111	206	390	11200	572	B 128	232	364	5500
ARSENIC	--	--	B 3	S 41.2	--	BW 3.2	--	--	B 6.2	--	--	--	--	--
BARIUM	--	--	B 22.8	268	B 34.4	B 41.8	B 13.7	B 16.6	B 145	B 8.3	B 23	B 19.5	B 20.4	--
BERYLLIUM	--	--	--	B 3.1	--	--	--	--	B 1.5	--	--	--	--	--
CADMIUM	--	--	--	9.8	--	--	--	--	B 4.0	--	--		--	--
CALCIUM	32000	58000	34800	185000	491000	E 61700	E 12300	E 12900	50100	E 9550	E 55500	E 51000	E 14500	80000
CHROMIUM	--	--	--	135	B 5.3	--	--	--	27.7	B 3.0	--	--	27.7	16
COBALT	--	--	--	18.6	--	--	--	--	B 4.5	--	--	--	--	--
COPPER	--	--	--	60.9	--	--	--	--	56.4	--	--	--	--	--
IRON	2100	--	841	678000	4640	743	211	260	11000	898	--	169	366	25000
LEAD	--	--	--	352	165	W 4.6	BW 2.5	W 4.6	248	W 4.9	BW 2.6	W 3.4	20.1	15
MAGNESIUM	8100	13000	B 328	178000	E 5520	8760	B 3270	B 3390	5710	B 1150	8180	7440	B 3550	6400
MANGANESE	--	65	49.7	1710	72.6	79.1	31.3	43.2	105	23.7	57.5	41.9	38.4	370
MOLYBDENIUM	NA	NA	--	--	--	--	--	--	--	--	--	--	--	
NICKEL	--	--	B 4.1	155	--	B .7	--	--	B 22.4	B 5.5	B 7.2	B 4.2	B 4.6	--
POTASSIUM	17000	39000	B 3260	8800	B 4360	6680	6670	6870	B 1170	B 1490	14400	12200	6660	9200
SILVER	--	--	--	N 10.3	--	--	--	--	--	--	--	--	--	--
SODIUM	180000	250000	29100	373000	288000	74500	57500	59100	111000	35600	171000	185000	59700	94000
VANADIUM	--	--	--	59.9	B 15.1	--	--	--	B 30.5	--	--	--	--	--
ZINC	--	--	48.7	172000	E 410	49.1	B 19	34.5	3980	63.2	67.7	45.9	32.8	11000
GENERAL INORGANIC PARAMETERS	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L
CYANIDE	0.01	0.01	0.01	0.04	0.01	0.016	0.019	0.01	0.015	0.01	0.019	0.01	0.015	--

Table 2-10. Wetland Surface Water Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa, Florida (continued)

WIS STATION ID	01-CPW	01-SFW	01-NOW	01-UNC	01-CLW	02-CLW	03/CLW	04-CLW	01-SOW	02-SOW	03-SOW	04-SOW	02-CLW	01-UNC
BDRI STATION ID	01-RFW	02-REW												01-SAP
AWRI/FS STATION ID			A	B	5R	6	7	8	D	E	F	G	C	
	CYPRESS POND	SPRAY FIELD	NORTH WETLAND	UNNAMED CREEK	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOUTH WETLAND	CENTRAL WETLAND	UNNAMED CREEK
	11/28/89	11/28/89	11/30/89	11/30/89	1/4/90	11/30/89	11/30/89	11/30/89	11/28/89	11/28/89	11/28/89	11/28/89	11/30/89	1/9/90
	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	MG/L	UG/L
PURGEABLE ORGANIC COMPOUNDS														
ACETONE	--	--	B 0.0100	B 0.0100	--	--	--	BJ 0.0090	--	--	--	--	B 0.0100	--
CARBON DISULFIDE	.039	.042	--	--	--	--	--	--	0.0580	0.0280	J 0.0340	0.0190	--	--
ETHYLTRIAZOLE	.200 JN	--	--	--	--	--	--	--	--	--	--	--	--	
METHYL ETHYL KETONE	.010 UR	.010 UR	--	--	--	--	--	--	--	--	--	--	--	--
METHYLENE-CHLORIDE	--	--	--	--	--	--	--	BJ 0.0810	--	--	--	--	--	--
EXTRACTABLE ORGANIC COMPOUNDS														
BIS (2-ETHYLHEXYL) PHTHALATE	--	--	--	BJ 0.0040	--	BJ 0.0030	BJ 0.0040	BJ 0.0030	BJ 0.0080	B 0.0120	BJ 0.0070	BJ 0.0080	BJ 0.0030	--
BUTYL BENZYL PHTHALATE	--	--	--	--	--	--	--	--	J 0.0090	J 0.0030	--	--	--	--
DI-N-BUTYL PHTHALATE	--	--	--	J 0.0020	--	--	J 0.0020	--	B 0.0920	B 0.0910	B 0.0470	B 0.0180	--	--
DI-N-OCTYL PHTHALATE	--	--	--	--	--	--	--	--	--	B 0.0110	--	--	--	--
IDENO (1.2.3-CD) PYRENE	--	--	--	--	--	--	--	--	--	B 0.0110	--	--	--	--
4-METHYL PHENOL	--	--	--	--	--	--	--	--	--	--	J 0.0030	--	--	
ORGANOCHLORINE PESTICIDE ANALYSIS														
PCB-1260	--	--	--	--	--		--	0.0010	--	--	--	--	--	--

Table 2-10. Wetland Surface Water Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa, Florida (continued)

The following are AWRI/FS qualifiers:

B - This flag is used when the analyte is found in the associated blank as well as in the sample.

E - This flag identifies compounds whose concentrations exceed the calibration range of the GC/MS instrument for that specific analysis.

J - Indicates an estimated value.

The following are AWRI/FS qualifiers for inorganic analysis:

N - The spiked sample recovery was not within control limits.

S - The value reported was determined by the Method of Standard Additions (MSA).

W - The post-digestion spike for furnace AA analysis is outside of the 85-115% control limits while sample absorbance is less than 50% of the spike absorbance.

BDRI Station qualifier:

NA - Not analyzed.

General Data Qualifier:

– The analyte was analyzed for but not detected.

WIS: U.S. EPA, Region 4, Environmental Services Division, Ecological Services Branch. Ausust 1990. Wetland Impact Study and Environmental Assessment. Bay Drums, Peak Oil, and Reeves Southeastern Superfund Sites, Tampa, Florida.

BDRI: U.S. EPA, Region 4, Environmental Services Division, Environmental Compliance Branch, June 25, 1990. Bay Drums First Draft Working Document.

AWRI/FS: Canonie Environmental. February 1990. Area-Wide RI/FS. Bay Drums, Peak Oil, and Reeves Southeastern Sites, Tampa, Florida.

Sources: Data for samples 01-CPW and 01-SFW from BDRI.
Data for all other samples for AWRI/FS.

Table 2-11. Wetland Sediment Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa Florida (U.S. EPA, 1990c)

WIS STATION ID	01-CPW	01-SFW	01-NOW	01-UNC	$01\text{-}CLW_a$	02-CLW	03-CLW	04-CLW	01-SOW	02-SOW	03-SOW	04-SOW	02-CLW	01-UNC
BORI STATION ID	01-RFW	02-RFW												01-SAP
AWRI/FS STATION ID			A	B·	5R	6	7	8	D	E	F	G	C	
	CYPRESS POND	SPRAY FIELD	NORTH WETLAND	UNNAMED CREEK	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOW WETLAND	CENTRAL WETLAND	UNNAMED CREEK
	11/28/89 1115	11/28/89 1355	11/30/89	11/30/89	1/4/90	11/30/89	11/30/89	11/30/89	11/30/89	11/28/89	11/28/89	11/28/89	11/30/89	1/9/90
INORGANIC ELEMENTS	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG
ALUMINUM	4900	440	B 4070	E 3790	*0.70	E 1030	E 1120	E 794	E 1640	E 2160	E 1360	E 839	E 629	220J
ANTIMONY	--	--	--	--	--	--	B 3.70	--	--	--	--	--	--	--
ARSENIC	--	--	12.70	--	B 0.0014	--	B 0.80	--	--	B 1.0	B 1.40	--	--	--
BARIUM	22	--	57.70	B 26.70	B 0.0164	B 13.10	B 8.50	B 4.90	B 4.0	B 7.0	B 7.40	B 4.80	B 11.30	--
CADMIUM	--	--	B 1.10	B 0.800	--	B 0:45	--	--	B 0.36	--	--	--	--	--
CALCIUM	2200	--	E 5910	E 8210	B 1.220	E 3340	E 1060	BE 590	BE 737	E 1850	E 1550	E 1590	BE 597	550
CHROMIUM	8.1	--	22.30	21.9	0.0044	8.60	2.600	2.10	3.300	3.50	3.30	2.50	2.0	--
COBALT	--	--	B 1.70	B 1.70	--	B 0.91	--	--	--	--	--	--	--	--
COPPER	--	--	12.90	11.10	B 0.0057	11.600	B 1.10	B 0.72	B 0.63	B 1.60	B 1.60	B 0.88	--	--
IRON	720	89	1980	22600	* 0.414	1520	308	161	208	486	279	232	150	390J
LEAD	43J	2.9J	266	70.80	N 0.4150	65.300	11.30	8.40	7.90	11.10	6.90	7.10	38.10	18
MAGNESIUM	170	--	B 217	B 401	B 0.0366	B 240	B 153	B 83.90	B 47.10	B 70.60	B 73.10	B 66.90	B 83.10	--
MANGANESE	--	--	21.60	62.60	0.0092	10.40	2.90	B 1.70	B 1.60	3.10	B 2.20	B 1.30	B 1.80	--
MERCURY	--	--	*1.10	*0.22	--	*0.24	*0.14	*0.11	--	*0.08	*0.08	*0.06	*0.09	--
NICKEL			B 4.50	9.70	--	B 2.40	B 1.30	B 0.64	--	B 2.20	B 1.60	--	--	--
POTASSIUM			B 250	B 226	--	B 261	B 220	B 0.198	B 156	B 152	B 187	B 148	B 157	--
SELENIUM	3.5J	--	--	--	--	--	--	--	--	B 0.75	--	--	--	--
SODIUM	550	--	B 255	1280	--	--	B 332	B 286	B 192	B 161	B 421	B 341	B 200	--
VANADIUM	13	--	B 6.60	B 6.80	B 0.0015	B 3.30	B 1.90	B 1.50	B 1.60	b 2.80	B 2.50	B 1.80	B 1.10	1.3
ZINC	29	--	EN 355	EN 11200	N 0.5410	EN 402	EN 42.60	EN 32.4	EN 140	EN 206	EN 234	EN 109	EN 25.60	--
:NERAL INORGANIC PARAMETERS														
CYANIDE	--	--	N 0.54	N 5.70	--	--	--	--	--	--	--	--	--	--

Table 2-11. Wetland Sediment Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa Florida (continued)

WIS STATION ID BORI STATION ID AWRI/FS STATION ID	01-CPW 01-RFW CYPRESS POND 11/28/89	01-SFW 02-RFW SPRAY FIELD 11/28/89	01-NOW A NORTH WETLAND 11/30/89	01-UNC$_a$ B UNNAMED CREEK 11/30/89	01-CLW 5R CENTRAL WETLAND 1/4/90	02-CLW 6 CENTRAL WETLAND 11/30/89	03-CLW 7 CENTRAL WETLAND 11/30/89	04-CLW 8 CENTRAL WETLAND 11/30/89	01-SOW D SOUTH WETLAND 11/30/89	02-SOW E SOUTH WETLAND 11/28/89	03-SOW F SOUTH WETLAND 11/28/89	04-SOW G SOUTH WETLAND 11/28/89	02-CLW C CENTRAL WETLAND 11/30/89	01-UNC 01-SAP UNNAMED CREEK 1/9/90
PURGEABLE ORGANIC COMPOUNDS	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG
ACETONE	R	R	B 0.1300	B 0.0440	B 0.0800	B 0.060	B 0.3800	B 0.130	B 0.0160	B 0.0440	B 0.0500		B 0.0250	--
ENZENE	R	R	J 0.003	--	--	J 0.0020	J 0.0110	J 0.0050	--	--	--		--	--
2-BUTANONE	R	R	0.033	J 0.0110	--	--	--	--	0.0140	--	--		0.0320	--
METHYLENE CHLORIDE	R	R	B 0.1300	B 0.0080	B 0.0210	B 0.0290	B 0.1900	B 0.1300	B 0.0330	B 0.0150	B 0.0130		B 0.0140	--
TOLUENE	R	R	B 0.0140	BJ 0.007	0.0340	B 0.0520	B 0.9300	B 0.5800	B 0.0810	B 0.1500	B 0.0140		B 0.0380	--
XYLENE(S)	R	R	J 0.0070	--	--	--	--	J 0.0140	--	--	--		--	--
EXTRACTABLE ORGANIC COMPOUNDS														
(3-AND/OR 4-) METHYLPHENOL	290J	--	--	--	--	--	--	--	--	--	--	--	--	--
BENZOIC ACID	--	--	J 0.1900	--	J 0.0870	J 0.2600	J 0.5400	J 0.0440	--	--	--	--	--	--
BENZO(A)-ANTHRACENE	--	--	--	--	J 0.0820	--	J 0.3600	--	--	--	--	--	--	--
BENZO(B)-FLUOR-ANTHENE	--	--	--	JX 0.120	JX 0.2700	--	J 0.2000	--	--	--	--	--	--	
BENZO(K) FLUOR-ANTHENE	--	--	--	JX 0.1200	JX0.2700	--	J 0.1700	--	--	--	--	--	--	
BENZO-(G.H.I.) PERYLENE	--	--	--	--		--	J 0.2200	--	--	--	--	--	--	--
BENZO(A) PYRENE	--	--	--	J 0.0560	J 0.0910	--	J 0.2400	--	--	--	--	--	--	--
BIS (2-ETHYL-HEXYL) PHTHALATE	--	--	J 0.1400	J 0.2300	J 0.0490	J 0.2300	J 0.4300	J 0.0510	--	--	J 0.0530	J 0.0750	J 0.0850	--
BUTYL BENZYL PHTHALATE	--	--	--	--	--	--	J 0.2200	--	--	--	--	--	--	--
CHRYSENE	--	--	--	J 0.0600	J 0.1200	--	J 0.2000	--	--	--	--	--	--	--

Table 2-11. Wetland Sediment Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa Florida (continued)

WIS STATION ID	01-CPW	01-SFW	01-NOW	01-UNC	01-CLW	02-CLW	03-CLW	04-CLW	01-SOW	02-SOW	03-SOW	04-SOW	02-CLW	01-UNC
BDRI STATION ID	01-RFW	02-RFW												01-SAP
AWRI/FS STATION ID			A	B	5R	6	7	8	D	E	F	G	C	
	CYPRESS POND	SPRAY FIELD	NORTH WETLAND	UNNAMED CREEK	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	CENTRAL WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOUTH WETLAND	SOUTH WETLAND	CENTRAL WETLAND	UNNAMED CREEK
	11/28/89	11/28/89	11/30/89	11/30/89	1/4/90	11/30/89	11/30/89	11/30/89	11/30/89	11/28/89	11/28/89	11/28/89	11/30/89	1/9/90
	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG	MG/KG
DIBENZO (A.H) ANTHRACENE	--	--	--	--	--	--	J 0.2000	--	--	--	--	--	--	--
DI-N-BUTYL-PHTHALATE	--	--	--	1.3000	--	J 0.1100	J 0.1500	--	--	--	--	J 0.3900	J 0.0740	--
DI-N-OCTYL PHTHALATE	--	--	--	--	--	--	J 0.1500	--	--	--	--	--	--	--
FLUORANTHENE	--	--	--	--	J 0.0750	--	--	--	--	--	--	--	--	--
HEXADECENOIC ACID	3000JN	--	--	--	--	--	--	--	--	--	--	--	--	
INDENO-(1.2.3-CD) PYRENE	--	--	--	--	--	--	J 0.2200	--	--	--	--	--	--	--
NITROBENZENE	1800UR	920UR	--	--	--	--	--	--	--	--	--	--	--	--
4-NITRO PHENOL	--	--	--	--	--	--	J 0.2200	--	--	--	--	--	--	--
PHENOL	--	--	--	J 0.0910	--	--	J 0.3700	--	J 0.0540	--	J 0.0760	J 0.3600	J 0.0450	--
PYRENE	--	--	--	J 0.0610	J 0.0800	--	--	--	--	--	--	--	--	--
?GANOCHLORINE PESTICIDES ANALYSIS														
GAMMA CHLORDANE	--	--	J 0.0970	--	--	J 0.1000	--	--	--	--	--	--	--	--
4.4'-DDD	--	--	J 0.0320	--	--	--	--	--	--	--	--	--	--	--
4.4'-DDE	--	--	0.1200	--	--	0.1200	--	--	--	--	--	--	--	--
PCB-1260	--	--	0.0810	--	0.2600	J 0.4800	--	--	--	J 0.4900	--	--	--	--

Table 2-11. Wetland Sediment Analytical Data Summary, Bay Drums, Peak Oil, and Reeves SE Areawide Wetland Impact Study, Tampa Florida (continued)

The following are AWRI/FS qualifiers:

B - This flag is used when the analyte is found in the associated blank as well as in the sample.
J - Indicates an estimated value.
X - Other specific flags and footnotes may be required to properly define the results.

The following are AWRI/FS qualifiers for inorganic analysis:

B - The reported value is less than the contract required detection limit (CRDL) but greater than the instrument detection limit (IDL).
E - The reported value is estimated because of interference.
N - The spiked sample recovery was not within control limits.
* - Refer to the original lab report for the Form 1, Sample Data Summary, Case Narrative.
Also the "X" flag definition is specific for each result; refer to the original lab report to find out what combination of flags "X" stands for.

The following are BDRI station qualifiers:

J - Estimated value.
N - Presumptive evidence of presence of material.
R - Quality control indicates that data are unusable, compound may or may not be present, resampling and reanalysis are necessary for verification, the value is that reported by the laboratory. All purgeable organic compounds were reported as "R" for sediment samples 01-CPW and 01-SFW.
U - Material was analyzed for but not detected. The number shown is the minimum quantitation limit.

General data qualifier:

– The analyte was analyzed for but not detected.

WIS: U.S. EPA, Region 4, Environmental Services Division, Ecological Services Branch. August 1990.
Wetland Impact Study and Environmental Assessment. Bay Drums, Peak Oil, and Reeves Southeastern Superfund Sites, Tampa, Florida.

BDRI: U.S. EPA, Region 4, Environmental Services Division, Environmental Compliance Branch, June 25, 1990.
Bay Drums First Draft Working Document.

AWRI/FS: Canonie Environmental. February 1990. Area-Wide RI/FS. Bay Drums, Peak Oil, and Reeves Southeastern Superfund Sites, Tampa, Florida.
[a]It appears that metals and cyanide values reported in the AWRI/FS as originating with the Compuchem subcontractor are reported about 3 orders of magnitude lower than is probable. We have interpreted these values for the purpose of this WIS to be g/kg rather than mg/kg.

Sources: Data for samples 01-CPW and 01-SFW from BDRI.
Data for all other samples from AWRI/FS.

Table 2-12. Summary of EPA Ambient Water Quality Criteria and Screening Concentrations for the Protection of Freshwater Biota (U.S. EPA, 1990c, adapted from table 4.28)

TSS790 SCREENING LIST
UPDATE: JANUARY 1991
EPA REG IV - WATER MANAGEMENT DIVISION
304(a) SCREENING VALUES AND RELATED INFORMATION FOR TOXIC POLLUTANTS

DATE REVISED		COMPOUND	EPA DETECTION LEVEL [40 CFR 136] (µg/L)	Ref. Method	(µg/L)	Ref. Method	FRESHWATER Screening Value (Maximum) (µg/L)	Screening Value (Continuous) (µg/L)	95% LC50 Value (µg/L)	CRITERIA DATES
							Hardness(mg/l as CaCO3):		50.0	
							pH:		6	
		PRIORITY POLLUTANTS								
4/89	1	m Antimony (B)	200	204.1	3	204.2	1300 2s	160 2s	1300	10/80, 1/87:RfD 0.0004
7/89	2	m Arsenic (c)	2	206.3	1	206.2	360 *III	190 *III	720 *III	1/85:aq life, 6/21/88:q1* 1.75 Admin. memo
7/90	3	m Beryllium (c)	5	210.1	0.2	210.2	16 6s	0.53 1s	16 6s	10/80, 1/90:q1* 4.3
7/90	4	m Cadmium (H)	5	213.1	0.1	213.2	1.79 *	0.66 *	3.59 *	10/80,1/85:aq life, 10/89:RfD 0.0005(water) 0.001(food)
12/89	5	m Chromium (III) (H)	50	218.1	1	218.2	984.32 *	117.32 *	1968.63 *	10/80, 1/85:aq life, 3/88:RfD 1
7/90	5	m Chromium (VI)	5	218.4			16 *	11 *	32 *	10/80, 1/85:aq life, 3/88:RfD 0.005
1/91	6	m Copper (H)	20	220.1	1	220.2	9.22 *	6.54 *	18.45 *	10/80, 1/85:aq life
4/89	7	m Lead (H)	100	239.1	1	239.2	33.78 *	1.32 *	67.57 *	10/80, 1/85:aq life
6/89	8	m Mercury	0.2	245.1	--		2.40 *	0.012 *T	4.8 *	10/80, 1/85:aq life, 2/89:RfD 0.0003
4/89	9	m Nickel (H)	40	249.1	1	249.2	789.00 *	87.71 *	1578.01 *	10/80, 9/86:aq life, 3/88:RfD 0.02
7/90	10	m Selenium	2	270.3	2	270.2	20.00 *	5.00 *	40 *	10/80, 9/87:aq life
7/90	11	m Silver (H, B)	10	272.1	0.2	272.2	1.23 *	0.012 2s	1.23 *	10/80, 6/88:RfD 0.003
11/89	12	m Thallium	100	279.1	1	279.2	140.00 3s	4.00 2s	140	10/80
1/91	13	m Zinc (H)	5	289.1	0.05	289.2	65.04 *	58.91 *	130.09 *	10/80, 2/87:aq life
7/90	14	Cyanide	5	335.3	--		22 *	5.2 *	44 *	10/80, 1/85:aq life, 3/88:RfD 0.02
4/89		*Asbestos (c)*					--	--	--	*10/80*
7/89		2,3,7,8-TCDD-Dioxin (c)	0.00001	hrms	0:002	613	0.1	0.00001 T	0.1	2/84:q1* 156000
4/89	1	v Acrolein	nr	624x	0.7	603	6.8 3s	2.1 2s	6.8	10/80
4/89	2	v Acrylonitrile (c)	nr	624x	0.5	603	755 4s	75.5	755	10/80, 2/89:q1* 0.54
4/89	3	v Benzene (c)	4.4	624	0.2	602	530 7s	53	530	10/80, 12/88:q1* 0.029
1/91	5	v Bromoform (c)	4.7	624	0.2	601	2930 2s	293	2930	10/80, 6/88:q1* CHCl3, 9/90:q1* 0.0079
6/89	6	v Carbon Tetrachloride (c)	2.8	624	0.12	601	3520 3s	352	3520	10/80, 3/88:q1* 0.13
1/91	7	v Chlorobenzene	6	624	0.25	601	1950 5s	195	1950	10/80, 11/90:RfD 0.02
1/91	8	v Chlorodibromomethane (c)	3.1	624	0.09	601	--	--	--	10/80, 6/88:q1* CHCl3, 11/90:q1* 0.084
4/89	9	v Chloroethane	nr	624	0.52	601	--	--	--	10/80
7/90	10	v 2-Chloroethylvinyl Ether (c)	nr	624	0.13	601	35400 1s	3540	35400	10/80
4/89	11	v Chloroform (HM, c)	1.6	624	0.05	601	2890 3s	289	2890	10/80, 6/88:q1* 0.0061
1/91	12	v Dichlorobromomethane (c)	2.2	624	0.1	601	--	--	--	10/80, 6/88:q1* CHCl3, 10/90:q1* 0.13
4/89	14	v 1,1-Dichloroethane	4.7	624	0.07	601	--	--	--	10/80
4/89	15	v 1,2-Dichloroethane (c)	2.8	624	0.03	601	11800 3s	2000 1s	11800	10/80, 3/88:q1* 0.091
4/89	16	v 1,1-Dichloroethylene (c)	2.8	624	0.13	601	3030 3s	303	3030	10/80, 12/88:q1* 0.6
4/89	17	v 1,2-Dichloropropane	6	624	0.04	601	5250 3s	525	5250	10/80
11/89	18	v 1,3-Dichloropropylene (Cis)	5		0.34		606 2s	24.4 1s	606	10/80, 3/88:RfD 0.0003
11/89		v 1,3-Dichloropropylene (Trans)	nr	624	0.2	601	606 2s	24.4 1s	606	10/80, 3/88:RfD 0.0003
4/89	19	v Ethylbenzene	7.2	624	0.2	602	4530 5s	453	4530	10/80, 3/88:RfD 0.1
7/90	20	v Methyl Bromide	nr	624	1.18	601	1100 1s	110	1100	10/80, 8/90:RfD 0.0014

Table 2-12. Summary of EPA Ambient Water Quality Criteria and Screening Concentrations for the Protection of Freshwater Biota (continued)

TSS790 SCREENING LIST
UPDATE: JANUARY 1991
EPA REG IV - WATER MANAGEMENT DIVISION
304(a) SCREENING VALUES AND RELATED INFORMATION
FOR TOXIC POLLUTANTS

DATE REVISED	COMPOUND	EPA DETECTION LEVEL [40 CFR 136] (µg/L)	Ref. Method	(µg/L)	Ref. Method	FRESHWATER Screening Value (Maximum) (µg/L)		Screening Value (Continuous) (µg/L)		95% LC50 Value (µg/L)	CRITERIA DATES
4/89	21 v Methyl Chloride (HM, c)	nr	624	0.08	601	55000	1s	5500		55000	10/80, 6/88:q1* CHCl3
4/89	22 v Methylene Chloride (c)	2.8	624	0.25	601	19300	3s	1930		19300	10/80, 1/89:q1* 0.0075
6/89	23 v 1,1,2,2-Tetrachloroethane (c)	6.9	624	0.03	601	932	3s	240	1s	932	10/80, 3/88:q1* 0.2
4/89	24 v Tetrachloroethylene (c)	4.1	624	0.03	601	528	5s	84	1s	528	10/80
1/91	25 v Toluene	6	624	0.2	602	1750	5s	175		1750	10/80, 3/88:RfD 0.3, 8/90:RfD 0.2
7/90	26 v 1,2-Trans-Dichloroethylene	1.6	624	0.1	601	13500	1s	1350		13500	10/80, 1/89:RfD 0.02
7/90	27 v 1,1,1-Trichloroethane	3.8	624	0.03	601	5280	2s	528		5280	10/80, 6/88:RfD 0.09
4/89	28 v 1,1,2-Trichloroethane (c)	5	624	0.02	601	3600	3s	940	1s	3600	10/80, 3/88:q1* 0.057
7/89	29 v Trichloroethylene (c)	1.9	624	0.12	601						10/80
4/89	31 v Vinyl Chloride (c)	nr	624	0.18	601	--		--		--	10/80
1/91	1 a 2-Chlorophenol	3.3	625	0.31	604	438	5s	43.8		438	10/80, 8/88:RfD 0.005
1/91	2 a 2,4-Dichlorophenol	2.7	625	0.39	604	202	3s	36.5	1s	202	10/80, 6/88:RfD 0.003
1/91	3 a 2,4-Dimethylphenol	2.7	625	0.32	604	212	3s	21.2		212	10/80, 11/90:RfD 0.02
4/89	4 a 2-Methyl-4,6-Dinitrophenol	24	625	16	604	23	4s	2.3		23	10/80
4/89	5 a 2,4-Dinitrophenol	42	625	13	604	62	3s	6.2		62	10/80, 3/88:RfD 0.002
4/89	6 a 2-Nitrophenol	3.6	625	0.45	604	--		3500		--	10/80
4/89	7 a 4-Nitrophenol	2.4	625	2.8	604	828	3s	82.8		828	10/80
1/91	8 a 3-Methyl-4-Chlorophenol	3	625	0.36	604	3	1s	0.3		3	10/80
1/91	9 a Pentachlorophenol (pH)	3.6	625	7.4	604	3.32 *		2.10 *		29.98 *	10/80, 9/86:aq life, 6/88:RfD 0.03
1/91	10 a Phenol	1.5	625	0.14	604	1020	16s	256	1s	1020	10/80, 6/89:RfD 0.6
7/90	11 a 2,4,6-Trichlorophenol (c)	2.7	625	0.64	604	32	3s	3.2		32	10.80, 6/90:q1* 0.011
1/91	1 bn Acenaphthene	1.9	625	1.8	610	170	2s	17		170	10/80, 11/90:RfD 0.06
1/91	2 bn Acenaphthylene	3.5	625	2.3	610	--		--		--	10/80
1/91	3 bn Anthracene	1.9	625	0.66	610	--		--		--	10/80, 9/90:RfD 0.3
4/89	4 bn Benzidine (c)	44	625	--		250	4s	25		250	10/80, 3/88:q1* 230
4/89	5 bn Benzo(a)Anthracene (PAH, c)	7.8	625	0.013	610	--		--		--	10/80
4/89	6 bn Benzo(a)Pyrene (PAH, c)	2.5	625	0.023	610	--		--		--	10/80
4/89	7 bn 3,4-Benzofluoranthene (PAH, c)	2.5	625	0.018	610	--		--		--	10/80
1/91	8 bn Benzo(ghi)Perylene	4.1	625	0.076	610	--		--		--	10/80
4/89	9 bn Benzo(k)Fluoranthene (PAH,c)	2.5	625	0.017	610	--		--		--	10/80
4/89	10 bn Bis(2-Chloroethoxy)Methane	5.3	625	0.5	611	--		--		--	10/80
'/89	11 bn Bis(2-Chloroethyl)Ether (c)	5.7	625	0.3	611	23800	1s	2380		23800	10/80, 3/85:q1* 1.1
/89	12 bn Bis(2-Chloroisopropyl)Ether	5.7	625	0.8	611	--		--		--	10/80, 10/89:RfD 0.04
39	13 bn Bis(2-Ethylhexyl)Phthalate (c, B	2.5	625	2	606	1110	2s	<0.3	2s	1110	10/80, 2/89:q1* 0.014
89	14 bn 4-BromophenylPhenyl Ether	1.9	625	2.3	611	36	2s	12.2	1s	36	10/80
,1/89	15 bn Butylbenzyl Phthalate	2.5	625	0.34	606	330	4s	22	2s	330	10/80, 9/89:RfD 0.2
1/91	16 bn 2-Chloronaphthalene	1.9	625	0.94	612	--		--		--	10/80, 11/90:RfD 0.08
4/89	17 bn 4-Chlorophenyl Phenyl Ether	4.2	625	3.9	611	--		--		--	10/80
1/91	18 bn Chrysene (PAH, c)	2.5	625	0.15	610	--		--		--	10/80
4/89	19 bn Dibenz(a,h)Anthracene (PAH, c)	2.5	625	0.03	610	--		--		--	10/80
11/89	20 bn 1,2-Dichlorobenzene	nr	624	1.9	625	158	4s	15.8	3s	158	10/80, 8/89:RfD 0.09
4/89	21 bn 1,3-Dichlorobenzene	nr	624	1.9	625	502	3s	50.2		502	10/80
7/90	22 bn 1,4-Dichlorobenzene	nr	624	4.4	625	112	5s	11.2		112	10/80
1/91	23 bn 3,3'-Dichlorobenzidine (c)	16.5	625	0.13	605	--		--		--	10/80, 8/90: q1* 0.45

Table 2-12. Summary of EPA Ambient Water Quality Criteria and Screening Concentrations for the Protection of Freshwater Biota (continued)

TSS790 SCREENING LIST
UPDATE: JANUARY 1991
EPA REG IV - WATER MANAGEMENT DIVISION
304(a) SCREENING VALUES AND RELATED INFORMATION
FOR TOXIC POLLUTANTS

DATE REVISED	COMPOUND	EPA DETECTION LEVEL [40 CFR 136] (µg/L)	Ref. Method	(µg/L)	Ref. Method	FRESHWATER Screening Value (Maximum) (µg/L)	Screening Value (Continuous) (µg/L)	95% LC50 Value (µg/L)	CRITERIA DATES
11/89	24 bn Diethyl Phthalate	1.9	625	0.49	606	5210 2s	521	5210	10/80, 9/87:RfD 0.8
4/89	25 bn Dimethyl Phthalate	1.6	625	0.29	606	3300 2s	330	3300	10/80
4/89	26 bn Di-n-Butyl Phthalate	2.5	625	0.36	606	94 6s	9.4	94	10/80, 1/87:RfD 0.1
4/89	27 bn 2,4-Dinitrotoluene (c)	5.7	625	0.02	609-EC	3100 2s	310	3100	10/80
4/89	28 bn 2,6-Dinitrotoluene	1.9	625	0.01	609-EC	--	--	--	10/80
4/89	29 bn Di-n-Octyl Phthalate	2.5	625	3	606	--	--	--	10/80
4/89	30 bn 1,2-Diphenylhydrazine (c)	20	1625			27 2s	2.7	27	10/80, 3/88:q1* 0.8
1/91	31 bn Fluoranthene	2.2	625	0.21	610	398 2s	39.8	398	10/80, 9/90:RfD 0.04
1/91	32 bn Fluorene	1.9	625	0.21	610	--	--	--	10/80, 9/90:RfD 0.04
4/89	33 bn Hexachlorobenzene (c, B)	1.9	625	0.05	612	--	--	--	10/80
7/89	34 bn Hexachlorobutadiene (c)	0.9	625	0.34	612	9 5s	0.93 1s	9	10/80, 3/88:q1*, 6/89:q1* 0.078
1/91	35 bn Hexachlorocyclopentadiene	nr	625	0.4	612	0.7 4s	0.07	0.7	10/80, 3/88:RfD 0.007
4/89	36 bn Hexachloroethane (c)	1.6	625	0.03	612	98 5s	9.8	98	10/80, 3/88:q1* 0.014
4/89	37 bn Indeno(1,2,3-cd)Pyrene (PAH, c)	3.7	625	0.043	610	--	--	--	10/80
1/91	38 bn Isophorone (c)	2.2	625	15.7	609-EC	11700 2s	1170	11700	10/80, 9/89:RfD 0.2, 8/90:q1* 0.0041
4/89	39 bn Naphthalene	1.6	625	1.8	610	230 4s	62 1s	230	10/80
12/89	40 bn Nitrobenzene	1.9	625	13.7	609-EC	2700 2s	270	2700	10/80, 5/88:RfD 0.0005
10/90	41 bn N-Nitrosodimethylamine (c)	nr	625	0.15	607	--	--	--	10/80, 3/88:q1* 51
1/91	42 bn N-Nitrosodi-n-Propylamine (c)	nr	625	0.46	607	--	--	--	10/80, 3/88:q1* 7.0
4/89	43 bn N-Nitrosodiphenylamine (c)	1.9	625	0.81	607	585 2s	58.5	585	10/80, 3/88:q1* 0.0049
1/91	44 bn Phenanthrene (B)	5.4	625	0.64	610	--	--	--	10/80
1/91	45 bn Pyrene	1.9	625	0.27	610	--	--	--	10/80, 9/90:RfD 0.03
11/89	46 bn 1,2,4-Trichlorobenzene	1.9	625	0.05	612	150 4s	44.9 1s	150	10/80
7/90	1 p Aldrin (c)	1.9	625	0.004	608	3 *	0.3	3 *	10/80, 12/88: q1* 17
4/89	2 p a-BHC (c)	nr	625	0.003	608	--	500 p	--	10/80, 3/88:q1* 6.3
4/89	3 p b-BHC (c)	4.2	625	0	608	--	5000 p	--	10/80, 9/87:q1* 1.8
4/89	4 p g-BHC (c)	nr	625	0	608	2 *	0.08 *	2 *	10/80
4/89	5 p d-BHC (c)	3.1	625	0.009	608	--	--	--	10/80
4/89	6 p Chlordane (c)	nr	625	0.014	608	2.4 *	0.0043 *T	2.4 *	10/80, 3/88:q1* 1.3
6/89	7 p 4-4'-DDT (c)	4.7	625	0.012	608	1.1 *	0.001 *W	1.1 *	10/80, 8/88:q1* 0.34
7/90	8 p 4,4'-DDE (c)	5.6	625	0.004	608	105 1s	10.5	105	10/80, 8/88:q1* 0.34
7/90	9 p 4,4'-DDD (c)	2.8	625	0.011	608	0.064 8s	0.0064	0.064	10/80, 8/88:q1* 0.24
4/89	10 p Dieldrin (c)	2.5	625	0.002	608	2.5 *	0.0019 *T	2.5 *	10/80, 9/88:q1* 16
4/89	11 p a-Endosulfan	--a	625	0.014	608	0.22 *	0.056 *	0.22 *	10/80, 3/88:RfD 0.00005
4/89	12 p b-Endosulfan	--b	625	0.004	608	0.22 *	0.056 *	0.22 *	10/80, 3/88:RfD 0.00005
4/89	13 p Endosulfan Sulfate	5.6	625	0.066	608	--	--	--	10/80
7/90	14 p Endrin	nr	625	0.006	608	0.18 *	0.0023 *T	0.18 *	10/80, 9/88:RfD 0.0003
7/90	15 p Endrin Aldehyde	nr	625	0.023	608	--	--	--	10/80, 9/88:RfD 0.0003
4/89	16 p Heptachlor (c)	1.9	625	0.003	608	0.52 *	0.0038 *T	0.52 *	10/80, 3/88:q1* 4.5
12/89	17 p Heptachlor Epoxide (c)	2.2	625	0.083	608	0.52 *	0.0038 *T	0.52 *	10/80, 3/88:q1* 9.1
6/89	18 p PCB-1242 (PCB, c)	nr	625	0.065	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	19 p PCB-1254 (PCB, c)	36	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	20 p PCB-1221 (PCB, c)	30	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	21 p PCB-1232 (PCB, c)	nr	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	22 p PCB-1248 (PCB, c)	nr	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7

Table 2-12. Summary of EPA Ambient Water Quality Criteria and Screening Concentrations for the Protection of Freshwater Biota (continued)

TSS790 SCREENING LIST
UPDATE: JANUARY 1991
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304(a) SCREENING VALUES AND RELATED INFORMATION
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DATE REVISED	COMPOUND	EPA DETECTION LEVEL [40 CFR 136] (µg/L)	Ref. Method	(µg/L)	Ref. Method	FRESHWATER Screening Value (Maximum) (µg/L)	Screening Value (Continuous) (µg/L)	95% LC50 Value (µg/L)	CRITERIA DATES
11/89	24 bn Diethyl Phthalate	1.9	625	0.49	606	5210 2s	521	5210	10/80, 9/87:RfD 0.8
4/89	25 bn Dimethyl Phthalate	1.6	625	0.29	606	3300 2s	330	3300	10/80
4/89	26 bn Di-n-Butyl Phthalate	2.5	625	0.36	606	94 6s	9.4	94	10/80, 1/87:RfD 0.1
4/89	27 bn 2,4-Dinitrotoluene (c)	5.7	625	0.02	609-EC	3100 2s	310	3100	10/80
4/89	28 bn 2,6-Dinitrotoluene	1.9	625	0.01	609-EC	--	--	--	10/80
4/89	29 bn Di-n-Octyl Phthalate	2.5	625	3	606	--	--	--	10/80
4/89	30 bn 1,2-Diphenylhydrazine (c)	20	1625			27 2s	2.7	27	10/80, 3/88:q1* 0.8
1/91	31 bn Fluoranthene	2.2	625	0.21	610	398 2s	39.8	398	10/80, 9/90:RfD 0.04
1/91	32 bn Fluorene	1.9	625	0.21	610	--	--	--	10/80, 9/90:RfD 0.04
4/89	33 bn Hexachlorobenzene (c, B)	1.9	625	0.05	612	--	--	--	10/80
7/89	34 bn Hexachlorobutadiene (c)	0.9	625	0.34	612	9 5s	0.93 1s	9	10/80, 3/88:q1*, 6/89:q1* 0.078
1/91	35 bn Hexachlorocyclopentadiene	nr	625	0.4	612	0.7 4s	0.07	0.7	10/80, 3/88:RfD 0.007
4/89	36 bn Hexachloroethane (c)	1.6	625	0.03	612	98 5s	9.8	98	10/80, 3/88:q1* 0.014
4/89	37 bn Indeno(1,2,3-cd)Pyrene (PAH, c)	3.7	625	0.043	610	--	--	--	10/80
1/91	38 bn Isophorone (c)	2.2	625	15.7	609-EC	11700 2s	1170	11700	10/80, 9/89:RfD 0.2, 8/90:q1* 0.0041
4/89	39 bn Naphthalene	1.6	625	1.8	610	230 4s	62 1s	230	10/80
12/89	40 bn Nitrobenzene	1.9	625	13.7	609-EC	2700 2s	270	2700	10/80, 5/88:RfD 0.0005
10/90	41 bn N-Nitrosodimethylamine (c)	nr	625	0.15	607	--	--	--	10/80, 3/88:q1* 51
1/91	42 bn N-Nitrosodi-n-Propylamine (c)	nr	625	0.46	607	--	--	--	10/80, 3/88:q1* 7.0
4/89	43 bn N-Nitrosodiphenylamine (c)	1.9	625	0.81	607	585 2s	58.5	585	10/80, 3/88:q1* 0.0049
1/91	44 bn Phenanthrene (B)	5.4	625	0.64	610	--	--	--	10/80
1/91	45 bn Pyrene	1.9	625	0.27	610	--	--	--	10/80, 9/90:RfD 0.03
11/89	46 bn 1,2,4-Trichlorobenzene	1.9	625	0.05	612	150 4s	44.9 1s	150	10/80
7/90	1 p Aldrin (c)	1.9	625	0.004	608	3 *	0.3	3 *	10/80, 12/88: q1* 17
4/89	2 p a-BHC (c)	nr	625	0.003	608	--	500 p	--	10/80, 3/88:q1* 6.3
4/89	3 p b-BHC (c)	4.2	625	0	608	--	5000 p	--	10/80, 9/87:q1* 1.8
4/89	4 p g-BHC (c)	nr	625	0	608	2 *	0.08 *	2 *	10/80
4/89	5 p d-BHC (c)	3.1	625	0.009	608	--	--	--	10/80
4/89	6 p Chlordane (c)	nr	625	0.014	608	2.4 *	0.0043 *T	2.4 *	10/80, 3/88:q1* 1.3
6/89	7 p 4-4'-DDT (c)	4.7	625	0.012	608	1.1 *	0.001 *W	1.1 *	10/80, 8/88:q1* 0.34
7/90	8 p 4,4'-DDE (c)	5.6	625	0.004	608	105 1s	10.5	105	10/80, 8/88:q1* 0.34
7/90	9 p 4,4'-DDD (c)	2.8	625	0.011	608	0.064 8s	0.0064	0.064	10/80, 8/88:q1* 0.24
4/89	10 p Dieldrin (c)	2.5	625	0.002	608	2.5 *	0.0019 *T	2.5 *	10/80, 9/88:q1* 16
4/89	11 p a-Endosulfan	--a	625	0.014	608	0.22 *	0.056 *	0.22 *	10/80, 3/88:RfD 0.00005
4/89	12 p b-Endosulfan	--b	625	0.004	608	0.22 *	0.056 *	0.22 *	10/80, 3/88:RfD 0.00005
4/89	13 p Endosulfan Sulfate	5.6	625	0.066	608	--	--	--	10/80
7/90	14 p Endrin	nr	625	0.006	608	0.18 *	0.0023 *T	0.18 *	10/80, 9/88:RfD 0.0003
7/90	15 p Endrin Aldehyde	nr	625	0.023	608	--	--	--	10/80, 9/88:RfD 0.0003
4/89	16 p Heptachlor (c)	1.9	625	0.003	608	0.52 *	0.0038 *T	0.52 *	10/80, 3/88:q1* 4.5
12/89	17 p Heptachlor Epoxide (c)	2.2	625	0.083	608	0.52 *	0.0038 *T	0.52 *	10/80, 3/88:q1* 9.1
6/89	18 p PCB-1242 (PCB, c)	nr	625	0.065	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	19 p PCB-1254 (PCB, c)	36	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	20 p PCB-1221 (PCB, c)	30	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	21 p PCB-1232 (PCB, c)	nr	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7
6/89	22 p PCB-1248 (PCB, c)	nr	625	nr	608	0.2 7s	0.014 *W	0.2	10/80, 5/89:q1* 7.7

Table 2-12. Summary of EPA Ambient Water Quality Criteria and Screening Concentrations for the Protection of Freshwater Biota (continued)

KEY

m: metal
c: carcinogen, 10-6 risk level
O: based on organoleptic data
MCL: SDWA value
W: Final Residue Value based on wildlife feeding study
T: based on marketability of fish
X: not recommended if compound known to be present in sample
nr: not reported
hrms: high resolution mass spectroscopy
HM: halomethane, human health criteria apply to total halomethanes
PAH: polynuclear aromatic hydrocarbon, human health criteria apply to total PAHs
V: volatile compounds
a: acidic compounds
EC: electron capture detector
FI: flame ionization detector
PCB: polychlorinated biphenyl criteria apply to total PCBs
TRC: measured as total residual chlorine
q1*: Cancer Potency Factor
*: criterion
III: trivalent species
VI: hexavalent species
s: number of species
lr: for long term irrigation of sensitive crops (minimum standard)
p: lowest plant value reported
BCF: bioconcentration factor = tissue concentration divided by water concentration
d: see table Ambient Water Quality Criteria for Ammonia-1984 EPA 440/5-84-001
CHCl3: Based on chloroform criteria
RfD: verified Reference Dose for Noncarcinogens
e: see table Ambient Water Quality Criteria for Ammonia (Saltwater) EPA 440/5-88-004
f/l: number of fibers per liter of water - based on consumption of water only
H: based on hardness equation
pH: based on pH equation
bn: base neutral compounds
f: freshwater organisms
e/c: estuarine/coastal organisms
oo: open ocean (marine) organisms

Hardness Equations:

COMPOUND	CMC	CCC	95% LC50
Cadmium	e(1.128(lnH)-3.828)	e(0.7852(lnH)-3.49)	2e(1.128(lnH)-3.828)
Chromium III	e(0.819(lnH)+3.688)	e(0.819(lnH)+1.561)	2e(0.819(lnH)+3.688)
Copper	e(0.9422(lnH)-1.464)	e(0.8545(lnH)-1.465)	2e(0.9422(lnH)-1.464
Lead	e(1.273(lnH)-1.46)	e(1.273(lnH)-4.705)	2e(1.273(lnH)-1.46)
Nickel	e(0.846(lnH)+3.3612)	e(0.846(lnH)+1.1645)	2e(0.846(lnH)+3.3612
Silver	e(1.72(lnH)-6.52)		e(1.72(lnH)-6.52)
Zinc	e(0.8473(lnH)+0.8604)	e(0.8473(lnH)+0.7614)	2e(0.8473(lnH)+0.8604)

screening value, also obtained from the relevant EPA ambient water quality criteria document, was used when available.

For sediments, analytical values were compared with biologically effective sediment concentrations from a recent publication of the National Oceanic and Atmospheric Administration (NOAA): "Technical Memorandum NOS OMA 52: The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program" (Long and Morgan, 1990). This memorandum compiles data from existing studies that link sediment concentrations of specific trace elements, PCBs, pesticides, and polycyclic aromatic hydrocarbons (PAHs) with predicted or observed biological effects (table 2-13). Data were screened and ordered to select concentrations that represent the lower 10th percentile of the screened available data (ER-L: Effects Range-Low), the 50th percentile of the screened available data (ER-M: Effects Range-Median), and the overall apparent effects threshold (AET).

Sediment concentrations of chemicals that exceeded the ER-L concentration were identified for each study site. The ER-L is not a particularly conservative value, since 10 percent of the studies conducted have demonstrated or predicted actual biological effects at concentrations at or below that level. No safety factor was applied to the ER-L in generating overage tables. Therefore, the ER-L may not always be adequately protective of the aquatic environment and measured concentrations that exceed the ER-L should be viewed as potential environmental hazards. (Although the NOAA document concentrates on estuarine sediments and sediments sampled for the AWIS are essentially freshwater, often the difference between effective concentrations for specific chemicals in freshwater and saltwater habitats is fairly narrow. No similar compilation and synthesis of sediment toxicity information was found or is believed to exist for freshwater systems.)

The benchmark concentration used for comparison with chemical concentrations in soil samples collected from the Superfund sites was also the ER-L. Although based on sediment rather than soil literature, the NOAA sediment benchmarks represent the only consolidated source of guidance for toxic effects in solid media at this point. Furthermore, there is often little difference in the physical characteristics of soils and sediments here: both are primarily sand, the water table is within inches of the soil surface over most of the site, and precise definition of whether a specific sample represents soil or sediment may, in many cases, be a function of recent rain events and water table level at the time of sampling. Nevertheless, the level of uncertainty associated with using the ER-L as a benchmark for the concentration of a chemical in soil will be greater than its use as a benchmark for the concentration of a chemical in sediments.

From tabulated environmental concentration/benchmark ratios it was apparent that the greatest potential for environmental effects associated with contaminants in wetland sediments was concentrated in a few specific areas. The highest ratio of sample concentration to benchmark (table 2-14) was for chlordane in the North Wetland: 194 times benchmark (194X). That same area contained levels of 4,4′-DDE (dichlorodiphenyldichloroethylene), 4,4′-DDD (tetrachlorodiphenylethane), total lead, and mercury at 60X, 16X, 8X, and 7X their benchmark values, respectively. The North Wetland area historically has received drainage from Bay Drums, Peak Oil, and Reeves Southeastern.

Table 2-13. Summary of ER-L, ER-M, and Overall Apparent Effects Thresholds Concentrations for Selected Chemicals in Sediment (Dry Weight)

Chemical Analyte	ER-L Concentration	ER-M Concentration	ER-L:ER-M Ratio	Overall Effects Apparent Threshold	Subjective Degree of Confidence in ER-L/ER-M Values
Trace Elements (ppm)					
Antimony	2	25	12.5	25	Moderate/moderate
Arsenic	33	85	2.6	50	Low/moderate
Cadmium	5	9	1.8	5	High/high
Chromium	80	145	1.8	NO	Moderate/moderate
Copper	70	390	5.6	300	High/high
Lead	35	110	3.1	300	Moderate/high
Mercury	0.15	1.3	8.7	1	Moderate/high
Nickel	30	50	1.7	NSD	Moderate/moderate
Silver	1	2.2	2.2	1.7	Moderate/moderate
Tin	NA	NA	NA	NA	NA
Zinc	120	270	2.2	260	High/high
Polychlorinated Biphenyls (ppb)					
Total PCBs	50	400	7.6	370	Moderate/moderate
DDT and Metabolites (ppb)					
DDT	1	7	7	6	Low/low
DDD	2	20	10	NSD	Moderate/low
DDE	2	15	7.5	NSD	Low/low
Total DDT	3	350	117	NO	Moderate/moderate
Other Pesticides (ppb)					
Lindane	NA	NA	NA	NA	NA
Chlordane	0.5	6	12	2	Low/low
Heptachlor	NA	NA	NA	NSD	NA
Dieldrin	0.02	8	400	NO	Low/low
Aldrin	NA	NA	NA	NSD	NA
Endrin	0.02	45	2250	NSD	Low/low
Mirex	NA	NA	NA	NSD	NA
Polynuclear Aromatic Hydrocarbons (ppb)					
Acenaphthene	150	650	4.3	150	Low/low
Anthracene	85	960	11.3	300	Low/moderate
Benzo(a)anthracene	230	1600	7	550	Low/moderate
Benzo(a)pyrene	400	2500	6.2	700	Moderate/moderate
Benzo(e)pyrene	NA	NA	NA	NSD	NA
Biphenyl	NA	NA	NA	NSD	NA
Chrysene	400	2800	7	900	Moderate/moderate
Dibenz(a,h)anthracene	60	260	4.3	100	Moderate/moderate
2,6-dimethylnaphthylene	NA	NA	NA	NSD	NA
Fluoranthene	600	3600	6	1000	High/high
Fluorene	35	640	18.3	350	Low/low
1-methylnaphthalene	NA	NA	NA	NSD	NA
2-methylnaphthalene	65	670	10.3	300	Low/moderate
1-methylphenanthrene	NA	NA	NA	NSD	NA
Napthalene	340	2100	6.2	500	Moderate/high
Perylene	NA	NA	NA	NSD	NA
Phenanthrene	225	1380	6.1	260	Moderate/moderate
Pyrene	350	2200	6.3	1000	Moderate/moderate
2,3,5-trimethylnaphthalene	NA	NA	NA	NSD	NA
Total PAH	4000	35000	8.8	22000	Low/low

ER-L = Effects Range-Low. That concentration equivalent to the lower 10 percentile of the screened available data. Indicates the low end of the range of concentrations in which effects were observed or predicted.

ER-M = Effects Range-Median. That concentration equivalent to the 50 percentile point in the screened available data.

NSD = Not Sufficient Data

NA = Not Available

Modified from: Long, E.R. and L.G. Morgan. March 1990. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Stations and Trends Program. NOAA Technical Memorandum NOS OMA 52. Office of Oceanography and Marine Assessment. National Oceanographic and Atmospheric Administration. Seattle, Washington.

Table 2-14. **Ratio of Analyte Concentration in Wetland Sediments to a Biologically Effective Concentration (U.S. EPA, 1990c)[a]**

JIS STATION ID BDRI STATION ID AWRI/FS STATION ID	01-CPW 01-RFW POND 11/28/89 1115	01-SFW 02-RFW SPRAY FIELD 11/28/89 1355	01-NOW A NORTH WETLAND 11/30/89	01-UNC B UNNAMED CREEK 11/30/89	01-CLW 5R CENTRAL WETLAND 1/4/90	02-CLW 6 CENTRAL WETLAND 11/30/89	03-CLW 7 CENTRAL WETLAND 11/30/89	04-CLW 8 CENTRAL WETLAND 11/30/89	01-SOW D SOUTH WETLAND 11/30/89	02-SOW E SOUTH WETLAND 11/28/89	03-SOW F SOUTH WETLAND 11/28/89	04-SOW G SOW WETLAND 11/28/89	02-CLW C CENTRAL WETLAND 11/30/89	01-UNC 01-SAP UNNAMED CREEK 1/9/90
INORGANIC ELEMENTS														
ANTIMONY	--	--	--	--	--	--	1.85	--	--	--	--	--	--	--
LEAD	1.22	--	7.60	2.02	--	1.86	--	--	--	--	--	--	1.09	--
MERCURY	--	--	7.33	1.46	--	1.60	--	--	--	--	--	--	--	--
ZINC	--	--	2.95	93.3	--	3.35	--	--	1.16	1.72	1.95	--	--	--
EXTRACTABLE ORGANIC COMPOUNDS														
BENZO(A)-ANTHRACENE	--	--	--	--	--	--	1.56	--	--	--	--	--	--	--
DIBENZO (A,H)-ANTHRACENE	--	--	--	--	--	--	3.33	--	--	--	--	--	--	--
ORGANOCHLORINE PESTICIDES ANALYSIS														
GAMMA CHLORDANE	--	--	194.00	--	--	--	--	--	--	--	--	--	--	--
4,4'-DDD	--	--	16.00	--	--	--	--	--	--	--	--	--	--	--
4,4'-DDE	--	--	60.00	--	--	60.00	--	--	--	--	--	--	--	--
PCB-1260	--	--	1.62	--	5.20	9.60	--	--	--	9.8	--	--	--	--

[a]This table compares sample concentration, Table 4.11, with the ER-L, Table 4.13, for each element or compound listed in both tables.

Another unnamed wetland undergoing an environmental risk is a small wet area that serves, along with the North Wetland, as the headwaters of the Unnamed Creek. This wet area receives drainage from Reeves Galvanizing, and sediments were found to contain 93X the benchmark value for zinc. Sediments here also contained very high concentrations of iron (22,600 ppm), aluminum (3,790 ppm), and other metals that contribute to the potential environmental hazard posed by this site. Because there are no benchmark values for these additional metals, they do not appear on the overage tables.

One of four Central Wetland stations exceeded sediment benchmarks for 4,4′-DDE and PCB-1260 by 60X and 10X, respectively. One of four South Wetland stations exceeded the benchmark by 10X. No other wetland stations exceeded sediment benchmarks by more than 5X.

Risks Based on Biological Observations. Field observations on the number of taxa (species diversity) of benthic macroinvertebrates were used as a method to evaluate risks associated with existing conditions. A total of 53 kinds of benthic macroinvertebrates were collected from the five wetlands. The taxa checklist reveals similar species richness (number of taxa) at all wetland sites with the exception of a station in the South Wetland. At this station, the diversity of benthic macroinvertebrates was significantly reduced to 11 taxa compared to a range of 19 to 23 taxa for the other wetlands sampled. Elevated conductivity of the surface waters coincided with the reduced numbers of taxa. The Spray Field Wetland and the Cypress Pond Wetland also shared a similar conductivity regime, possibly indicating ground-water inflow. The elevated conductivity values, however, did not appear to be a factor affecting the diversity of macroinvertebrates in the other wetlands under study; the Cypress Pond and Spray Field Wetlands featured the most diverse community of benthic macroinvertebrates sampled.

Mercury concentrations in tissues were typically lower than national mean values, but three of four samples for fish and crayfish taken from the reference wetlands exceeded criteria proposed for the protection of birds that may prey on them.

Bioassay results and macrobenthic community analysis supported the potential for risk identified for the Unnamed Creek headwater area (table 2-15). High toxicity to several test species was demonstrated and the area was essentially devoid of benthic life. The relationship among toxicity, benthic community, and benchmark excesses was not as clear for the North Wetland station.

To summarize the results in biota, waters of the North, Central, and South Wetlands showed little toxicity to the organisms tested. The sediments of each wetland were at least chronically toxic to *Daphnia*.

The water and sediments of the Unnamed Creek were moderately to highly toxic to almost all organisms tested. Of all wetland sample locations tested, this is by far the one most needing further attention as evidenced by toxicity test results, benthic community structure, bioaccumulative contaminants, and water and sediment chemistry. Use of multiple measurement endpoints provides a basis for assessing the efficacy of future remediation efforts at these sites.

Table 2-15. Ratio of Areawide Hydrologic Study Surface Water Constituents to EPA Ambient Water Quality Criteria and Screening Values (U.S. EPA, 1990c)

Sample Designation	Analyte[1]	Hardness Adjusted AWQC or SV+ (ug/L) CMC	CCC	MC[2] (ug/L)	Ratio[3] MC/CMC	MC/CCC
4W, Drainage to N Wetland[4]	Aluminum	750	87	257		3.0
	Iron	-	1000	5630		5.6
	Lead (H)	125.30	4.88	5.1		1.0
	Zinc (H)	155.63	140.96	37.9		
4W (R) Drainage to N Wetland	Methylene chloride	19300+	1930+	2.0		
9W (01-UNC), Creek N of SE Gal[5]	Aluminum	750	87	5500	7.3	63.2
	Chromium III (H)	3410.57	406.52	-		
	Chromium VI	16	11	-		1.4
	Chromium (T)	-	-	16		
	Iron	-	1000	25000		25
	Lead (H)	233.12	9.08	15		1.7
	Zinc (H)	235.26	213.08	11000	46.8	51.6
10W, Creek N of SE Gal[6]	Aluminum (H)	750	87	372		4.3
	Copper (H)	24.66	15.95	6.7B		
	Iron	-	1000	3080		3.1
	Lead (H)	127.58	4.97	5.0W		1.0
	Zinc (H)	157.50	142.66	1410	9.0	9.9

1 Includes analytes found both in AWQC listings (Table 4.28) and in surface water samples (Table 4.26).
Total chromium measured in samples; criteria calculated for trivalent and hexavalent chromium. Calculated ratios for chromium hold only if all measured chromium is present in that form.

2 Measured concentration (Tables 4.26).

3 Ratios calculated only for those measured concentrations that exceed CMC or CCC.

4 Water hardness = 140 mg/L as $CaCO_3$.

5 Water hardness = 228 mg/L as $CaCO_3$.

6 Water hardness = 142 mg/L as $CaCO_3$.

CMC = Criterion Maximum Concentration. The 1-hour average concentration not be exceeded more than once every 3 years on the average.

CCC = Criterion Continuous Concentration. The 4-day average concentration not be exceeded more than once every 3 years on the average.

(H) = CMC and CCC have been adjusted for water hardness.

(T) = Total

(R) = Resampled for volatiles.

\+ = Screening value.

B = Analyte found in associated blank as well as in sample.

W = The posot-digestion spike for furnace AA analysis is outside of the 85 to 115% control limits, while sample absorbance is less than 50% of the spike absorbance.

The sediments of the Cypress Pond Wetland were highly toxic to fish, daphnids, algae, and bacteria, and the source of this toxicity should be further explored. The contamination of reference sites is highly likely in industrial areas, indicating the need for a suite of measurement endpoints.

The apparent toxicity of the sediment does not appear to have impaired wetland functions—balanced communities of plants and animals remain.

Comments on Risk Characterization

- *Evaluating risks by ratio to benchmarks (AWQC for water, NOAA Effects Range-Low for sediments) is valid, but limited: it does not account for possible synergistic effects of exposure to multiple contaminants and does not consider site-specific conditions that could affect the bioavailability and toxicity of the chemicals.*

- *The case study presents information on risks but these are not brought together to provide an overview of risk. The information is not clearly related to the assessment of wetland integrity. If the data on contaminant concentration, toxicity assays, and bioaccumulation were put into a spatial context, that could be used to assess risk and to determine how much of the resource was at risk as a result of contamination.*

- *A useful addition to the risk characterization would be a summary of the measurement endpoints and statements about what these mean in terms of assessment endpoints.*

- *This case study offers a number of valuable lessons on the application of risk assessment. One is the necessity for multiple measurement endpoints. In situations such as this, where one is dealing with multiple and in many cases unknown stressors, it is essential to have more than one measurement endpoint. For example, had the sole endpoint in this assessment been a comparison of affected wetlands to the chosen reference wetland, this analysis would have failed because the reference was found to be contaminated. A suite of metrics also provides the assessor with one measure of uncertainty in the assessment, i.e., a clearer picture of the weight of evidence behind a particular conclusion.*

- *A second lesson in this study is the clear need for: (a) care in selecting reference sites, (b) caution in interpreting the data from those sites, and (c) the inappropriateness of relying on reference sites as the sole standard for assessing risk. It is noted that for Superfund sites, good reference sites may be hard to identify because there are frequently other industries in the area. Given the natural range of variability in ecological systems, wisdom (or statistics) would dictate using more than one reference site.*

- *A third lesson from this study is the need for: (a) better benchmarks for freshwater sediments and soils; (b) test organisms for detecting direct sediment toxicity; (c) rapid bioassessment metrics for wetland species; and (d) models useful in assessing risk at Superfund sites. In developing metrics, consideration should be given to using measures other than (or in addition to) species abundance or presence/absence data; for example, species-specific biomass information may be a more sensitive metric.*

Comments on Risk Characterization (continued)

- *Several questions that should be addressed in an ideal risk assessment at a Superfund site include: (1) What are the ecological resources in the area of the site? (2) What are the risks associated with the range of human impacts in this area (e.g., habitat alteration, chemical contaminants, hydrologic change)? (3) What are the risks caused by the site to adjacent ecological resources? What is the spatial extent of these risks?*

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SECTION THREE

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

SPECIAL REVIEW OF THE GRANULAR FORMULATIONS OF CARBOFURAN BASED ON ADVERSE EFFECTS ON BIRDS

AUTHOR AND REVIEWERS

AUTHOR

Clyde R. Houseknecht
Office of Water
U.S. Environmental Protection Agency
Washington, DC

REVIEWERS

Christian E. Grue (Lead Reviewer)
Washington Cooperative Fish
and Wildlife Research Unit
U.S. Fish and Wildlife Service
Seattle, WA

Joel Brown
Department of Biological Sciences
University of Illinois
Chicago, IL

Arthur L. Buikema
Biology Department
Virginia Polytechnic Institute and State University
Blacksburg, VA

Carolyn L. Fordham
Terra Technologies
Golden, CO

Clifford Hupp
Water Resources Division
U.S. Geological Survey
Reston, VA

Ronald J. Kendall
Institute of Wildlife and Environmental Toxicology
Clemson University
Clemson, SC

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LIST OF ACRONYMS

AI	active ingredient
EPA	U.S. Environmental Protection Agency
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
LC_{50}	lethal concentration to 50 percent of organisms tested
LD_{50}	lethal dose to 50 percent of organisms tested
OPP	Office of Pesticide Programs

ABSTRACT

The Federal Insecticide, Fungicide, and Rodenticide Act authorizes the cancellation of registration of a pesticide that produces an unreasonable risk to humans or the environment. The U.S. Environmental Protection Agency's (EPA's) Office of Pesticide Programs initiated a special review of the granular formulations of the broad-spectrum insecticide/nematicide carbofuran in light of the possible risks it may pose to birds. The special review process utilized data from laboratory toxicity studies, field studies, and reported incidents of bird kills to assess the potential for adverse impacts to avian species. Based on this information, EPA proposed to cancel registration for the use of granular carbofuran, concluding that granular carbofuran generally poses unreasonable risks to birds through direct and secondary poisoning.

3.1. RISK ASSESSMENT APPROACH

An overview of the risk assessment approach is shown in figure 3-1. The risk characterization method used in this case study is a combination of the quotient and the weight of evidence methods (U.S. EPA, 1990). This study describes the ecological risk analysis that formed the basis for the issuance of the carbofuran Position Document 2/3 (U.S. EPA, 1989). It follows the standard evaluation procedures used by the Office of Pesticide Programs (OPP) to determine the risks of pesticides to nontarget species (Urban and Cook, 1986).

A major strength of this case study is the quantity of data brought to bear on the risk characterization. Laboratory studies demonstrated that granular carbofuran is acutely toxic to birds, and the presumption of risk was confirmed by field studies and numerous reports of bird kills. The analysis did not deal with information about effects at the population or ecosystem levels, nor did it evaluate possible impacts on other organisms, such as small mammals.

The use of this case study alone as a model for the evaluation of pesticides in general may be limited, since other pesticides may differ from carbofuran in their mode of action, toxicity, environmental persistence, and the relative importance of direct and indirect effects. Critical to future pesticide studies is an assessment of contamination within habitat matrices, routes of exposure, and bioavailability.

3.2. STATUTORY AND REGULATORY BACKGROUND

A pesticide product may be sold or distributed in the United States only if it is registered or exempt from registration under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), as amended (7 U.S.C. §132 et seq.). Before a product can be registered unconditionally, it must be shown that it can be used without "unreasonable adverse effects on the environment" (FIFRA section 3[c][5]); that is, without causing "any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs, and benefits of the use of the pesticide" (FIFRA section 2[bb]). The burden of proving that a pesticide meets this standard for registration is at all times on the proponent of initial or continued registration. If at any time the Agency determines that a pesticide no longer meets this standard for registration, the Administrator may initiate proceedings to cancel or suspend the registration under FIFRA section 6.

The special review process provides a mechanism through which the Agency gathers information about pesticides that appear to pose risks of adverse effects to human health or the environment. Evidence of risk submitted to and/or gathered by the Agency must be evaluated and considered in light of benefit information. If the Agency determines that the risks appear to outweigh the benefits, the Agency can initiate action under FIFRA to cancel, suspend, and/or require modification of the terms and conditions of registration.

In 1985, the Agency determined that pesticide products containing granular carbofuran exceeded the existing risk criterion for avian toxicity set forth in 40 CFR 162.11(a)(3)(i)(B) and in 40 CFR 162.11(a)(3)(ii)(c). The Agency also determined that granular carbofuran met or exceeded the proposed risk criterion (50 *Federal Register* 12195; March 27, 1985). The proposed risk criteria are now in effect as set forth in 40 CFR 154.7.

Figure 3-1. Structure of Analysis for Carbofuran Effects on Birds

PROBLEM FORMULATION

Stressors: the granular formulation of the insecticide and nematicide carbofuran.

Ecological Components: birds (kills involving at least 30 species have been reported).

Endpoints: assessment endpoint is survival of birds that forage in agricultural areas. This was evaluated using laboratory and field measurement endpoints where lethality was documented.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
Estimates were developed on granule levels in surface and subsurface soils for various applications.	Laboratory toxicological data (LD_{50} or LC_{50}) for effects of carbofuran on birds were obtained. Field studies on bird mortalities following applications were performed.

RISK CHARACTERIZATION

Risk was evaluated using a form of the Quotient Method in which estimated exposure (granules/sq. ft.) was divided by the number of granules associated with the LD_{50} value. (The larger the number, the greater the acute risk.) An estimate was made of potential bird mortality for acreage treated with carbofuran. Field data were used to verify predictions.

3.3. CASE STUDY DESCRIPTION

3.3.1. Problem Formulation

Ecological Components. This assessment focuses on birds. Birds may be exposed by ingesting carbofuran granules as they forage for seeds or grit on or below the surface of the soil. They also may be exposed by feeding on birds or other animals that contain carbofuran granules or residues and that are moribund or have already died from carbofuran poisoning. Residues of carbofuran in birds confirm that exposure to carbofuran has occurred.

OPP evaluated whether or not birds would be present during and immediately after the application of carbofuran. This analysis was conducted for 10 of the crops for which use is registered (corn, sorghum, soybeans, rice, peanuts, tobacco, cotton, cranberries, pineseed orchards and seedlings, and sunflowers). These 10 crops account for 95 percent of the annual application of granular carbofuran. Representative bird species that were expected to be present in these crops are summarized in table 3-1. Many of these species were among those found killed by direct or secondary carbofuran poisoning.

Stressors. Carbofuran is the common name for 2,3-dihydro-2,2-dimethyl-7-benzofuranyl methylcarbamate. It is a broad-spectrum insecticide and nematicide registered for control of pests on 27 agricultural crops and for certain forest and pineseed orchard uses. Approximately 7 to 10 million lb of active ingredient (AI) of all carbofuran formulations are applied annually, with approximately 6 to 9 million lb AI of the annual usage accounted for by the granular formulations. FURADAN® is the only trade name currently used.

Carbofuran is an acute toxicant that inhibits cholinesterase and results in stimulation of the central, parasympathetic, and somatic motor systems. It is generally applied as a prophylactic treatment when seeds are planted at the beginning of the growing season.

OPP received documentation for more than 40 incidents of carbofuran-related bird kills involving nearly 30 species of birds (tables 3-2 and 3-3). The most commonly affected birds were waterfowl. However, Lapland longspurs, robins, several species of sparrows and other songbirds, marsh and shore birds, and others also were affected. The number of birds involved in any single incident ranged from 1 to more than 2,000 individuals. With two possible exceptions noted in table 3-2, these kills were attributed to use of the chemical according to label instructions.

Many of these kills resulted from secondary poisoning of avian predators and scavengers (table 3-3). The ability of granular carbofuran to cause secondary poisoning is an important factor in the cumulative avian risk. This is especially true for raptors, because they occupy an important niche in the food chain by controlling vertebrate populations. Because of the problems inherent in studying the effects of carbofuran on raptor populations (i.e., to effectively monitor secondary poisoning), OPP believes that these incidents constitute only a few of the secondary poisoning deaths caused by granular carbofuran.

Incidents of avian mortality attributable to carbofuran poisoning have occurred throughout the year (table 3-2), although most have occurred during the usual planting season of April through

Table 3-1. Representative Bird Species Likely to Be Exposed to Carbofuran (U.S. EPA, 1989)

Wading Birds

Great Blue Heron
Snowy Egret
Little Blue Heron
Cattle Egret
White Ibis
Glossy Ibis

Waterfowl

Fulvous Whistling Duck
Brant
Canada Goose[a]
–Aleutian Canada Goose
Wood Duck[a]
Green-Winged Teal[a]
American Black Duck
Mottled Duck
Mallard[a]
Northern Pintail[a]
Blue-Winged Teal[a]
Cinnamon Teal[a]
Northern Shoveler
Gadwall[a]
American Wigeon[a]
Canvasback
Ring-Necked Duck

Raptors

Black Vulture
Turkey Vulture
Mississippi Kite
Bald Eagle[a]
Northern Harrier[a]
Sharp-Shinned Hawk
Cooper's Hawk
Harris' Hawk
Red-Shouldered Hawk[a]
Broad-Winged Hawk
Swainson's Hawk
Red-Tailed Hawk[a]
Rough-Legged Hawk
Golden Eagle
American Kestrel
Apolomado Falcon
Peregrine Falcon

Rails and Allies

Black Rail
Sora
Purple Gallinule
American Coot[a]
Sandhill Crane
–Mississippi
Sandhill Crane
Whooping Crane

Shore Birds

Semipalmated Plover
Piping Plover
Killdeer[a]
Spotted Sandpiper
Semipalmated Sandpiper
Pectoral Sandpiper
Stilt Sandpiper
Laughing Gull

Game Birds

Ring-Necked Pheasant[a]
Greater Prairie-Chicken
–Attwater's Greater
Prairie-Chicken
Sharp-Tailed Grouse
Ruffed Grouse
Northern Bobwhite
California Quail
Mountain Quail
Wild Turkey
White-Winged Dove
Mourning Dove[a]
Common Ground Dove
Common Snipe
American Woodcock

Owls

Common Barn Owl
Eastern Screech Owl
Great Horned Owl
Barred Owl
Long-Eared Owl
Great Gray Owl
Short-Eared Owl[a]
Northern Saw-Whet Owl

Woodpeckers

Red-Headed Woodpecker
Red-Bellied Woodpecker
Red-Cockaded Woodpecker
Northern Flicker

Songbirds

Eastern Kingbird
Horned Lark
Blue Jay[a]
Scrub Jay
American Crow
Chihuahuan Raven
Common Raven
Tufted Titmouse
White-Breasted Nuthatch
Eastern Bluebird
American Robin[a]
Brown Thrasher
Northern Mockingbird
Loggerhead Shrike
Northern Shrike
European Starling
Northern Cardinal
Pyrrhuloxia
Rose-Breasted Grosbeak
Blue Grosbeak
Indigo Bunting
Painted Bunting
Rufous-Sided Towhee
Field Sparrow
Vesper Sparrow
Lark Sparrow
Lark Bunting
Savannah Sparrow[a]
Grasshopper Sparrow
Henslow's Sparrow
Seaside Sparrow
Song Sparrow
Lincoln's Sparrow
White-Throated Sparrow
Lapland Longspur[a]
Bobolink
Red-Winged Blackbird
Tricolored Blackbird
Eastern Meadowlark
Western Meadowlark
Yellow-Headed Blackbird
Rusty Blackbird
Brewer's Blackbird
Boat-Tailed Grackle[a]
Common Grackle[a]
Brown-Headed Cowbird
American Goldfinch

[a]Confirmed kill from carbofuran ingestion.

Table 3-2. Summary of Bird Kills Due to Poisoning by Direct Consumption of Carbofuran Granules (1973–1987) (U.S. EPA, 1989)

Site	Formulation	Occurrence	Location	Number of Birds
Corn	15G	August 1983	Maryland	200
		February 1984	Illinois	not known
		June 1986	Indiana	12
		1987	New York	3
	10G	1972	Wisconsin	11
		1973	Wisconsin	3
		May 1979	New York	10
Corn/Soybeans	15G	May & June 1983	New York	25
Potatoes	15G	June 1986	New York	20
	10G	November–December 1974	Canada	80
Rapeseed[a]	10CR[b]	May 1984	Canada	>2,000
Rice	5G	April–June 1984	California	39
		April 1985	California	95
		October 1985	California	100–135[c]
		April 1986	California	36
		October 1986	California	150[c]
		April 1987	California	4
Turnips[a]	10G	1975	Canada	1,100
		1977	Canada	50
		December 1973	Canada	50–60
		June 1986	Canada	500–1,000

[a]Carbofuran is not registered for use on this crop in the United States.
[b]10 percent corncob granule.
[c]These kills may have resulted from a misuse or from carbofuran applied earlier in the year.

Table 3-3. Summary of Bird Kills Due to Secondary Poisoning From Carbofuran Granules (1983–1986) (U.S. EPA, 1989)

Location	Date	Site	Species	Description
Utah	1983	Corn	Raven	—Two ravens contained residues up to 8.1 ppm and 38 granules —Another exhibited signs of poisoning but did not die
Maryland	1983	Corn	Northern Harrier	—Contained up to 21.8 ppm carbofuran and 23 granules —Another exhibited signs of poisoning but did not die
Iowa and Illinois	1984	Corn	Red-Shouldered Hawk	—Female found intoxicated after feeding on small mammals and birds; bird was sacrificed; gut contained 47 ug and gastrointestinal tissue 49.6 ug carbofuran
Virginia	1985	Corn	Bald Eagle	—One adult male dead at base of active nest with 59% brain acetylcholinesterase inhibition; gastrointestinal tract contained 0.64 ppm carbofuran —One dead eaglet in nest along with pigeon and grackle remains —Cornfields nearby treated with carbofuran; dead and dying pigeons and other birds found nearby
Virginia	1985	Corn	Bald Eagle	—Brain exhibited 60% acetylcholinesterase inhibition; esophagus contained 82 ppm, stomach 0.067 ppm, and duodenum 1.1 ppm carbofuran
Virginia	1986	Corn	Red-Tailed Hawk	—one found dead; contained 0.107 ppm carbofuran
California	1986	Rice	Red-Tailed Hawk	—two found dead during larger waterfowl kill incident
California	1986	Rice	Northern Harrier	—Found in incident above
California	1985	Rice	Northern Harrier	—64 ppm carbofuran recovered from crop contents, including flies, duck viscera, and maggots; 84% brain cholinesterase inhibition
New York	1985	Unknown	Red-Tailed Hawk	—Adult female found dead near nest with 0.06 ppm carbofuran in liver and remains of a starling and voles in stomach

June. In reviewing the information, OPP concluded that label-directed application of carbofuran presents a hazard to birds throughout the United States.

Endpoint Selection. OPP examined the use of granular carbofuran because of numerous kills related to its use. The assessment endpoint of concern was survival of birds that forage in agricultural fields and of birds that may prey on or scavenge other organisms that have been exposed. These were evaluated using laboratory measurements of acute toxicity as well as field studies.

Comments on Problem Formulation

General comment:

- *The present case study represents a unique situation and, as a model for future studies, may be most applicable for other granular formulations. Granular carbofuran is acutely toxic to birds, and poisoned animals either die or recover quickly. Environmental persistence is relatively short. The endpoint (mortality) is relatively easy to quantify because of the high acute toxicity of the chemical, and the habitat most frequently treated is open, freshly plowed fields. Furthermore, indirect effects (e.g., pesticide-induced reductions in food resources or habitat) were not important endpoints. Other pesticides will vary in mode of action, toxicity, environmental persistence, and the relative importance of direct and indirect effects. For these reasons, the use of this case study alone as a model for evaluations of other pesticides may be limited.*

3.3.2. Analysis: Characterization of Ecological Effects

To evaluate the acute toxicity of carbofuran to bird species, OPP reviewed several acute toxicity studies using the technical grade of the active ingredient (Tucker and Crabtree, 1970; Schafer et al., 1973; Schafer and Brunton, 1979; Hudson et al., 1984; Hill and Camaradese, 1984). These studies measured the single dose that kills 50 percent of the test organisms (LD_{50}). The results of these studies (table 3-4) indicate that carbofuran is very highly toxic to a variety of avian species but that the degree of sensitivity of bird species varies.

In addition, OPP evaluated a study (Balcomb et al., 1984a) that measured the toxicity of granules of FURADAN® 10G. The study demonstrated that the consumption of a single granule of the insecticide can be fatal to a small bird.

Data for the subacute dietary concentration that kills 50 percent of the test organisms (LC_{50}) were also evaluated (Shellenberger and Gough, 1972; Hill et al., 1975; Fink, 1974, 1976). These studies indicated that the 5-day subacute LC_{50}s range from 21 to 681 parts per million (ppm) for various species of birds. These LC_{50} values confirm that carbofuran is highly toxic to birds via the diet.

Table 3-4. Acute Oral Toxicity (LD_{50}) Values and Associated 95 Percent Confidence Intervals (CI) of Carbofuran to Birds (U.S. EPA, 1989)

Species	Age	Sex	LD_{50} (mg AI/kg of body weight)	CI
Fulvous Whistling Duck	3–4 mth	F	0.24	0.20–0.28
Mallard	36 h	NR	0.37	0.28–0.48
Mallard	7 d	NR	0.63	0.53–0.74
Mallard	30 d	NR	0.51	0.41–0.64
Mallard	6 mth	NR	0.41	0.33–0.52
Mallard	3–4 mth	F	0.40	0.32–0.50
Mallard	12 mth	F	0.51	0.41–0.64
Mallard	12 mth	M	0.48	0.38–0.60
Northern Bobwhite	3 mth	F	5.04	3.61–6.99
Northern Bobwhite	12 mth	M/F	12	7–19
Japanese Quail	0.5 mth	F	1.7	1.3–1.9
Japanese Quail	0.5 mth	M	1.9	1.7–2.1
Ring-Necked Pheasant	NR	NR	4.15	2.38–7.22
Rock Dove	NR	NR	1.33	NR
Red-Winged Blackbird	NR	M	0.42	NR
Brown-Headed Cowbird	NR	NR	1.33	NR
Common Grackle	NR	NR	1.33	NR
Starling	NR	NR	5.62	3.16–10.0
Quelea	NR	NR	0.42	NR
House Sparrow	NR	NR	1.33	NR

NR = not reported.

Six field studies were evaluated by OPP, and the results are summarized in table 3-5. These studies investigated the effects of carbofuran exposure resulting from label-directed, soil-incorporated uses of FURADAN® 10G and 15G applied as band and in-furrow applications and FURADAN® 10G applied with specialized equipment. The conditions surrounding these field studies and the hypotheses being tested varied, resulting in a lack of standardization among the studies. The studies evaluating secondary mortality were even less standardized than those evaluating direct mortality.

The field studies reported that birds were killed by direct poisoning with carbofuran. When possible, bird mortality per acre was estimated for each study. These estimates were based on (1) mortalities confirmed by residue analysis, (2) carcasses in which carbofuran was strongly implicated in the cause of death (granules in the digestive system, cholinesterase depression, no evidence of trauma), or (3) cases of probable sublethal poisoning in which a bird was judged unable to recover prior to being subjected to other sources of mortality such as predation.

The mortalities reported in these studies are probably underestimates because of problems associated with carcass search efficiency and removal of carcasses by predators and scavengers. Birds may not have been found because predators carried carcasses away from the site. Additionally, birds may have sought cover when dying and may not have been noticed.

Although the field studies were performed at different application rates with different methods of application, all resulted in granules being left unincorporated and available to birds. Also, each field study resulted in avian mortality and each had one or more factors (e.g., lack of sufficient area searched, lack of sufficiently trained personnel, failure to assess carcass removal by predators) indicating that the number of dead birds encountered in carcass searches was an underestimate of the actual impact of granular carbofuran on birds. In interpreting such studies, important considerations such as the habitat and ecology of the species exposed also must be considered.

As previously mentioned, EPA also has information on a number of carbofuran-related bird kills confirming that granular carbofuran is acutely toxic to birds. In assessing these bird kills, OPP considered only those incidents in which carbofuran was clinically diagnosed as the cause of mortality or was strongly implicated.

OPP has concluded that the number of reported bird kills underestimates the number of birds that may actually be killed from exposure to granular carbofuran. Several factors support this conclusion. First, no systematic or reliable mechanism exists for accurate monitoring and reporting of wildlife kills. OPP relies heavily on incident monitoring by states, and state efforts tend to be highly variable. Only a few states have trained and equipped personnel to respond to kill reports and to conduct the thorough investigation necessary to determine the pesticide and application rate used and whether label directions were followed. In addition, few states regularly report bird kills to EPA.

Second, even if dead birds are found, the observer may not attribute the deaths to an insecticide application. Field evidence indicates that carbofuran may cause bird mortality at least

Table 3-5. Summary of Field Studies for Granular Carbofuran (U.S. EPA, 1989)

Study	Use and Site	Formulation	Method	Rate(# AI/ acre)	Acres Searched	Mortalities	Mortalities/ acre	Estimated $LD_{50}s/ft^2$ in Treated Areas[a]		
								S	G	W
1	Corn	10G&15G[b]	Band	4	254	877	3.6	1,179	29	61
2	Corn	10G	In-furrow	1	92	10	0.1	62	1	3
3	Corn	10G	In-furrow	1	34	23	0.7	62	1	3
4[c]	Pineseed	10G	POWR-TIL							
	Florida			28.3	30	12	0.4	—	—	—
	South Carolina			25.2	30	26	0.9	59	1	3
	Mississippi			22.3	30	19	0.6	93	1	2
	Louisiana			25.2	27	39	1.4	94	2	5
5	Corn	10G&15G[b]	Band							
	Illinois			1	171	92	0.5	401	9	20
	Iowa				307	32	0.1	401	9	20
6	Corn	10G&15G[b]	In-furrow							
	Texas			1	214	58	0.3	62	1	3
7	Rice	3G	Broadcast	0.5	NR[d]	5	—	234	5	12

[a]S = songbirds, G = upland game birds, and W = waterfowl. Estimates are exclusive of turn-rows. Estimated $LD_{50}s/ft^2$ combine toxicity data from table 3-4 ($LD_{50}s$) and exposure data from table 3-6 (granules exposed/ft^2). $LD_{50}s$ (mg AI/kg) were converted to $LD_{50}s$ (number of granules) assuming 0.6 mg AI/granule and average bird body weight obtained from the literature.

[b]Because the study plots were close together, birds may have moved between plots. Researchers determined that carbofuran poisoning was the cause of death, but they could not determine whether the 10G or 15G was responsible. As the data for the 10G and 15G plots are not independent, EPA could not calculate separate mortality rates.

[c]Reported rates of incorporation efficiency were, in order, 100%, 99.5%, and 99.2%.

[d]NR = not reported. Therefore, mortality/acre could not be calculated.

60 days after it was first applied. Thus, a farmer or other observer not familiar with the site history may not attribute the death to carbofuran application. If a person does suspect that a bird may have been poisoned, the individual may not know to whom to report or may believe that there is some liability associated with reporting. Finally, problems associated with the reporting of bird kills are greater for small, more inconspicuous songbirds. Many small birds do not form large flocks, and small carcasses disappear more quickly than large ones. As a result, small dead birds are less likely to be noticed than large dead birds such as waterfowl.

Comments on Analysis: Characterization of Ecological Effects

Strengths of this case study include:

- *There is a considerable amount of field and laboratory data on the toxicity of the chemical.*

Limitations include:

- *This study does not include field data for mammals; such information would be helpful for future studies.*

- *By today's standards, the design and conduct of the field studies used to support this case study are inadequate. In the majority of these studies, there are no true control plots or determinations of carcass-search efficiency. However, in this case, the evidence for avian mortality following use of granular carbofuran is so great that deficiencies in the field studies do not affect the conclusions.*

3.3.3. Analysis: Characterization of Exposure

Birds may be exposed to carbofuran granules that are on the surface of the soil as well as below it. Balcomb (1980) has shown that the size of carbofuran granules overlaps that of grit and seeds consumed by birds. In addition, dead birds have been found with soil caked on their bills in fields treated with granular carbofuran. This suggests that they had been probing for food or grit shortly before death (I. Sunzenauer, personal communication).

Granules may be applied aerially or with ground equipment at the beginning of the growing season when corn or other crops are planted. Granules may be left on the soil surface following the use of band application methods or from incomplete incorporation following in-furrow application. Granules also may be left on the soil surface when machinery is being loaded, when planter shoes are lifted out of furrows to permit turning, and when planter shoes rise out of the soils of irregularly contoured fields.

Several investigators have confirmed that both band and in-furrow application of carbofuran or other granular pesticides using conventional commercial application equipment result in exposed granules on the soil surface (Beskid and Fink, 1981; Hummel, 1983; Dingledine, 1985). Erbach and Tollefson (1983) reported that 5.8 to 40.2 percent of granules remained unincorporated after band application. Hummel (1983) showed that in-furrow application results in approximately 99 percent incorporation.

Based on these reports, OPP conservatively estimated the number of granules that would remain exposed on the soil surface as a result of incomplete incorporation at 15 percent of the granules applied by band and 1 percent applied by in-furrow application. These percentages were used to estimate the number of exposed granules. The results are given in table 3-6.

DeWitt (1966) analyzed available avian laboratory and field studies and reported a possible relationship between the quantity of pesticide ingested by birds and the quantity of pesticide deposited per unit area. Given the finding by Balcomb et al. (1984a) that the consumption of a single granule can cause death in a small bird, such large quantities of exposed granules represent a significant risk to avian species.

Birds may be seen feeding in fields during spring planting operations, often following behind planting equipment. Some birds are probably attracted to soil invertebrates, seeds, or old crop remains, that may be brought to the surface by the planter. Birds foraging for seeds or grit may be unable to avoid ingesting granular pesticides.

Granules also may be applied aerially later in the season. These granules are not incorporated. Birds are exposed to granules falling on the field surfaces and into the leaf whorls of plants. Moreover, aerial application may contaminate more of the field edge than ground application at the beginning of the season because of the inaccuracy of the aerial placement of granules.

In addition, birds may ingest carbofuran that has been applied directly to water or to fields that are subsequently flooded. Numerous duck kills have resulted from the application of carbofuran granules to rice fields in California.

Poisoning of avian predators and scavengers may occur from ingesting food items that have been exposed to carbofuran. For example, Balcomb et al. (1984b) reported whole-body carbofuran residues ranging from 0.3 to 670 ppm in 11 of 12 samples of earthworms. These worms were found in furrows after carbofuran application to corn. Worms were washed prior to chemical analyses; thus, whole-body residues did not include additional carbofuran in granules attached to the body. Secondary poisoning incidents have involved bald eagles, red-tailed hawks, northern harriers, and other birds of prey. These species are attracted to dead and dying birds and small mammals affected by granular carbofuran.

3.3.4. Risk Characterization

Risks were evaluated, in part, by comparing estimated exposure levels of granules in surface soils with that calculated as the amount needed to kill 50 percent of the organisms (LD_{50}). DeWitt (1966) proposed the square foot as the unit area for determining risks to birds from

Table 3-6. Number of Exposed Carbofuran Granules After Band and In-Furrow Application (U.S. EPA, 1989)

Formulation	Application Rate (lb AI/a)	Application Method	Granules Exposed/ft^2
15G	3	7" band	596
	3	In-furrow	93
	1	7" band	198
	1	In-furrow	31
10G	3	7" band	837
	3	In-furrow	130
	1	7" band	279
	1	In-furrow	43

exposure to granular pesticides. The square foot is a useful dimension because (1) it is easy to visualize the number of granules in a relatively small area, and (2) most birds, even a small songbird, can readily forage over a square foot of soil surface. In keeping with DeWitt's suggestion, the following equation can be used to relate toxicity and exposure:

$$\text{Risk} = \frac{\text{Exposure}}{\text{Toxicity}} = \frac{\text{Granules/sq ft}}{\text{Granules}/LD_{50}} = \frac{LD_{50}\text{s}}{\text{ft}^2}$$

OPP estimated the number of avian $LD_{50}s/ft^2$ for each of the 10 crop uses listed in table 3-7. Table 3-7 presents the minimum and maximum values for broadcast, band, in-furrow, and turn-row areas, based on label-directed application rates. These estimates suggest that the uses of granular carbofuran pose a risk to individuals of each of the representative avian groups.

OPP estimated the potential magnitude of avian mortalities from direct carbofuran poisoning resulting from application to 9 of the 10 crops. This estimate was based on the number of acres treated per year (table 3-8) and the mortality in the field studies conducted in corn. (As stated earlier, mortality estimates are probably quite conservative.) If it is assumed that similar mortality occurs in all crops, there is a potential for several million avian deaths each year resulting from the use of granular carbofuran. Because of the difficulties involved in finding dead birds, these estimates may be low.

From an analysis of the laboratory toxicity data, estimated exposure data, field studies, and incident reports described above, OPP concluded that granular carbofuran posed a risk to birds based on its acute toxicity. OPP determined that the risks associated with the continued use of granular carbofuran outweigh possible benefits and that the granular formulations should be canceled to prevent unreasonable adverse effects to the environment. See the carbofuran Position Document 2/3 (U.S. EPA, 1989) for a comprehensive discussion of benefit analyses.

Ecological risk analyses always involve a degree of uncertainty with regard to characterization of ecological effects and exposure. For example, laboratory toxicity data are available for only a limited subset of the nearly 850 species of birds that breed in or pass through the United States. It is unlikely that the most sensitive species was tested for its susceptibility to carbofuran poisoning.

Similarly, exposure estimates were based on both the highest and lowest registered application rates for each site. This resulted in wide variations in the estimates of the number of granules exposed on the soil surface. It should be noted also that, in order to keep calculations manageable, exposure estimates were limited to granules on the soil surface. Additionally, it is assumed that birds can and do consume granules when they are available. There is little information, however, on the extent to which bird anatomy and behavior influence granule consumption.

Finally, in the LD_{50}/ft^2 calculations, it is assumed that the higher the number of $LD_{50}s/ft^2$, the greater the risk. The actual relationship between the number of available granules and the actual risk, however, is not known. These uncertainties are counterbalanced by the more than 40 actual incidents of avian mortality discussed above.

Table 3-7. Minimum and Maximum Values for Avian $LD_{50}s/Ft^2$ for 10 Crops (U.S. EPA, 1989)

	$LD_{50}s/Ft^2$			
	Songbirds		Upland Game Birds and Waterfowl	
Crop	Minimum	Maximum	Minimum	Maximum
Corn	68	2,211	6	187
Cotton	68	527	6	45
Cranberries	1,052	1,052	89	89
Peanuts	32	2,632	3	223
Pineseed	5,170	10,002	438	846
Rice	156	261	13	22
Sorghum	56	1,033	5	89
Soybeans	68	2,104	6	178
Sunflowers	68	1,368	6	116
Tobacco	1,368	3,159	114	267

Table 3-8. Annual Estimated Acreage Treated With Granular Carbofuran (U.S. EPA, 1989)

Crop	Acres Treated
Corn	4,500,000–5,500,000
Sorghum	640,000–2,040,000
Soybeans	210,000–280,000
Peanuts	136,000
Tobacco	20,000–40,000
Cotton	30,000
Cranberries	185
Pineseed orchards	400–1,250
Sunflowers	18,400–105,900

Comments on Risk Characterization

Strengths of the case study include:

- *The data upon which the risk characterization is based are substantial. Both field and laboratory data are employed. Even though there are inadequacies in the design of field studies, the evidence for avian mortality following use of granular carbofuran is so great that these inadequacies do not affect the conclusions. OPP believes that only for a few pesticides will there be an equal amount of data upon which to base a risk characterization for adverse effects on birds.*

Limitations include:

- *With respect to most other pesticides, this case study represents a rather unique situation and, as a model for future case studies, may be most applicable to other granular formulations. It may not be appropriate for other pesticide formulations, however, especially those that have long-term or sublethal effects.*

- *Although $LD_{50}s/ft^2$ appears to be a useful measure for assessing toxicity of carbofuran, it may not be appropriate for other pesticides, including other granular formulations. Determination of the amount of active ingredient of the pesticide of concern within different matrices, routes of exposure, and bioavailability is essential in future studies.*

- *Inadequacies identified in the field studies include: (a) control sites are lacking or inappropriate, as are data on carcass-search efficiencies; (b) possible synergistic effects resulting from the use of other pesticides are not examined; and (c) although the case study focuses on birds, determinations of possible impacts on other taxonomic groups, particularly small mammals, should have been incorporated.*

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SECTION FOUR

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

ECOLOGICAL EVALUATION OF A FRESHWATER STREAM AND WETLANDS NEAR AN INACTIVE COKE PRODUCTION PLANT

AUTHORS AND REVIEWERS

AUTHORS

Wayne S. Davis
Region 5
U.S. Environmental Protection Agency
Chicago, IL

John Bascietto
U.S. Department of Energy
Washington, DC

REVIEWERS

G. Allen Burton (Lead Reviewer)
Biological Sciences Department
Wright State University
Dayton, OH

Robert Brooks
School of Forest Resources
Pennsylvania State University
University Park, PA

Rick D. Cardwell
Parametrix, Inc.
Bellevue, WA

Michael J. Dover
The Cadmus Group, Inc.
Peterborough, NH

Anthony F. Maciorowski
Fish Culture and Ecology Laboratory
U.S. Fish and Wildlife Service
Kearneysville, WV

Joseph Makarewicz
Department of Biological Sciences
State University of New York
at Brockport
Brockport, NY

Thomas P. O'Connor
National Status and Trends Program
National Oceanic and Atmospheric
Administration
Rockville, MD

Jerry R. Schubel
Marine Sciences Research Center
State University of New York
at Stony Brook
Stony Brook, NY

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LIST OF ACRONYMS

ARAR	applicable or relevant and appropriate requirement
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
DO	dissolved oxygen
EPA	U.S. Environmental Protection Agency
HRS	Hazard Ranking System
NCP	National Contingency Plan
PAH	polycyclic aromatic hydrocarbon
RI/FS	Remedial Investigation/Feasibility Study
SARA	Superfund Amendments and Reauthorization Act of 1986

ABSTRACT

A small stream and wetlands in southeastern Ohio received long-term waste discharges from a coke production facility before the facility closed in the early 1980s. The area affected by these discharges was determined to be eligible for cleanup action under Superfund. A Remedial Investigation/Feasibility Study was initiated to evaluate the site's potential impacts on human health and the environment and to develop a cost-effective remedial action plan.

Coke production facilities have been associated with high discharge levels of polycyclic aromatic hydrocarbons and adverse effects on aquatic biota, particularly bottom-feeding fish. The study included an examination of the surface water and sediment chemistry for a wide array of inorganic and organic chemicals as well as the aquatic biota. Studies were conducted on fish, benthic macroinvertebrates, phytoplankton, and zooplankton communities. Bottom-feeding fish were examined for neoplasms.

Conclusions from the study were that:

- the lagoons are the most severely affected areas, being highly chemically contaminated, supporting a benthic community indicative of polluted conditions, and containing no fish life;
- nutrient enrichment and reduction in dissolved oxygen concentrations in the stream may be attributable to untreated sewage discharges into the stream;
- the fish community found in the backwater marsh (slough) and at the mouth of the stream consisted of species commonly found in the larger rivers of the area, reflecting the influence of these larger rivers;
- the fish examined did not have a significant incidence of liver neoplasia, but this may not have been a sensitive indicator; and
- an improved study plan might have provided a basis for identifying exposure-effects relationships with the data generated.

4.1. RISK ASSESSMENT APPROACH

This case study, which was originally prepared by Ballantyne et al. (1984) and IT Corporation (1990), represents a typical impact assessment (ecological reconnaissance study) and does not follow the ecological risk assessment process as defined in the framework (figure 4-1). It can, however, serve as a good example of a baseline risk assessment with some modification (as noted later in the Comments sections). Many contaminated sites are similar to this one and follow similar assessment scenarios. The section-by-section comments identify the study's deficiencies and recommend improvements. The study uses multiple measurement endpoints (chemical, physical, and biological) that have been documented elsewhere as being effective and valid and should be considered in future risk assessments.

4.2. STATUTORY AND REGULATORY BACKGROUND

Environmental assessment of a Superfund site is done in accordance with the U.S. Environmental Protection Agency's (EPA's) responsibility to protect public health and the environment at uncontrolled hazardous waste sites under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA). The regulation that enables EPA to carry out its responsibilities under CERCLA/SARA is the National Contingency Plan (NCP).

Under the NCP, EPA must evaluate a site for eligibility for certain cleanup actions under CERCLA/SARA authorities. Although it is not a risk assessment methodology, the Agency's Hazard Ranking System (HRS) is used to determine national priorities for cleanup of hazardous waste sites. The HRS scoring involves a detailed evaluation of exposure and hazardous potential relative to the known and potential contaminants at the site, the exposure pathways (air, ground water, surface water, direct contact, etc.), and the known or potential ecological component population. The process of remedy development starts with a Remedial Investigation/Feasibility Study (RI/FS), with the primary objectives of evaluating the site's potential impact on human health and the environment and developing a cost-effective remedial action plan. The NCP calls for the identification and mitigation of environmental impacts on these sites and the selection of remedial actions that are "protective of environmental organisms and ecosystems." Federal and state laws and regulations that aid in this process are potentially "applicable or relevant and appropriate requirements" (ARARs). Compliance with these laws and regulations increasingly requires that the site's ecological effects be evaluated and measures be taken to mitigate those adverse effects.

The Clean Water Act, as amended by the 1987 Water Quality Act, is another ARAR and major federal regulation that requires the maintenance and restoration of the chemical, physical, and biological integrity of the Nation's waters. Most Superfund sites potentially affect surface waters and need to be assessed for both onsite and offsite effects. Recently, EPA identified the biological integrity of surface waters as indicators of both chemical and physical stressors and as direct measurements of the aquatic life that is protected by federal and state regulations (U.S. EPA, 1991a). EPA recommends that an integrated approach be used for assessing aquatic resources utilizing chemical, biological, and physical measurement and assessment endpoints (U.S. EPA, 1988a,b; 1990a). A detailed discussion of the legal and technical requirements for environmental

Figure 4-1. Structure of Analysis for Inactive Coke Production Plant

PROBLEM FORMULATION

Stressors: inorganic and organic chemicals associated with coke production waste residuals; low dissolved oxygen levels associated with sewage discharges.

Ecological Components: benthic macroinvertebrates and zooplankton and fish of a stream and river.

Endpoints: assessment endpoint is biological integrity in surface water; measurement endpoints include structural properties of fish, benthic, and planktonic communities as well as fish histopathology.

ANALYSIS

Characterization of Exposure

Exposure was evaluated by measuring the chemical concentrations at six stream stations, two lagoon locations, and one river station.

Characterization of Ecological Effects

Effects were evaluated based on the literature and examination of community structure in affected and reference areas and by examining fish for tumors.

RISK CHARACTERIZATION

Risks were characterized by:

- comparing data on chemical concentrations to criteria,
- comparing biological communities in potentially affected and reference areas,
- examining the occurrence of tumors.

assessments at Superfund sites can be found in EPA's *Risk Assessment Guidance for Superfund: Environmental Evaluation Manual* (1989a).

4.3. CASE STUDY DESCRIPTION

4.3.1. Problem Formulation

Site Description. The site examined in this case study is an inactive coke production facility that operated for most of this century in southeastern Ohio. Products from the coking operation, which ended in the early 1980s, included crude tar, coke, light oil, and ammonia. From approximately 1920 through the late 1960s, wastewater and solid wastes generated in the coking process were discharged into wetlands east of the plant, adjacent to Ice Creek, which traverses the property. This creek is a tributary to the Ohio River, a major interstate river, and is located about 750 ft from the plant. The waste streams included process wastewater, coke and coke fines, decanter tank car sludge, boiler ash, and weak ammonia liquor.

Stressors. Information collected during the RI/FS identified contaminants within the stream, onsite lagoons, and wetlands including ammonia, benzene, cyanide, chlorides, metals, naphthalene, phenol derivatives, polycyclic aromatic hydrocarbons (PAHs), and phthalate esters. Routine parameter analyses to assist in identification of potential stressors included dissolved oxygen (DO), pH, temperature, conductivity, $CaCO_3$ hardness, and a subjective habitat review.

Methods used for the identification of stressors included chemical analyses of surface waters and sediments, biological community assessments, and histopathological examination of bottom-feeding fish. A brief summary of the specific methods used appears in appendix A.

Ecological Components. Ideally, because natural systems are composed of individual organisms, local populations, and communities, the risk to each of these components should be assessed. However, because of the large number of species present in natural systems and the complexity of the interspecies relationships, it is necessary to select representative components of the ecosystem and to develop appropriate endpoints. The Superfund program has addressed many of these issues in recent technical guidance and review documents (U.S. EPA, 1987, 1989a,b, c,d,e) and is in the process of developing supplemental guidelines to their *Environmental Evaluation Manual.*

This case study examined selected ecological components of the biological communities in the freshwater stream and associated wetlands and lagoons. These included benthic macroinvertebrates, plankton, and fish. Sessile aquatic life, such as benthic macroinvertebrates, are particularly useful indicators of local environmental effects due to their lack of mobility. Planktonic organisms are the basis of the aquatic food chains, and fish represent a high-level consumer in aquatic systems. Each of these groups served as indicators of the aquatic resources and were assessed in this case study. Fish populations (golden redhorse sucker and freshwater drum) were also used to indicate sublethal effects from sediment contaminant exposure through the incidence of liver neoplasia, although this is acknowledged to be an extreme indicator.

<u>Endpoint Selection</u>. The assessment endpoint in this study was the biological integrity in the surface waters adjacent to the site. The measurement endpoints used to evaluate the assessment endpoint included structural determinations of the fish, benthic macroinvertebrate, and planktonic communities in the surface waters, as well as the incidence of liver neoplasia and other anomalies in two fish populations. Condition factors for fish populations were also included in the analysis based upon weight and length of the individuals.

Comments on Problem Formulation

Strengths of the case study include:

- *The case study documents the history of the site and identifies the primary chemicals (stressors) of concern.*

- *The identification of the various levels of components and key references is good and often overlooked. The use of biotic metrics also is good; however, they should be briefly explained showing their many components and their relationship to the community structure and ecosystem functioning.*

- *The use of community-level endpoints as an in-field measure of condition complements the chemical-specific methods that are used to predict ecological effects. This approach is supported by EPA's recent* Policy on the Use of Biological Assessments and Criteria in the Water Quality Program *(1991a).*

Limitations include:

- *The test hypothesis is not stated.*

- *Literature information that establishes the adverse effect relationship between key ecological components (sensitivity, abundance, life history, and contaminant fate) and the contaminants of concern is not incorporated. This would involve defining the scope of the assessment and justifying why the study contaminants and components were selected (essentially this represents a preliminary risk assessment).*

- *The site description is weak. It should allow the reader to evaluate the appropriateness of the various ecological components, stressors, and endpoints. There may be other migratory species (fish and wildlife) that could visit the site area and be exposed to contaminated media.*

- *No mention is made of replicate sampling design or the possible importance of PAH photoactivation to highly toxic compounds at the site.*

Comments on Problem Formulation (continued)

•Key species (e.g., based on sensitivity, endangered status, abundance, ecosystem niche) should be identified, with a discussion of relevant information on life history, food, and habitat requirements that may affect exposure-specific stressors. This information is available from preliminary site surveys, regional experts, data bases, and the literature.

•A complete risk assessment for the site should have included waterfowl and mammal species that make use of the lagoons and areas around them. The current study presents information on only the aquatic component.

General comments:

•Many of the endpoints that were measured could have been supplemented by more contemporary analytical techniques or field collection methods (Karr et al., 1986; Klemm et al., 1990). Endpoints that were not examined include sediment toxicity and habitat effects. Quantification of habitat conditions is important to determine the ecological expectations of the area, and whole-sediment toxicity testing using a variety of species (e.g., amphipod, chironomid, crustacean) would have provided an indication of potential and actual ecological risks due to sediment contamination.

•A multimetric approach for assessing ecological community health is recommended similar to EPA's Rapid Bioassessment Protocols (Plafkin et al., 1989). This study was conducted in Ohio, a state that has established methods and standard operating procedures for assessing the biological integrity of aquatic systems; their approach and techniques should have been employed (Ohio EPA, 1989, 1990). The Ohio EPA uses ecoregion-based, multiassemblage (fish and benthos), and multimetric endpoints (Index of Biotic Integrity, Modified Index of Well-Being, Invertebrate Community Index, and Qualitative Habitat Evaluation Index). It is important to consult state biologists and scientists to determine whether special or deliberate sampling methods and interpretation of results are needed.

•Shannon's diversity index is not recommended as a primary method for assessing benthic community health due to its inconsistent relationship to ecological health (Hughs, 1978; Chadwick and Canton, 1984; Washington, 1984; Resh and Jackson, 1990; Davis and Lathrop, 1992). The use of Shannon's diversity index for assessing the benthic macroinvertebrate community in this study is misleading because of the reference to Wilhm's (1970) pollution classification, which is calculated using base 2, while those in this study are calculated using base 10.

4.3.2. Analysis: Characterization of Exposure

Sample Locations. Six sampling stations were established for chemical and biological community sampling along the linear section of Ice Creek that received discharges and runoff from the plant area (figure 4-2). Stations IC-1 and IC-2 were reference stations in the creek, upstream from the plant area. Station IC-3 was adjacent to the upper part of the plant site, and Station IC-4 was established near the east side of the slough area to observe the potential effects of an unidentified effluent discharge (possibly an untreated sewage outfall). Station IC-5 was at the lower end of the plant site near the west side of the slough area, and Station IC-6 was downstream of the plant site a few hundred feet away from the Ohio River. Two lagoon-wetland stations (LG1 and LG2) were established along lagoon transects. The mouth of Ice Creek at the Ohio River was station OR-1, just downstream from Station IC-6.

The investigation of fish neoplasia occurrences was conducted in three reaches of the creek—two reference reaches and one test reach. The upstream reference reach was located starting from Station IC-2 and proceeding about 1,000 ft downstream, about halfway to Station IC-3. The test reach was located starting just downstream of Station IC-3 covering a distance of about 750 ft downstream. The second reference reach was located in the major interstate river upstream from the confluence with the tributary extending a distance of about 1 mile.

Sample Matrix. Water sampling investigations were designed using the results of the previous phases of the remedial investigation. Table 4-1 presents the matrix of parameters that were sampled at each site. Acid and base/neutral priority pollutants, cyanide, benzene, chloride, sulfate, arsenic, and metals such as lead, mercury, and cadmium were considered potential contaminants for analysis screens (results are in appendix B). Standard physical water quality parameters measured included DO, pH, temperature, specific conductance, alkalinity (mg/L $CaCO_3$), and total hardness (mg/L $CaCO_3$). Ammonia (mg/L NH_3-N) was also measured. Sediments were analyzed for arsenic, heavy metals, and organic priority pollutants. Biological exposures were assessed through community studies of the benthic macroinvertebrates, fish, and plankton and examination of liver neoplasia in bottom-feeding fish.

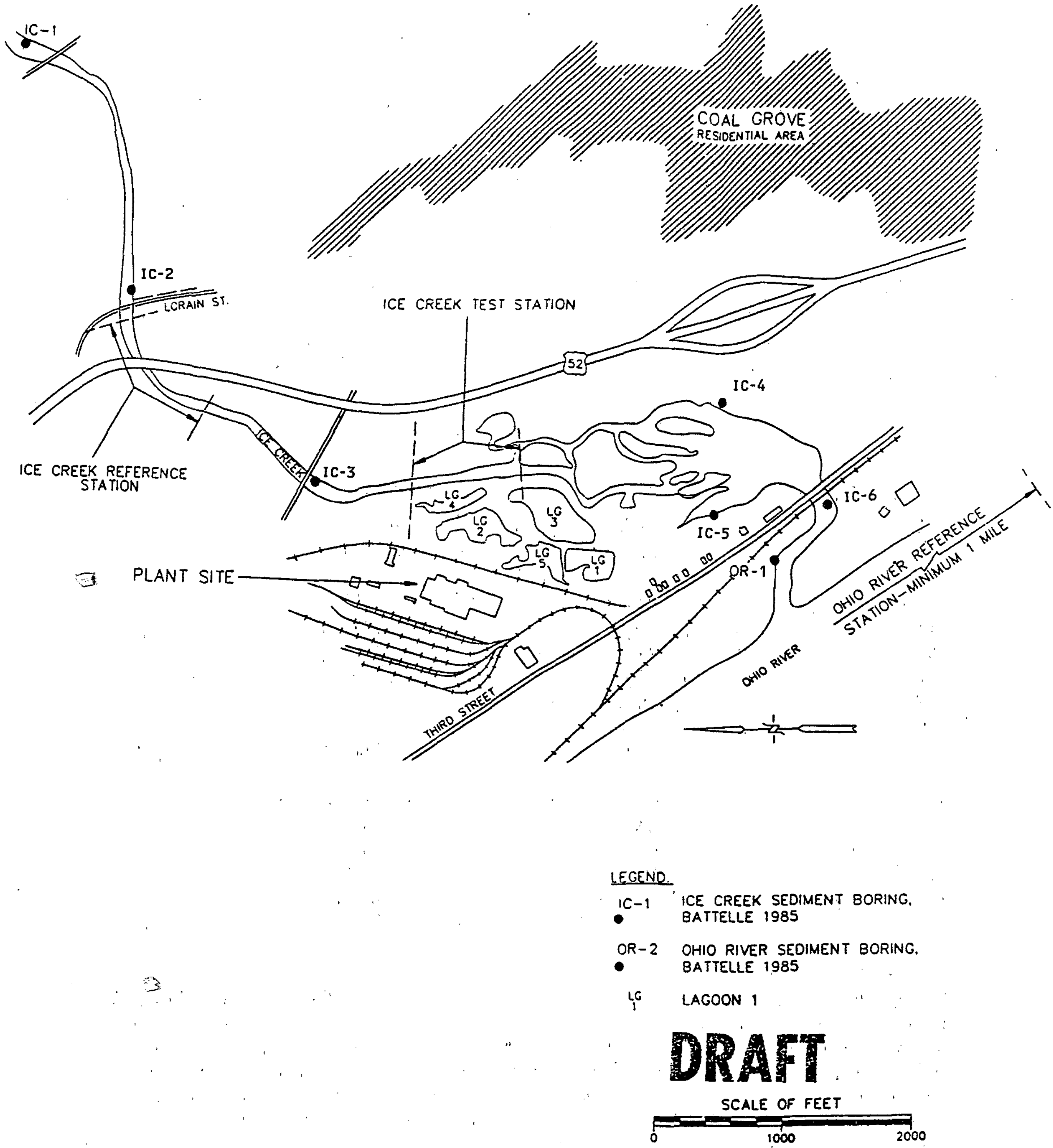

Figure 4-2. Sampling location (IT Corporation, 1990)

Table 4-1. Sample Matrix (Ballantyne et al., 1984)

Parameter	Station Number IC-1	IC-2	IC-3	IC-4	IC-5	IC-6	LG1	LG2	OR-1
Water chem.	X	X	X	X	X	X	X	X	X
Sediment chem.	X	X	X	X	X	X	X	X	X
Benthos	X	X	X	X	X	X	X	X	X
Zooplankton						X	X		
Phytoplankton						X	X		
Fish	X	X	X	X	X	X			X
Pathology		X	X						X

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *Chemicals were measured in various environmental media.*

Limitations include:

- *Little information is provided allowing the user to evaluate exposure. The hydrodynamics of surface water and ground water infiltration is not described and no information is provided on the loading of stressors. The migration of contaminants from the lagoon-depositional areas to other locations is unknown.*

- *Data on tissue residues are not reported, nor are key fish condition factors including age, size, sex, spawning status, and lipid content.*

- *An attempt should have been made to locate reference (least affected) stations in a nearby basin for comparison with the results of this study, since the upstream stations of the creek appeared to be affected. In the absence of using reference locations from a nearby watershed, more care should have been taken in selecting the stations that were sampled. The most upstream station was only a few thousand feet away from the potential influence of the facility and was subject to potential roadway influences. Station IC-3 was adjacent to the facility but could have been a few hundred feet further downstream, or another sample location should have been added just upstream of the influence of the lagoons. Stations IC-4 and IC-5 were located on either side of the wetland/backwater area adjacent to the lagoons and were not representative of the creek habitat. A better station selection would have been in the middle of the main creek channel that fed the backwater. Station IC-6 was properly located immediately downstream of the backwater in a channel connecting the creek to the Ohio River. The river station was not located in the river itself, but was a little further downstream from Station IC-6 in the connecting channel. Most of the stations, with the possible exception of the three upstream stations, are directly influenced by flow and depth changes in the river during dry weather. In wet weather events, it is likely that the creek upstream of the facility receives some contaminant contribution. Ohio River stations directly upstream and downstream from the confluence with the creek should have been included. The transitory nature of fish is not recognized, nor are habitat effects on biotic indices.*

- *Method detection limits are relatively high for most of the compounds, especially mercury, and this makes it difficult to conduct a definitive assessment.*

4.3.3. Analysis: Characterization of Ecological Effects

Ecological effects were characterized by identifying state water quality standards, examining biological community structure, and studying the histopathology of fish populations.

Water Quality Standards. Water quality standards are adopted by states and are legal requirements for establishing the desired condition of a water body. Water quality standards are ARARs that must be addressed at all Superfund sites. State standards consist of three parts: (1) the beneficial use designated for that water body, (2) numerical or descriptive criteria that measure specific conditions of the water body designed to protect the beneficial use, and (3) an antidegradation statement to ensure that high-quality waters are not arbitrarily lowered to meet the standards. The predominant criteria used to measure the condition of the water body have been the numerical chemical criteria that are based on considerations of the magnitude, duration, and frequency values for protection against both acute and chronic toxicity. Recently, direct measures of the biological community structure and function have been shown to improve dramatically EPA's ability to assess the attainment of a water body's use. As a result, EPA is requiring the states to develop biological criteria and adopt them into their water quality standards by September 30, 1993 (U.S. EPA, 1990b; 1991b). Several states currently use biological criteria as a regulatory tool, either alone or in combination with other ecological parameters (U.S. EPA, 1990b).

Biological Community Structure. Community structure can be measured at specific locations in areas suspected of contamination and at reference locations either in the same body of water in areas thought to be unaffected by contamination or in a nearby body of water that is similar in characteristics. Selection of reference locations is critical in providing the desired, or "least affected" condition, with which the community structure of the test location will be compared.

Fish Histopathology. The morphological condition of the fish community can provide an important indication of the biological integrity of a body of water. The incidence of tumors, lesions, and other anomalies is generally recorded for the entire fish community. Histopathological analyses, however, are usually limited to a few indicator species due to the time involved for a complete analysis. As with the biological community structure, comparisons of tumors, lesions, and anomalies are made between the affected locations and the reference locations. It is as critical to choose a proper reference site as it is to make a proper selection of the indicator species subjected to the histopathological evaluations. The results of the biological studies are presented in appendix C.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *A variety of good data are generated, namely water and sediment chemistry and benthic macroinvertebrate and fish community data.*

Limitations include:

- *No quantification of the relationship between exposure and the probability of adverse ecological response exists. Biological, chemical, and physical components all vary and should be acknowledged. Biotic and chemical criteria address some of the varying exposure issues. The biological data could have been analyzed to a much greater extent to define relationships. The data could be compared to ecoregion biocriteria reference stations such as those of the Ohio EPA that are incorporated into state water quality standards (Ohio Administrative Code 3745-1, adopted in February 1990, effective May 1990).*

- *Water column and sediment toxicity testing of multiple trophic levels would significantly improve the characterization of ecological effects and could address gradient (concentration) effects via spatial sampling and/or sample dilution. Toxicity testing might be considered only if sites are predicted to be toxic based on elevated stressor levels or "Equilibrium Partitioning." Key indicator, endangered, target, or important ecosystem species (ecological components) could be monitored based on community indices, acute-chronic effects, and/or tissue residues.*

- *The selection of reference stations for the histopathological assessment is not appropriate. The upstream reference station chosen was within the influence of the facility. Also, no consideration is given to the transitory nature of the fish and their relationship with the major river. Because there were no natural or artificial barriers to fish movement, it is likely that the fish collected in the reference area also inhabited the test location at various times.*

- *The use of liver neoplasia incidence in freshwater drum and golden redhorse sucker as an indicator of aquatic community effects from contaminated sediments may not be a sufficiently sensitive indicator because the incidence is related to gross pollution by PAHs; a more sensitive indicator is needed for an appropriate ecological assessment. There is also no demonstrated susceptibility of the chosen populations for neoplasia development due to chemical stressors.*

4.3.4. Risk Characterization

Risks Based on Comparisons to Criteria. The chemistry data (appendix B) showed no measured excesses of the aquatic life standards, with the exception of ammonia-nitrogen, even when compared with chronic water quality standards that are more stringent than acute standards. Ammonia and specific conductances were an order of magnitude higher in the lagoons than at any of the stations. Ammonia exceeded aquatic criteria at Stations LG1, LG2, IC-3, IC-4, IC-6, and OR-1. The potential untreated sewage outfall at Station IC-6 could have contributed to the increased contaminant levels at the downstream stations.

Priority organic pollutants were not detected in surface waters. Acid and base/neutral priority pollutants were sampled in the sediments but were not found. PAHs were elevated in the sediments in both lagoons and Station IC-6, which received direct discharge from the lagoons. However, the PAHs do not appear to be mobilized from the sediments or adversely affecting the aquatic community, as evidenced by the absence of fish liver neoplasia. No established sediment quality criteria were identified to compare with the sediment chemistry; therefore, the environmental significance of the values of PAHs in the sediments from the stream and lagoons could not be directly assessed from the exposure information. Toxicity testing was not conducted with the sediments or surface waters.

Risks Based on Biological Community Surveys. The results of the biological surveys are summarized in appendix C. Benthic macroinvertebrate structure was measured by the number of individuals, the number of taxa, and the Shannon diversity indices (table 4-C1), which were all very low and indicative of polluted conditions (Wilhm, 1970). The dominant species were tubificid worms, which can tolerate low levels of DO and high organic enrichment. The highest tubificid concentrations occurred at Station IC-4, downstream of the untreated sewage outfall. Organisms with higher DO requirements were found in the shoreline qualitative samples. Stream habitats at the reference stations were not comparable to the test stations, due mainly to a sandy substrate upstream and much siltier sediments downstream. Statistical tests such as Kruskal-Wallis test (Hollander and Wolfe, 1973) and Shannon's diversity index were calculated and showed no significant differences in species diversity among the reference and test stations.

Twenty-seven fish species and one amphibian were collected from the stream (table 4-C2). The stations averaged 13 species with a range of 10 to 16 species. The proximity of Stations IC-6 and OR-1 resulted in the reach between these two stations being sampled as one location, and the data were combined. Emerald shiners, bluegill, and gizzard shad were the most numerous species, but no significant differences occurred in the number of species found at any of the test stations compared with the reference stations. Downstream stations generally produced larger specimens, perhaps reflecting the use of these reaches by fish that are characteristic of the large river. Upstream samples had greater numbers of first year class shad, minnow, and shiners. Condition factors were calculated for sport fish and dominant species (table 4-C3). Mean values were comparable to the expected ranges of weight and length for the area (Carlander, 1969; Bennett, 1970). Although no fish were observed in the lagoons, many turtles and waterfowl were observed there. Overall, the fish community appeared to be more diverse than the macroinvertebrate community. The fish appeared to be robust and relatively free from disease. The backwater areas of the stream may serve as refuge for fish from the larger river.

The dominant phytoplankton species in the lagoons was the blue-green alga, *Anabaena*, and both lagoons supported limited zooplankton communities (table 4-C4). The density of phytoplankton cells in Lagoon 1 was almost 80 times greater than in Lagoon 2. Conversely, the density of zooplankton was about three times greater in Lagoon 2 than in Lagoon 1. The lagoons collect surface water runoff from the site, as well as ground water percolating through the surrounding area, and contain waters that are in direct contact with highly contaminated sediments. This factor likely accounts for the limited phytoplankton and zooplankton communities, and probably accounts for the total lack of fish life. It is reasonable to conclude that the lagoons could contain fish because of their connection to the stream during flooding.

Risks Based on Fish Histopathology. The examination of fish liver neoplasia was designed to determine whether there was an adverse effect on the aquatic fauna due to contaminated creek sediments. No neoplastic lesions were observed in the liver of golden redhorse sucker or freshwater drum from any of the reference or test station populations. Fin rot in freshwater drum was 44 percent in both creek stations and 70 percent in the major river. For the golden redhorse, the incidence of fin rot was 51 percent in the creek test area and 19 percent in the test reference area.

Conclusions. Based on the three methods used to evaluate conditions, the principal conclusion drawn from this study is that the Superfund site and the creek sediments do not have an adverse effect on the aquatic fauna.

Comments on Risk Characterization

Strengths of the case study include:

- *The use of biological community assessment and population measures greatly complements the chemical-specific results. Such an integrated approach to evaluating the quality of the water resources is strongly advocated. EPA has prepared a number of guidance documents and critical review documents for use by EPA programs to facilitate the utilization and application of ecological assessments, including many documents specifically directed toward the Superfund process (U.S. EPA, 1987, 1989a,b,c,d,e, 1990a). EPA also supports regional efforts, such as the annual Midwest Pollution Control Biologists Meeting in Chicago, to report on new and promising ecological assessment methods (U.S. EPA, 1988b, 1989f, 1990c). These documents, as well as others cited in this case study, should be obtained and carefully reviewed for application to Superfund ecological assessments.*

Limitations include:

- *This study was an impact assessment that did not consider the probability of effects or uncertainty.*

Comments on Risk Characterization (continued)

- *A better selection of endpoints and sample locations was needed before the study was initiated for a more definitive exposure assessment and subsequent risk characterization. In this case study, a conclusion was reached that the contaminated sediments did not adversely affect the aquatic life in the creek based on a study of liver neoplasia from two nonindigenous species of fish. This is a nonsensitive measure that may require high contaminant levels to produce an effect. More sensitive measures to characterize the risk could have included whole-sediment toxicity testing coupled with a more extensive and frequent chemical analysis of the sediment and a more rigorous analysis of the benthic community by using more useful biological metrics and by using artificial substrates.*

- *Overall, the conclusion that no community impacts are associated with the Superfund site or creek sediments is not adequately substantiated.*

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APPENDIX A

SUMMARY OF METHODS USED FOR IDENTIFICATION OF STRESSORS

Summary of Methods Used for Identification of Stressors

Biological Community. Benthic macroinvertebrates were sampled in the lagoons, stream, and river and were assessed primarily by calculating Shannon-Weiner's diversity index for the benthic macroinvertebrates (Ballantyne et al., 1984) and enumerating the organisms. Samples were taken from representative habitat types at each station with petite ponar grab samplers.

Fish populations were sampled in the stream and river by seining with a 6 by 8-foot, 1/4-inch mesh seine for 30 minutes per sampling effort. The community was assessed by calculating Carlander's condition index for fish (Carlander, 1969). The lagoons showed no evidence of use by fish life. The histopathological analysis was conducted by determining the percentage of incidences of liver neoplasia in golden redhorse suckers and freshwater drum.

Phytoplankton and zooplankton were sampled in the lagoons. The number of plankton cells per liter of water sample for the phytoplankton and zooplankton were counted. The number of taxa and individual abundances were also listed for each of the biological communities sampled. Plants and animals were usually identified to genus.

Fish Neoplasia Occurrences. The occurrence of tumors in bottom-feeding fish was used as an indicator of effects related to exposure to PAH. Freshwater drum and golden redhorse sucker were examined; the brown bullhead—known to be susceptible to PAH-induced neoplasia—would have been used but these animals were not found to occur in the study area. All of the fish collected during the study were examined for external tumors, fin rot, hemorrhaging, and parasites. Only target fish specimens (freshwater drum and golden redhorse sucker) not showing any observable capture-related damage were retained for histopathological examination, so external injuries related to capture would not be confused with environmental effects.

Chemical Analysis. Surface water samples were collected mid-channel into 1-L Nalgene bottles for metal analysis and were adjusted to pH 2–3. Samples for cyanide were adjusted to pH 11–12. Surface waters analyzed for organic priority pollutants were collected in 2-L glass jars with Teflon-lined lids and kept at 4°C until analyses were conducted. Surface water samples collected for classical parameters were analyzed the same day.

Sediments were collected along a representative transect of the station using a petite ponar grab sampler. Composites of the grabs for the transects were made by manual mixing. The samples were stored at 4°C in 1-L Nalgene bottles for metals and in 1-L glass jars with Teflon-lined lids for organic priority pollutants. Analyses for metals and priority organic pollutants were conducted using EPA-approved methods.

APPENDIX B

COMPOUNDS AND PARAMETERS SAMPLED IN SURFACE WATERS AND SEDIMENTS

Table 4-B1. Surface Waters: Toxic Pollutants (Ballantyne et al., 1984)[a]

Station	Zinc	Cadmium	Lead	Arsenic	Selenium	Mercury	Benzene	Cyanide
LG1	75	<0.8	6.9	5.4	6.0	<1.0	<0.1	<100.0
LG2	77	1.8	7.5	5.0	0.6	<1.0	<0.1	<100.0
IC-1	48	1.3	11.4	<2.0	<0.5	<1.0	<0.1	<100.0
IC-2	26	0.8	9.4	<2.0	<0.5	<1.0	<0.1	<100.0
IC-3	22	<0.8	11.6	<2.0	<0.5	<1.0	<0.1	<100.0
IC-4	87	<0.8	8.0	3.1	<0.5	<1.0	<0.1	<100.0
IC-5	56	<0.8	6.7	<2.0	<0.5	<1.0	<0.1	<100.0
IC-6	34	<0.8	4.5	4.0	<0.5	<1.0	<0.1	<100.0
OR-1	36	<0.8	6.9	3.6	<0.5	<1.0	<0.1	<100.0
Standard[b]	>99	0.8-3.1	30.0	190.0	24.0	0.2	---	8.1

[a]All units are $\mu g/L$.
[b]State water quality standards based upon a 30-day average for worst-case comparisons.
Note: A "<" sign indicates the method detection limits.

Table 4-B2. Surface Waters: Classical Parameters (Ballantyne et al., 1984)

Station	DO (mg/L)	pH	Temp. (°C)	Conduct. (μmhos/cm)	Hardness (mg/L $CaCO_3$)	Total Ammonia (mg/L NH_3-N)	
LG1	9.1	8.6	20.3	1193	544	12.5	(0.4)[a]
LG2	9.3	8.3	21.0	741	336	0.86	(0.6)
IC-1	8.8	7.3	19.0	548	252	0.003	(3.5)
IC-2	8.9	7.6	19.1	562	224	0.04	(2.4)
IC-3	6.7	7.3	19.6	650	288	0.101	(3.3)
IC-4	3.6	7.4	18.9	618	236	3.10	(3.1)
IC-5	9.8	7.3	25.4	502	180	1.6	(2.3)
IC-6	9.4	7.0	25.2	438	168	6.4	(2.9)
OR-1	8.8	7.2	25.6	411	124	5.5	(2.6)

[a]Approximate state standard at pH and temperature of sample, 30-day average for worst-case situations.

Table 4-B3. Sediments: Polycyclic Aromatic Hydrocarbons (Ballantyne et al., 1984)

Parameter[a]	Station Numbers								
	OR-1	LG1	LG2	IC-1	IC-2	IC-3	IC-4	IC-5	IC-6
Acenaphthene	ND[b]	4.0	ND	ND	ND	ND	ND	ND	15.4
Anthracene	ND	6.0	21.1	ND	ND	ND	ND	ND	9.3
Benzo(a)pyrene	4.02	9.9	90.8	ND	ND	ND	ND	ND	49.9
Chrysene	6.0	13.1	90.3	ND	ND	ND	ND	ND	44.2
Fluoranthene	8.75	31.2	140	0.56	ND	ND	ND	3.7	57.1
Fluorene	ND	11.2	ND	ND	ND	ND	ND	ND	9.7
Naphthalene	ND	52.6	44.2	ND	ND	ND	ND	ND	92.5
Phenanthrene	5.65	22.1	83.4	ND	ND	ND	ND	ND	38.7
Pyrene	7.01	25.1	100.0	0.44	ND	ND	ND	ND	43.5

[a]All units are mg/kg.
[b]ND = not detected.

APPENDIX C

RESULTS OF BIOLOGICAL SURVEYS

Table 4-C1. Benthic Macroinvertebrates Collected in Petite Ponar Grab Samples (Ballantyne et al., 1984)

	Station								
Taxon	IC-1	IC-2	IC-3	IC-4	IC-5	IC-6	OR-1	LG1	LG2
DIPTERA									
Chironomidae									
Chironomus			14	45	7	1	1		
Crytochironomus	1	1				1	4		
Polypedilum	1								
Tanytarsus	1								
Tanypus				8	4	3	3	50	
Parachironomus								2	
Psectrotanypus			3	1	12	2	16	2	
Procladius			6		1		3		
Chaoboridae									
Chaoborus							1		
EPHEMEROPTERA									
Baetisca	4								
Caenis	1		1						
Hexagenia		1	1						
ODONATA									
Progomphus	4								
Argia					1				
Plathemis							2		4
COLEOPTERA									
Elmidae									
Dubiraphia			1				1		
Dytiscidae									
Laccophilus									11
OLIGOCHAETE									
Tubifex			31	238	66	71	176		
Branchiura			5	59	30	3	28		
Lumbriculus							2		
Dero							1		
HIRUDINEA									
Illindodella					1				
AMPHIPODA									
Gammarus						1	11		
GASTROPODA									
Physa								1	
BIVALVIA									
Corbicula			1			1	2		
Sphaerium			4			1	1		
Number of Individuals	12	2	67	351	122	84	252	55	15
Number of Taxa	6	2	9	5	9	6	15	4	2
Shannon's H	0.68	0.3	0.69	0.41	0.58	0.34	0.5	0.1	0.25

Table 4-C2. Numbers and Species of Fishes Collected (Ballantyne et al., 1984)

	Station					
	IC-1	IC-2	IC-3	IC-4	IC-5	OR-1
Cyprinidae						
Carp (*Cyprinus carpio*)					2	4
Silverjaw minnow (*Ericymba buccata*)	26					
Emerald shiner (*Notropis antherinoides*)	30	62	290	23	16	45
Steelcolor shiner (*Notropis whipplei*)	1					
Bluntnose minnow (*Pimephales notatus*)	27	38	25		6	18
Percidae						
Fantail darter (*Etheostoma flabellare*)	2					
Johnny darter (*Etheostoma nigrum*)	11	1				
Yellow perch (*Perca flavescens*)						
Centrarchidae						
Northern rock bass (*Ambloplites rupestris*)		2				
Warmouth sunfish (*Chaenobryttus gulosus*)	1		3	1	4	7
Green sunfish (*Lepomis cyanellus*)	4	34	12	1	10	1
Bluegill (*Lepomis macrochirus*)	52	13	31	45	110	112
Longear sunfish (*Lepomis megalotis*)	6	15	10	7	15	24
Longear sunfish x Bluegill hybrid	1			5	26	10
Longear sunfish x Green sunfish hybrid	1	1	1	2	34	24
Green sunfish x Bluegill hybrid				1	2	3
Northern largemouth black bass (*Micropteus salmoides*)	6	7	7	13	34	9
White crappie (*Pomoxis annularis*)					1	4
Black crappie (*Pomoxis nigromaculatus*)					1	
Pumpkinseed sunfish (*Lepomis gibbosus*)		9			2	
Catostomidae						
Quillback carpsucker (*Carpiodes cyprinus*)				21	2	
Creek chubsucker (*Erimyzon oblongus*)		11				
Clupeidae						
Gizzard shad (*Dorosoma cepedianum*)		115	15	80	100	31
Skipjack herring (*Pomolobus chrysochloris*)				2		
Sciaenidae						
Freshwater drum (*Aplodinotus chrysochloris*)				7		
Ictaluridae						
Channel catfish (*Ictalurus punctatus*)						1
Flathead catfish (*Pilodictis olivaris*)				1		
Amphibia						
Central mudpuppy (*Necturus maculosus*)			1			
Total Number of Species	13	12	10	15	16	14
Total Number of Individuals	168	308	395	209	365	293

Table 4-C3. Condition Factors Calculated for Fish (Carlander, 1969)

Station	Species	Condition Factor	Normal Range	Range
IC-2	Largemouth bass	5.2	---	4.6-5.5
IC-3	Bluegill	5.8	5.6-6.0	7.1-8.0
IC-5	Bluegill	7.1	6.4-8.2	7.1-8.0
	Crappie	5.0	---	4.6-5.5
	Carp	1.5	1.49-1.51	1.2-2.9
IC-6	Bluegill	7.0	6.0-7.7	7.1-8.0
	Largemouth bass	5.0	---	4.6-5.5
	Flathead catfish	1.0	---	0.9-1.1
	Gizzard shad	1.6	1.2-2.0	0.9-2.2
OR-1	Bluegill	7.7	5.7-11.0	7.1-8.0
	Largemouth bass	5.4	4.7-6.0	4.6-5.5
	Channel catfish	0.98	---	0.8-1.2

Table 4-C4. Plankton Collected in Lagoons 3 and 4 (Ballantyne et al., 1984)

Phytoplankton[a]	Station LG1	Station LG2	Zooplankton[a]	Station LG1	Station LG2
Chlorophyta			Copepoda		
Chlorogonium			*Cyclops*	52	34
Rhizoclonium	1,460	---	Rotifera		
Ulothrix	2,230	1,670	*Pleurotrocha*	---	170
			Polyartha	---	32
Cyanophyta			*Keratella*	---	4
Anabena	3.7×10^6	3.12×10^4	*Asplanchna*	6	---
Microchaete	---	416	Ciliophora		
Nostoc	208	6,250	*Bursaria*	46	---
Spirulina	3,300	---			
Chrysophyta					
Dipolneis	208	416			
Fragilaria	208	625			
Gomphonema	625	1,040			
Melosira	---	830			
Oscillatoria	2,910	1,250			
Euglenophyta					
Euglena	208	416			

[a]All units are cells/L.

SECTION FIVE

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

COMMENCEMENT BAY TIDELANDS ASSESSMENT

AUTHORS AND REVIEWERS

AUTHORS

Patricia A. Cirone
Region 10
U.S. Environmental Protection Agency
Seattle, WA

Robert A. Pastorak
PTI Environmental Services
Bellevue, WA

REVIEWERS

Donald P. Weston (Lead Reviewer)
Horn Point Environmental Laboratory
University of Maryland
Cambridge, MD

James J. Anderson
School of Fisheries Research
University of Washington
Seattle, WA

Jay S. Jacobson
Boyce Thompson Institute
Cornell University
Ithaca, NY

Larry Kapustka
Ecological Planning and Toxicology, Inc.
Corvallis, OR

Wayne G. Landis
Institute of Environmental Toxicology
and Chemistry
Western Washington University
Bellingham, WA

Thomas Sibley
School of Fisheries Research
University of Washington
Seattle, WA

Frieda B. Taub
School of Fisheries Research
University of Washington
Seattle, WA

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LIST OF ACRONYMS

AET	Apparent Effects Threshold
CERCLA	Comprehensive Environmental Response, Conservation, and Liability Act of 1980
EP	Equilibrium Partitioning
EPA	U.S. Environmental Protection Agency
MLLW	mean lower low water
NCP	National Contingency Plan
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
SARA	Superfund Amendments and Reauthorization Act of 1986

ABSTRACT

During an ecological assessment for the Commencement Bay, Washington, nearshore/tideflats area, field studies were designed to document the extent of sediment contamination and adverse biological effects, including sediment toxicity, alterations of benthic macroinvertebrate assemblages, chemical residues in tissues of crab and English sole, and liver lesions in English sole. During the Superfund remedial investigation for the site, indices of biological effects and sediment quality values were used to identify and prioritize problem areas for possible source control and/or sediment remedial action. The multi-indicator approach, based on chemical and biological variables, provides a powerful weight of evidence for ecological assessment. In the absence of biological data, sediment quality values may be used to interpret historical sediment chemistry data to predict the occurrence of adverse biological effects. The approach could be improved by defining assessment endpoints more explicitly, validating sediment quality values, and incorporating a probabilistic approach to exposure and risk assessment.

5.1. RISK ASSESSMENT APPROACH

The Commencement Bay (Puget Sound, Washington) case study combines a retrospective assessment (i.e., determination of effects that have already occurred) based on indices of biological effects with a predictive assessment of potential biological effects based on site-specific sediment quality criteria (figure 5-1). The study approach was based on three premises: (1) site-specific field data were needed to establish cleanup goals, (2) no single chemical or biological indicator could be used to define areas at risk, and (3) adverse biological effects were linked to sediment contamination and chemical-biological relationships could be characterized empirically.

The primary approach to identifying and ranking problem areas in the Commencement Bay ecosystem relied on direct measurements of sediment chemistry, sediment toxicity (i.e., bioassays), benthic macroinvertebrate abundances, concentrations of contaminants in English sole and crab, and prevalence of liver lesions in English sole. The amphipod mortality bioassay and oyster larvae developmental abnormality bioassay were used to characterize sediment toxicity during the remedial investigation. Microtox bioassays of sediments were performed to provide ancillary data, but Microtox response was not one of the indices of biological effects used in the remedial investigation.

Because recent historical data on sediment contamination were available at additional stations throughout the study area where biological data had not been collected, the Apparent Effects Threshold (AET) approach was developed to predict the presence or absence of specific biological effects based on chemical data alone. The derivation of the AET is explained in section 5.3.2., Analysis: Characterization of Ecological Effects. However, these empirical thresholds for chemical concentrations associated with biological effects do not establish cause and effect relationships. Exceedances of AET values were used to identify problem sediments and to identify and rank problem chemicals during source evaluation.

5.2. STATUTORY AND REGULATORY BACKGROUND

Concerns about the potential ecological and human health effects of hazardous substances in sediments of the nearshore area of Commencement Bay led to the addition of the Commencement Bay nearshore/tideflats area to the National Priorities List of Comprehensive Environmental Response, Conservation, and Liability Act (CERCLA) sites on September 8, 1983. CERCLA, as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986, requires the U.S. Environmental Protection Agency (EPA) to ensure the environment is protected when (1) remedial alternatives are selected and (2) the degree of cleanup needed is assessed. As mandated under CERCLA and the National Contingency Plan (NCP), a Remedial Investigation and Feasibility Study is required to define risks to public health and the environment. The major focus of the remedial investigation for the Commencement Bay Superfund site was characterizing impacts to aquatic organisms of exposure to contaminated marine sediments. The feasibility study screened remediation alternatives for their effectiveness in reducing risks. In the absence of regulatory standards or guidelines for establishing cleanup criteria for contaminated sediments, a decision-making approach based on chemical and biological indicators and sediment quality objectives was developed specifically for the Commencement Bay nearshore/tideflats investigations.

Figure 5-1. Structure of Analysis for Commencement Bay, Washington

PROBLEM FORMULATION

Stressors: complex mixture of organic and inorganic chemicals in marine sediments.

Ecological Components: benthic macroinvertebrates and fish.

Endpoints: assessment endpoints are the health and condition of selected ecological components. Multiple measurement endpoints were used at different levels of biological organization.

ANALYSIS

Characterization of Exposure

Exposure was evaluated by measuring concentrations of chemicals in sediments. A model was used to predict natural recovery.

Characterization of Ecological Effects

Effects were evaluated by examining benthic abundance, occurrence of liver abnormalities in fish, and various measures of sediment toxicity.

RISK CHARACTERIZATION

Risks were estimated using two basic methods:

1. comparison of conditions at contaminated sites to benchmarks or reference locations.

2. application of apparent effects threshold (AET) values for chemical concentrations in sediments.

Study areas were ranked with regard to potential risk or impact.

In 1989, a Record of Decision was signed that presented the remediation actions for the Commencement Bay nearshore/tideflats Superfund site. The site assessment in the Remedial Investigation and Feasibility Study concluded that actual or threatened releases of hazardous substances from this site, if not corrected by response actions, present an imminent and substantial endangerment to public health, welfare, or the environment.

5.3. CASE STUDY DESCRIPTION

5.3.1. Problem Formulation

Site Description. The study area is described in U.S. EPA (1985). The Commencement Bay nearshore/tideflats study site is located in a heavily industrialized area at the southern end of the main basin of Puget Sound (figure 5-2). The tideflats area, formed by the Puyallup River delta, comprises seven waterways, associated shoreline, and waters of depths less than 60 feet below mean lower low water (MLLW). Various industrial and municipal sources are located on filled areas of the tideflats, including a pulp mill, petroleum refineries and storage facilities, chemical manufacturers, aluminum processors, a shipbuilding/repair yard, and numerous storm drains. A municipal sewage treatment plant discharges into the Puyallup River immediately upstream of the tideflats area. The nearshore portion of the site is northwest of the tideflats, including waters of depths less than 60 feet below MLLW. The city of Tacoma and a major copper smelter (now closed) are located within the nearshore area. Contaminants in the waterways or sediments of the nearshore area also may have originated from drainage associated with creeks, the Puyallup River, seeps, and open channels, or nonpoint sources such as spills and atmospheric deposition.

Stressors. Toxic contaminants have been identified in sediments throughout the Commencement Bay study area (Malins et al., 1980; Washington Department of Ecology/EPA, 1985; Becker et al., 1990). Chemical analyses were completed for over 190 samples of surface and subsurface sediments collected from intertidal and subtidal areas of the Commencement Bay Superfund site (summarized in U.S. EPA, 1985). Routine analyses were conducted for about 150 chemicals. Chemicals detected in more than two-thirds of the surface sediments include phenol, 4-methylphenol, polycyclic aromatic hydrocarbons (PAHs), 1,4-dichlorobenzene, polychlorinated biphenyls (PCBs), dibenzofuran, and metals. The chemicals of concern based on sediment chemistry include 8 metals and 18 organic compounds. Chemicals of concern were selected if their concentrations in Commencement Bay Superfund site sediments exceeded the range of reference concentrations for Puget Sound. Concentrations of several contaminants in the study area sediments are substantially elevated above those characteristic of reference areas (table 5-1).

Bioaccumulation of toxic substances and associated abnormalities observed in indigenous fish and crabs found in Commencement Bay (Malins et al., 1980; Washington Department of Ecology/EPA, 1985) led to the conclusion that the contaminants were potentially hazardous to biota. In particular, PCBs were detected in muscle and liver tissues of English sole (*Parophys vetulus*) throughout the study area at concentrations substantially elevated above those found in reference areas. Other contaminants, especially PAHs, are not frequently detected in tissue samples because they are readily metabolized by fish and crabs, but PAH was suspected as a cause of the observed liver abnormalities in English sole. Alterations of benthic macroinvertebrate

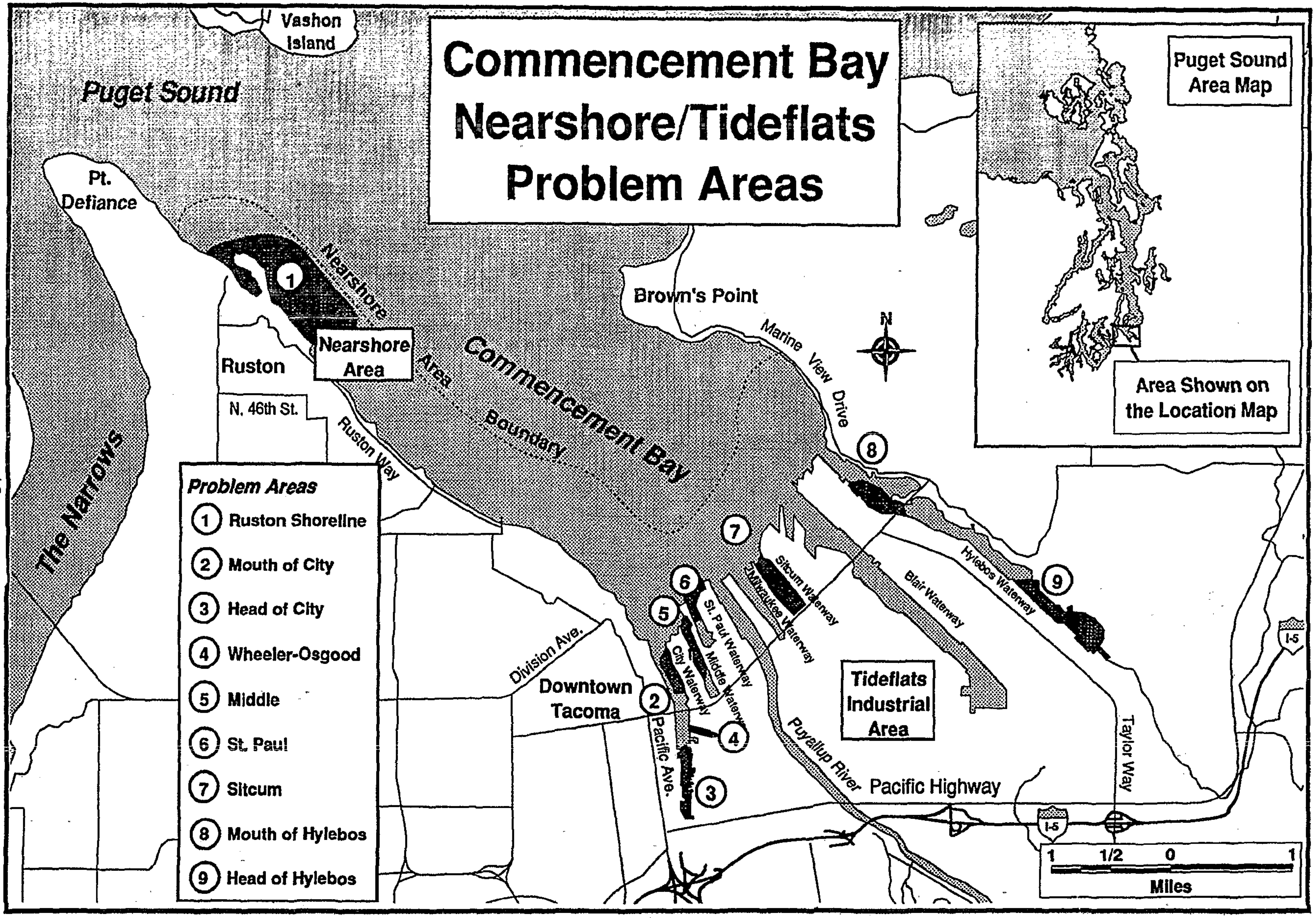

Figure 5-2. Commencement Bay nearshore/tideflats study area (U.S. EPA, 1989b)

Table 5-1. Summary of Chemicals With Sediment Elevations Above Reference (EAR) Between 100 and 1,000x and Greater Than 1,000 Averaged Over Commencement Bay (Washington Department of Ecology/U.S. EPA, 1985)

Chemicals >100x and <1,000x Reference	Waterway, Segment, or Station
Aromatic hydrocarbons (4-6 rings)	Hylebos Waterway
	City Waterway
	Ruston Shore
Aromatic hydrocarbons (1-3 rings)	City Waterway
	Middle Waterway
	Ruston Shore
Bis(2-ethylhexy)phthalate	City Waterway
Isopimaradiene	Inner Hylebos Waterway
	Sitcum Waterway
	St. Paul Waterway
	Inner City Waterway
Kaur-16-ene (tentative identification)	Inner City Waterway
	Hylebos Waterway
	Sitcum Waterway
	Milwaukee Waterway
	St. Paul Waterway
	Middle Waterway
	City Waterway
1-Methyl-2-(1-methylethyl)benzene	St. Paul Waterway
2-Methylnaphthalene	City Waterway
	Middle Waterway
1-Methylpyrene	Inner Hylebos Waterway
2-Methylpyrene	Inner Hylebos Waterway
2-Methoxyphenol	Sitcum Waterway
	Milwaukee Waterway
	Middle Waterway
	City Waterway
Total chlorinated butadienes	Hylebos Waterway
Total PCBs	Hylebos Waterway
Antimony	Ruston-Pt. Defiance
Arsenic	Ruston-Pt. Defiance
Copper	Ruston-Pt. Defiance
Lead	Ruston-Pt. Defiance
Mercury	Ruston-Pt. Defiance
Chemicals Exceeding 1,000x Reference	
Benzo(a)pyrene	Hylebos Waterway
4-Methylphenol	St. Paul Waterway
2-Methyoxyphenol	St. Paul Waterway
Phenanthrene	Ruston Shore
Trichlorobutadienes	Hylebos Waterway
Tetrachlorobutadienes	Hylebos Waterway
Antimony	Ruston Shore
Arsenic	Ruston Shore
Copper	Ruston Shore
Mercury	Ruston Shore

assemblages also were associated with sediment contamination in selected areas of the nearshore/tideflats zone (Becker et al., 1990).

Ecological Components. Sediments of the study area support a diverse assemblage of benthic organisms that can be directly influenced by sediment contamination. Toxicity may be acute or chronic depending on the contaminant, its concentration, and the sensitivity of the component. Many fish and crab species that live in close association with the sediment also feed on benthic organisms and are exposed to contaminants through the food chain. Surrogates for various groups of organisms including fish, crustaceans, and bivalves were represented in the measurement endpoints as environmental indicators of the populations or communities at risk.

Benthic macroinvertebrates. Benthic macroinvertebrates are an integral part of the Puget Sound estuarine ecosystem. Many benthic macroinvertebrate species are sedentary and consume organic materials associated with sediments. Because of their direct interaction with sediments and their sensitivity to organic enrichment and chemical contamination, they are excellent indicators of the areal extent and magnitude of environmental stress.

In the Commencement Bay study, 407 species of benthic macroinvertebrates were collected (Washington Department of Ecology/EPA, 1985). The major taxonomic groups were *Polychaeta* (marine worms), *Bivalvia* (clams), *Nematoda* (round worms), *Crustacea* (e.g., amphipods and cumaceans), *Echinodermata* (e.g., sea cucumbers and brittle stars), *Oligochaeta* (e.g., tubificid worms), and *Sipuncula* (marine worms). Two species (the polychaete *Tharyx multifilis* and the bivalve mollusc *Axinopsida serricata*) accounted for 59 percent of the benthic macroinvertebrates collected. These two species and an associated assemblage of species characterized much of the waterway system. Nematodes were the third most abundant group overall because of their high densities at a few stations. Crustaceans such as the ostracods *Euphilomedes producta* and *E. carcharodonta* and the tanaid shrimp *Leptochelia dubia* were also abundant. Commercially harvestable species of bivalve shellfish are not found in the Commencement Bay waterways.

Fishery resources. Commencement Bay supports important fishery resources. Four salmonid species (chinook, coho, chum, and pink) and steelhead inhabit Commencement Bay for part of their life cycle. These anadromous fish have critical estuarine migratory and rearing habitat requirements. Adults pass through the bay en route to their spawning grounds, and juveniles reside in nearshore habitats. Recreational and commercial harvesting of these species occurs in the bay. Inshore marine fish resources, which include flatfish such as English sole, rock sole, flathead sole, c-o sole, sand sole, starry flounder, and speckled sanddab, are the most abundant within the waterways. Rock sole, c-o sole, and special species of rockfish are most abundant along the outer shoreline.

Endpoints. The assessment endpoint in this program was the health and condition of selected components (benthic invertebrates and fish) with regard to contaminated sediments. Two indices were used as a basis for this assessment: (1) biological effects and (2) AET values. Each involves elements of both exposure and effects.

Multiple measurement endpoints at different levels of biological organization were evaluated. These endpoints included organism-level responses in sediment toxicity bioassays, population abundances of benthic macroinvertebrate species, community indices (e.g., species richness and community similarity), and biomarkers (i.e., tissue residues of contaminants and histopathology).

Sediment toxicity bioassays. Whole sediment toxicity was measured in the laboratory based on the amphipod, oyster larvae, and Microtox (saline extract) bioassays. The amphipod bioassay is an important indicator because it measures acute lethality in a crustacean species (*Rhepoxynius abronius*) that resides in the study area and is an important prey item for higher trophic-level species, especially various fishes (Swartz et al., 1985; PSEP, 1986). Use of *R. abronius* to determine the acute lethality of field-collected sediments has been documented by Swartz et al. (1982, 1985), Chapman et al. (1982a, b), and Chapman and Fink (1984). Amphipods are relatively sensitive to toxic chemicals and are highly likely to be exposed to particulate contaminants because they burrow in and feed on sediment material. The oyster larvae test measures the prevalence of developmental abnormalities in larvae exposed to sediments for 48 hours (Chapman et al., 1982b; PSEP, 1986). The oyster *Crassostrea gigas* resides in Puget Sound, although it is not found in the study area. The life stages tested (embryo and larva) are also very sensitive to toxic chemicals. The primary endpoint represents a sublethal effect that may reduce survival of larvae or affect population recruitment. The Microtox bioassay is an acute test that measures the reduction in luminescence of bacteria exposed to an extract of sediment (PSEP, 1986). Bacteria play key roles in ecosystems as decomposers and primary species in detrital-based food webs. Moreover, the Microtox test is a sensitive indicator of the effects of toxic chemicals on oxidative enzyme systems common among diverse taxa.

Benthic macroinvertebrates. The abundances of benthic macroinvertebrate species were determined from field-collected samples. Benthic macroinvertebrates are valuable indicators because they live in direct contact with sediments, are relatively stationary, and are important components of estuarine ecosystems. If sediment-associated impacts are not detected by analyses of benthic macroinvertebrates, then it is unlikely that similar population-level impacts are present in other biotic groups such as fish or plankton.

Changes in benthic macroinvertebrate assemblages were based on community-level endpoints. Community indices included the relative abundances of major taxa (i.e., polychaetes, crustaceans, molluscs, total benthos); species richness; and Bray-Curtis similarity. Only decreases in abundances of major taxa relative to reference area values were used to identify and rank problem areas.

Bioaccumulation. Contaminant concentrations in muscle tissue of English sole and Dungeness crab (*Cancer magister*) were measured as an indicator of exposure. Only the English sole data were used to identify and rank problem areas because contaminants were detected relatively infrequently in the crab muscle tissue.

Histopathology. Histopathological analyses were conducted on the livers of English sole (summarized in U.S. EPA, 1985). The prevalence of all identifiable lesions was determined. The prevalence of major lesions (i.e., preneoplastic nodules, megalocytic hepatosis, nuclear polymorphisms, and neoplasms) was the primary indicator used to identify and rank problem areas. These biomarkers have been associated with exposure to toxic chemicals, particularly PAH compounds, but causal relationships were not firmly established during the Commencement Bay Superfund investigations. Liver lesions are not definitely known to result in adverse effects on organism survival or reproduction.

> ***Comments on Problem Formulation***
>
> *Strengths of the case study include:*
>
> - *The extensive sediment chemical analyses and care in their quality assurance provide a high degree of confidence in this data set. Table 5-1 provides a good summary of the chemicals that served as the focus for much of the remedial investigation.*
>
> - *A number of ecological components and endpoints are used to quantify impact. These include several bioassay species, benthic community composition, and fish histopathology. It is also noteworthy that all these ecological components can be justified on ecological grounds. The investigators avoided limiting ecological components of concern to commercially important species or to those selected for the sake of political expediency.*

5.3.2. Analysis: Characterization of Ecological Effects

Characterization of ecological effects for the Commencement Bay Superfund project relied mainly on statistical comparisons of study sites with reference areas to define significant ($P \leq 0.05$) biological effects. Two reference areas (Carr Inlet and Blair Waterway) were used for comparisons with contaminated sites in the Commencement Bay study area. The relationships between exposure and effects for single chemicals could not be developed because complex mixtures of chemicals were found in sediments of contaminated areas. The two methods developed for characterizing effects—Indices of Biological Effects and AET—are described below followed by an overview of the program design. The two methods incorporate elements of both exposure and effects. As such, they also may be used directly within the risk characterization.

Indices of Biological Effects. A series of indices was developed based on the magnitude of observed contamination (i.e., sediment contamination) and biological effects as determined by the measurement endpoints (sediment toxicity, benthic macroinvertebrate, bioaccumulation, and histopathology variables). The indices have the general form of a ratio between the value of an effect at a Commencement Bay site and the value at a reference site. The ratios are structured so that the value of the index increases as the deviation from background conditions increases. Each ratio is termed an "elevation above reference" index. These indices are not used in lieu of the original data (e.g., contaminant concentrations), but are considered complementary forms of information. The original data are used to evaluate whether there are statistically detectable increases in contamination or effect variables and to evaluate quantitative relationships among these variables. The indices are used to reduce large data sets into interpretable numbers that reflect the different levels of contamination and effects among subareas. A matrix is constructed to integrate the individual indices in an overall evaluation and prioritization of problem areas.

Apparent Effects Threshold. Because biological effects data were not available for all portions of the study area where chemical data were available, a method was developed to estimate threshold concentrations of contaminants above which biological effects would be expected. The data base generated for each of three site-specific biological indicators (i.e., benthic macroinvertebrate abundances, amphipod mortality bioassay, and oyster larvae abnormality bioassay) was used to develop these AET values. These three indicators were selected because of their sensitivity to sediment contamination, availability of standard protocols, and ecological relevance. The AET also can be established for biological indicators that reflect areawide conditions (i.e., over multiple sediment stations) such as bioaccumulation and histopathology in fish, but the uncertainty in determining exposure area concentrations for areawide indicators is relatively high. AET values are compared with measured concentrations of sediment contaminants in a predictive method to determine the potential risk to aquatic organisms.

An AET for a chemical is defined as the concentration in sediments above which statistically significant biological effects (relative to reference sediments) would always be expected (Barrick, 1985; Chapman et al., 1982b). The AET approach uses matched (i.e., synoptically collected) data on sediment chemistry, sediment toxicity bioassays, and benthic macroinvertebrate effects (figure 5-3). To derive an AET value, sampling stations are arranged in a sequence according to the concentration of the chemical for which the AET is being determined (figure 5-3). Next, adverse effects are defined for a given biological endpoint as a statistically significant difference ($P \leq 0.05$) between conditions in a study area relative to conditions in an appropriate reference area. Stations that exhibit adverse effects are identified. The AET value is set by the no-effect station with the highest chemical concentration (i.e., all stations with chemical concentrations above the AET showed significant biological effects for the given endpoint).

To apply a set of AET values, if any chemical exceeds its AET for a particular biological indicator, then an adverse biological effect is predicted for that indicator. If all chemical concentrations are below their respective AET for a particular biological indicator, then a lack of impact is predicted for that indicator. Thus, the potential for adverse ecological effects is assessed essentially by a quotient method. The AET method does not include a probabilistic estimate of risk. Moreover, the AET approach does not prove cause-effect relationships between contaminants and effects. Nevertheless, AET values may account for some interactive effects of chemicals and for unmeasured chemicals that vary with quantifiable contaminants.

Since the development of the AET approach in the Commencement Bay Superfund project, AET values have been developed for 64 organic chemicals and metals in Puget Sound and for 4 separate biological indicators (amphipod, oyster larvae, Microtox bioassays, and benthic macroinvertebrate abundances). When applied as a set of sediment-quality screening criteria to independent data, these AET values have displayed a high reliability in predicting biological effects, while maintaining a low rate of false positives (Barrick and Beller, 1989). The AET approach is most predictive when applied to a large data base with a wide diversity of chemical

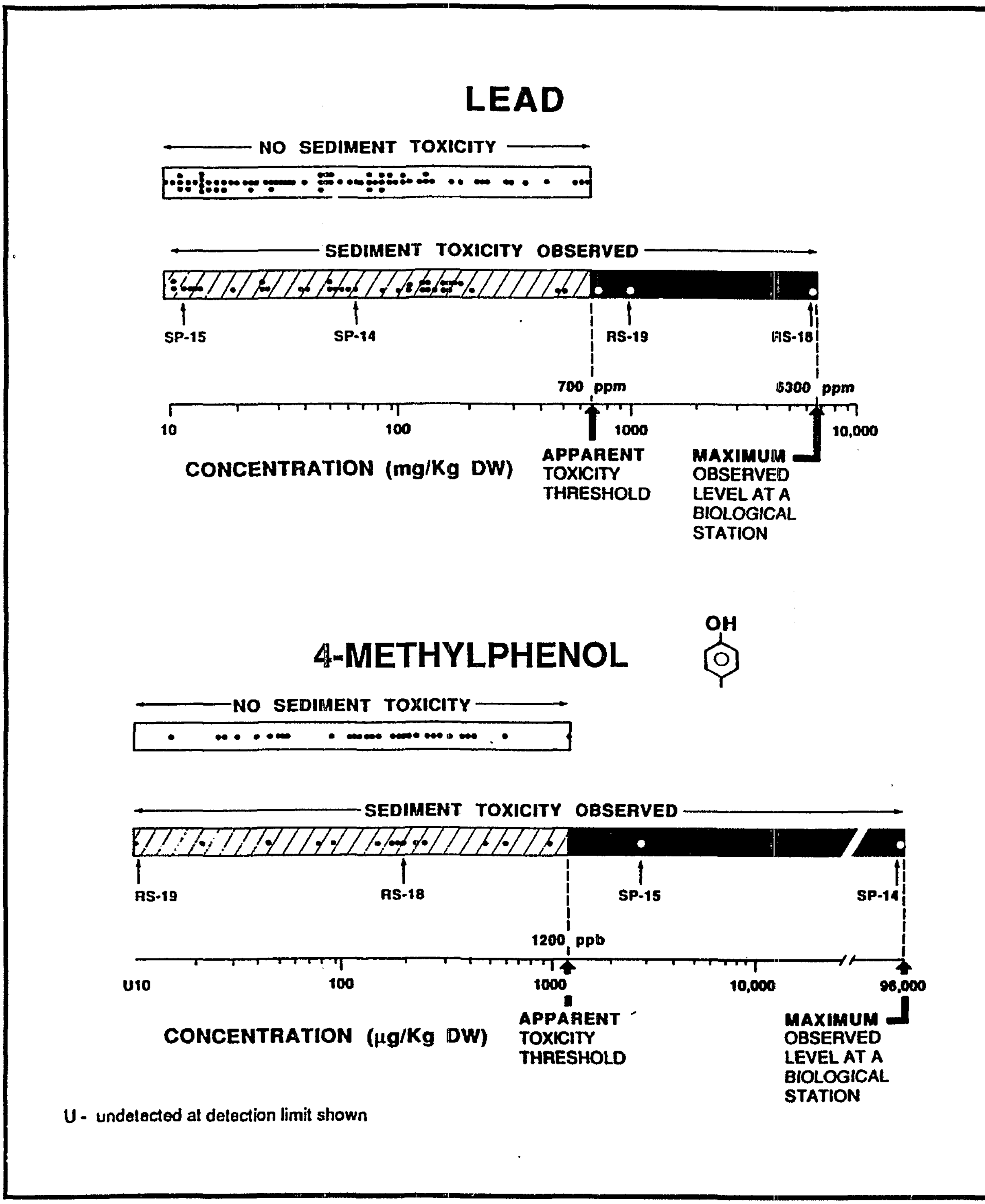

Figure 5-3. The AET approach applied to sediments tested for lead and 4-methylphenol concentrations and toxicity response during biossays (U.S. EPA, 1985)

contaminants, each represented by a broad range of concentrations. The use of 30 to 50 stations for each contaminant of concern is recommended to reduce the uncertainty associated with tests for statistical significance on small data bases.

Selection of Reference Areas. Carr Inlet was selected as a reference area because:

- a complete data set was available for Carr Inlet, including synoptic data for metals, organics, grain size, organic carbon content, and other conventional variables; and

- the lowest detection limits for most substances of concern in Puget Sound embayments were available for Carr Inlet.

The range of sediment types represented by the Carr Inlet stations sampled did not, however, encompass many of the fine-grained sediments characteristic of the Commencement Bay waterways. Therefore, it was necessary to define Blair Waterway as an internal (or nearfield) reference area. Blair Waterway was selected as a reference area because:

- it was the least chemically contaminated of the seven waterways;

- only one significant bioassay result (amphipod mortality) was found from the 12 stations tested in Blair Waterway; and

- fine-grained sediments at Blair Waterway stations spanned a range (37 to 84 percent) similar to that observed for all waterway stations in Commencement Bay except Hylebos Waterway.

Sediment Toxicity. The amphipod mortality and oyster larvae developmental bioassays were conducted on intact sediment samples (i.e., nondilution tests) and on sediment samples diluted with clean sediments. In the present study, exposure to sediments from 18 of the 52 stations tested induced statistically significant ($P \leq 0.05$) acute lethality in the amphipod *R. abronius* as compared with a reference area sediment. Statistically significantly elevated ($P \leq 0.05$) oyster larvae abnormalities were observed at 15 of the 52 stations when compared with the reference station.

Sediments from 24 of the 52 stations tested had statistically significant ($P \leq 0.05$) toxicities in either one or both of the amphipod and oyster larvae bioassays. Sediments from 10 stations were toxic in both bioassays. In some areas, sediments were toxic to the extent that a 90 percent dilution was needed to reduce amphipod toxicity responses to reference values. The level of agreement (43 percent) between lethal toxic effects for an adult organism (amphipod test) and sublethal toxic effects for a fertilized egg (oyster larvae test) enhances the weight of evidence supporting toxicity of the sediment contaminants.

Ancillary data for Microtox bioassay response were also collected. Although the Microtox data were not used as part of the ecological assessment approach for the Commencement Bay remedial investigation (Washington Department of Ecology/U.S. EPA, 1985), Williams et al. (1986) showed a significant overall concordance (Kendall's coefficient of concordance = 0.64,

$P \leq 0.05$) among the amphipod, oyster larvae, and Microtox bioassays. Comparisons of the quantitative responses for the three bioassays, however, revealed substantial heterogeneity, indicating the value of a diversity of toxicity tests in wide-scale surveys of sediment contamination.

Benthic Macroinvertebrates. To develop indices of benthic degradation as decision criteria, abundances of major benthic invertebrate taxa at potentially affected sites were compared statistically with invertebrate abundances at reference sites. A statistically significant decrease ($P < 0.05$) in the abundance of a major taxon was considered a benthic impact. At each station, indices were based on the abundance of the total assemblage (i.e., total taxa) and the abundance of polychaetes, molluscs, and crustaceans. Only *depressions* in abundance were defined as adverse impacts because an increase in abundance of one or more major taxa without a corresponding significant ($P < 0.05$) decrease in other taxa was not considered to be an adverse response at the community level.

Significant benthic effects were observed at 18 of 50 stations sampled for benthic macroinvertebrates. Benthic macroinvertebrate assemblages in Commencement Bay were found to be distinct from assemblages at reference stations, as evidenced by reduced numbers of species, high dominance, and enhanced total abundances within waterways (summarized in U.S. EPA, 1985). Dominance was evaluated as the proportion of total abundance represented by the five numerically dominant taxa in each area. Dominance ranged from 63 percent to 95 percent within the waterways, but was only 36 percent along the northwestern nearshore area and only 44 percent in Carr Inlet. The overall high abundances of a mixed polychaete-mollusc assemblage indicated that effects to benthic communities were localized. Areas having depressed abundances of at least two major taxonomic groups were limited to discrete areas in the waterways and one station near the copper smelter located along the northwestern nearshore area. The organic content of the sediments appeared to account for a considerable amount of faunal variation. In some areas (e.g., the head of City Waterway near a major storm drain), organic enrichment of the sediments was attributable to anthropogenic sources.

Bioaccumulation. Concentrations of metals in English sole muscle tissues were relatively homogeneous across locations in the study area (summarized in U.S. EPA, 1985). The maximum concentrations of most metals in fish were less than two times the average reference concentrations, but concentrations of copper in fish tissue were significantly elevated (3 to 9 times) in fish from several stations. Concentrations of lead and mercury were elevated in Dungeness crabs. Maximum concentrations of these metals were about 5 times the reference concentrations. PCBs were detected in all fish and crabs sampled. Maximum concentrations of PCBs in English sole muscle tissue exceeded reference concentrations by an order of magnitude.

Histopathology. The histopathological analyses indicated that the prevalence of liver abnormalities (e.g., preneoplastic nodules, megalocytic hepatosis, and nuclear polymorphisms) was significantly elevated ($P \leq 0.05$) in English sole collected from the Commencement Bay Superfund site compared with those from the Carr Inlet reference stations. The incidence of liver lesions was greatest in fish from areas with the highest concentrations of sediment-associated contaminants. The effects of these lesions on the fish are unknown. In this study, fish with serious liver lesions did not exhibit reduced condition (weight at a given length) when compared with fish without lesions.

Exposure-Effects Relationships and AET. Single-chemical relationships between exposure and effects could not be established from the field studies because organisms were exposed to complex mixtures of chemicals. A field-based method such as the AET approach incorporates the net effects of a variety of factors including interactive effects of chemicals, unmeasured chemicals or stressors, matrix effects, and bioavailability. Presumably, the AET value represents a threshold point on the concentration-response curve, above which significant effects ($P \leq 0.05$) are observed, but the quantification of a concentration-response relationship for a given chemical is confounded by the effects of a complex mixture in the field.

While traditional exposure-response curves for individual chemicals cannot be easily developed from field data, trends in the magnitude of biological response variables relative to distance from a source may be used to derive an *in situ* relationship between exposure and effects. In the Commencement Bay study, as the abundances of major benthic taxa increased with increasing distance from the four major sources of contamination, the toxicity response variables in the three sediment bioassays generally declined along the same spatial gradients (figure 5-4, from Becker et al., 1990). In these source areas, spatial gradients of contamination that were defined independently based on the distributions of contaminants corresponded to the gradients in biological responses.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *The characterization of ecological effects uses multiple indicators of biological impacts. Ecological effects are evaluated by three independent bioassays as well as benthic invertebrate analysis. Concordance of all these measures provides the basis for identifying and quantifying the potential effects.*

Limitations include:

- *The selection of an appropriate reference area is important to any risk assessment and essential to the Commencement Bay study because all biological and chemical measures are expressed relative to reference sites. Some limitations are associated with the reference areas selected. The Carr Inlet sites differ in sediment type and the later selection of Blair Waterway appears to be an* a posteriori *attempt to salvage the reference area concept. Blair Waterway may be the least polluted area of those studied, but it is hardly a pristine environment unaltered by the urbanization and industrialization of the tideflats.*

- *Degradation of benthic communities is characterized only by a decrease in the abundance of total amphipods, molluscs, polychaetes, or total macrofauna. While some species may decrease in abundance due to pollution, more pollution-tolerant species are likely to increase, making changes in abundance at a major taxon level an insensitive indicator. Defining degradation only by a decrease in abundance is particularly weak for polychaetes and, by inference, total macrofauna.*

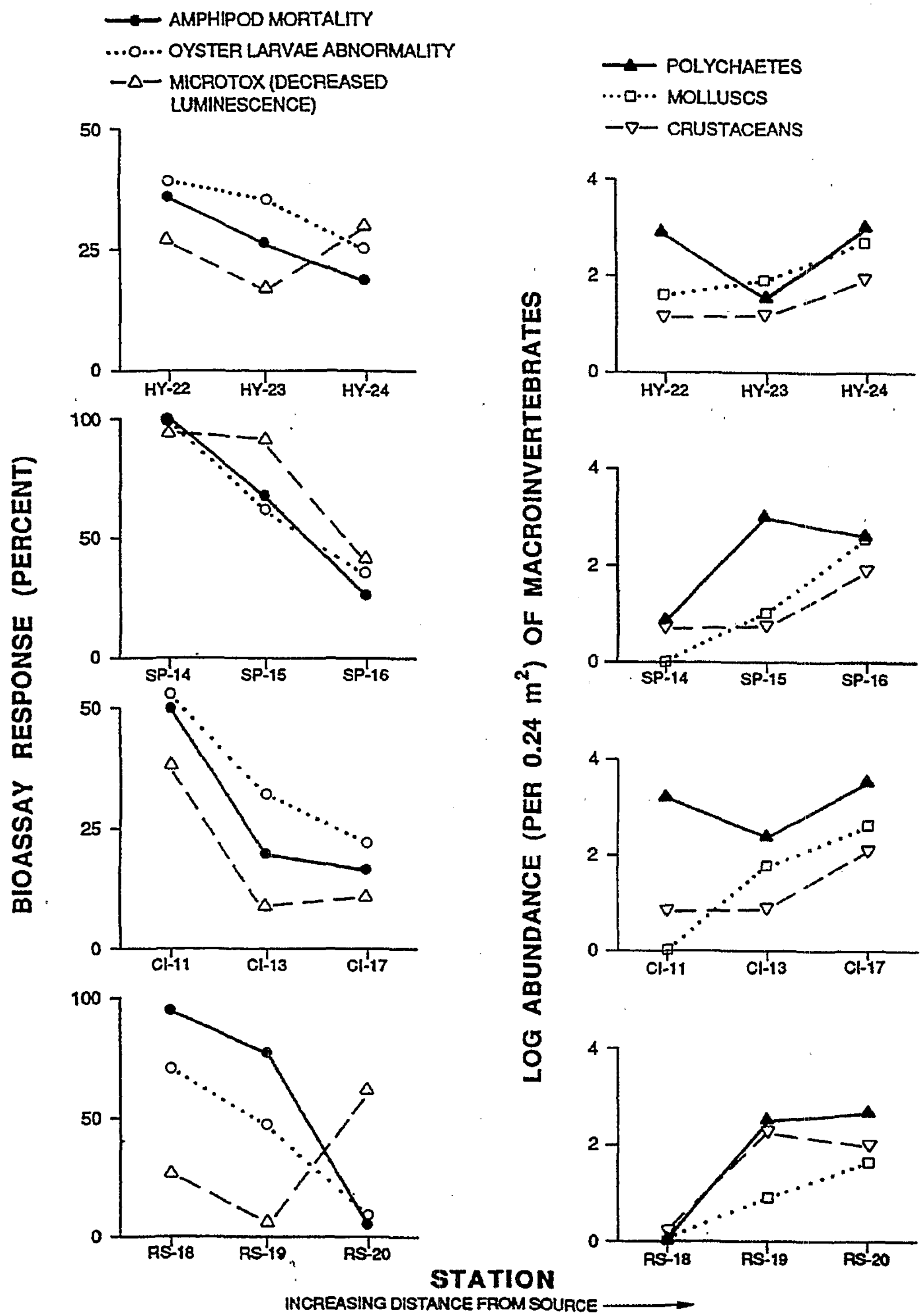

Figure 5-4. Bioassay responses and abundances of benthic macroinvertebrate taxa near the four major contaminant sources evaluated in Commencement Bay (Becker et al., 1990)

5.3.3. Analysis: Characterization of Exposure

The study of environmental effects in Commencement Bay focused on exposure to contaminated sediments. The data on water column concentrations of contaminants were not used to estimate risks to aquatic organisms because of the potential for high spatial and temporal variability, the effects of environmental factors such as currents or salinity, and the association of most of the contaminants of concern with particulate matter that was deposited in the sediments.

Station Locations. After a review of historical information and data from the preliminary survey (summarized in U.S. EPA, 1985), sediment stations in the study area were selected for sample collection and analysis. Stations were selected to:

- fill data gaps;
- define known areas of contamination more precisely; and
- determine gradients of contamination in relation to suspected sources.

Magnitude, Frequency, and Duration of Exposure. The magnitude of exposure is determined by the concentrations of contaminants in sediments. Sediments throughout the Commencement Bay study area contain concentrations of one or more toxic contaminants that exceed levels commonly found in reference areas. Table 5-1 lists chemicals or chemical groups with sediment concentrations that were elevated between 100 and 1,000 times reference conditions at one or more stations and chemicals that had concentrations that were 1,000 times greater than reference area concentrations. Because the sediments represent a sink for contaminants, exposure is essentially continuous for organisms that reside in or on the sediments. Water-column species may be periodically exposed to contaminants by resuspension of bottom sediments. The frequency of sediment resuspension events was not characterized in the Commencement Bay project.

Natural recovery of contaminated sediments is the process whereby the contamination in the upper sediment layers is reduced due to elimination of sources, burial, biodegradation, biological uptake, and diffusion (U.S. EPA, 1989b). A mass balance equation predicting sediment concentration in relation to source loading, sedimentation rates, sediment mixing, biodegradation, and diffusion was used to describe the recovery process. Results of this modeling suggested that natural sedimentation (i.e., control of anthropogenic sources was assumed) is expected to cover the study site with 10 centimeters of relatively clean sediments over a 10-year period.

Comments on Analysis: Characterization of Exposure

- *The characterization of exposure often takes the form of a predictive modeling exercise to quantify the probability and magnitude of contaminant exposure given some particular toxicant discharge or remedial action. Such an approach is not relevant to the Commencement Bay investigation where the toxicant release has already occurred, biological effects have been measured, and no remedial activities are addressed within this case study. In this case, characterization of exposure is implicit within the elements of both characterization of ecological effects and risk characterization.*

5.3.4. Risk Characterization

Ideally, characterization of ecological risks due to exposure to sediments would be supported by definitive cause-effect relationships between specific chemicals and biological endpoints. To date, however, very little information of this type is available for mixtures of chemicals in the environment. In lieu of definite risk estimates, relative measures of effect have been developed. In the Commencement Bay project, a decision-making approach based on empirical measures of risk (i.e., elevation above reference indices of biological effects) and a predictive assessment (i.e., AET approach) were developed to identify and rank problem areas for remediation.

Risks Based on Comparisons to Benchmarks. Evaluation of elevation above reference indices for each indicator showed that chemical contamination of sediment was spatially heterogeneous (table 5-1). The average values of elevation above reference indices are shown for each of the Commencement Bay stations in table 5-2 (U.S. EPA, 1989b).

The average concentrations of several organic compounds exceeded all Puget Sound reference conditions in all study areas. Average elevation above reference values for selected metals, sediment toxicity bioassay endpoints, and chemicals indicative of bioaccumulation were significantly elevated above reference stations in six of the seven study areas evaluated. Depressions of benthic macroinvertebrate taxa were statistically significant ($P<0.05$) in five of the seven study areas; liver lesions in English sole were statistically significant ($P<0.05$) in four of the seven study areas.

Action-level guidelines used to identify problem areas are summarized in table 5-3 (see U.S. EPA, 1985, for details). For example, areas requiring further evaluation of contaminant sources and remedial actions were identified when values for three or more indices were significantly elevated. Other combinations of significant indices and the magnitude of the elevations above reference triggered problem area definition as well (table 5-3). With the guidelines in table 5-3, problem areas were identified within the Commencement Bay Superfund site. Six segments within the large study areas had significant elevations above reference for all three site-specific indicators (contamination, toxicity, and benthic effects).

Table 5-2. Action Assessment Matrix of Sediment Contamination, Sediment Toxicity, and Biological Effects Indices for Commencement Bay Study Areas (U.S. EPA, 1985)

Variable	Study Area Elevations[a]								Reference Value[b]
	Hylebos	Blair	Sitcum	Milwaukee	St. Paul	Middle	City	Ruston	
Sediment Chemistry (ppb)									
Sb	10.0	4.0	8.0	3.6	4.2	9.3	7.0	510.0	110.0
As	12.0	7.6	11.0	3.6	2.2	9.6	7.5	620.0	3370.0
Cd	2.4	1.9	2.8	1.7	1.7	2.8	5.5	27.0	950.0
Cu+Pb+Zn	10.0	4.8	24.0	7.3	5.5	18.0	22.0	120.0	35,000.0
Hg	8.1	< 3.7	5.0	3.8	5.1	26.0	10.0	160.0	40.0
Ni	1.4	0.7	0.6	0.8	0.8	0.7	1.4	2.8	1,740.0
Phenol	< 6.4	< 5.2	4.3	< 2.1	12.0	11.0	9.4	4.5	< 33.0
PCP	1.7	< 2.3	< 2.1	< 0.9	U 1.9	5.6	1.9	< 1.0	U 33.0
LPAH	< 45.0	<28.0	<68.0	<60.0	<73.0	<110.0	120.0	< 87.0	< 41.0
HPAH	<120.0	<42.0	<65.0	<68.0	<27.0	< 97.0	<140.0	< 85.0	< 79.0
Cl. benzene	9.9	< 4.4	2.6	< 2.5	< 1.8	< 6.1	< 9.0	< 3.3	U 21.0
Cl. butadiene	130.0	< 2.7	< 2.4	< 1.2	< 1.3	< 6.8	< 1.9	1.7	U 62.0
Phthalates	4.0	< 2.6	0.58	< 0.66	< 0.56	< 5.1	< 7.1	4.5	<280.0
PCB	< 48.0	< 6.3	7.6	6.6	<17.0	8.5	< 12.0	19.0	< 6.0
4-Methylphenol	< 7.3	<12.0	10.0	13.0	1,300.0	< 33.0	30.0	< 10.0	< 13.0
Benzyl alcohol	5.0	< 2.2	2.4	3.4	< 6.7	3.3	4.7	< 1.2	U 10.0
Benzoic acid	< 0.7	< 3.2	0.5	< 0.7	< 1.0	U 0.1	< 1.6	< 0.5	<140.0
Dibenzofuran	29.0	25.0	73.0	59.0	52.0	80.0	58.0	<160.0	U 3.7
Nitrosodiphenylamine	2.1	< 2.4	< 7.3	U 1.0	U 1.2	< 1.0	14.0	< 22.0	U 4.1
Tetrachloroethene	12.0	< 0.6	U 1.0	–	U 1.0	–	U 1.0	–	U 10.0
Sediment Toxicity (%)									
Amphipod bioassay	2.1	1.9	2.9	2.4	4.8	1.4	2.7	3.9	9.3
Oyster bioassay	2.2	1.6	1.3	1.4	3.8	1.8	2.6	2.2	13.0
Infauna[c]									
Total benthos	1.2	1.0	0.7	0.8	1.9	1.5	0.7	0.6	d
Polychaetes	0.6	1.0	0.4	0.7	1.5	0.7	0.8	0.5	d
Molluscs	3.4	1.0	1.4	1.1	6.8	5.4	2.5	1.2	d
Crustaceans	1.5	1.0	3.8	0.4	1.0	4.6	1.2	0.7	d
Fish Pathology (%)									
Lesion prevalence	3.6	2.5	3.5	3.7	2.7	5.7	1.7	2.1	6.7
Fish Bioaccumulation (ppb)									
Copper	5.6	1.0	4.0	2.3	9.1	1.0	3.8	2.5	U 38.0
Mercury	1.5	0.93	0.8	1.6	0.76	1.3	0.82	0.96	U 55.0
Naphthalene	0.67	0.41	0.33	24.0	0.19	0.19	4.1	0.19	< 54.0
Phthalates	21.0	11.0	0.53	3.6	0.41	0.41	6.7	5.6	< 74.0
PCB[e]	9.2	7.0	4.8	2.8	1.1	4.7	9.8	1.9	< 36.0
DDE	3.8	5.1	3.3	3.4	1.7	1.7	6.2	2.9	< 1.8

[a]Boxed numbers represent elevations of chemical concentrations that exceed all Puget Sound reference area values, and statistically significant toxicity and biological effects at the $P<0.05$ significance level compared with reference conditions. The "U" qualifier indicates the chemical was undetected and the detection limit is shown. The "<" qualifier indicates the chemical was undetected at one or more stations. The detection limit is used in the calculations.
[b]Elevation above reference (EAR) values shown for each area are based on Carr Inlet reference values for each variable except for benthos (see footnote d).
[c]Infauna EAR are based on the ratio of population abundances at the reference site to those at the Commencement Bay sites. For example, the EAR for total benthos at Hylebos is 1.2, meaning that the abundance of benthic populations at the reference site is 20 percent greater than at the Commencement Bay site. Since decreases in abundance of infauna are considered to be adverse effects, higher rates of reference site to Commencement Bay site abundance reflect a greater likelihood of adverse biological effects.
[d]Different benthic reference values were used depending on sediment grain size.
[e]Locations where PCB concentrations are significantly elevated also pose a significant health risk to the exposed population.

Table 5-3. Action-Level Guidelines (Washington Department of Ecology/U.S. EPA, 1985)

Condition Observed	Threshold Required for Action
I. Any THREE OR MORE significantly elevated indices[a]	Threshold exceeded, continue with definition of problem area.
II. TWO significantly elevated indices	
1. Sediments contaminated, but below 80th percentile PLUS: Bioaccumulation without an increased human health risk relative to that at the reference area, OR Sediment toxicity with less than 50 percent mortality or abnormality, OR Major benthic invertebrate taxon depressed, but by less than 95 percent	No immediate action. Recommended site for future monitoring.
2. Sediments contaminated but below 80th percentile PLUS elevated fish pathology	Threshold for problem area definition exceeded if elevated contaminants are considered to be biologically available. If not, recommended site for future monitoring.
3. Any TWO significantly elevated indices, but NO elevated sediment contamination	Conduct analysis of chemistry to distinguish site from adjacent areas. If test fails, no immediate action warranted. Otherwise, threshold exceeded for characterization of problem area. Re-evaluate significance of chemical indicators.
III. SINGLE significantly elevated index	
1. Sediment contamination	If magnitude of contamination exceeds the 80th percentile for all study areas, recommended area for potential source evaluation at a low priority relative to areas exhibiting contamination and effects.
2. Bioaccumulation	Increased human health threat, defined as: Prediction of greater than or equal to 1 additional cancer cases in the exposed population for significantly elevated carcinogens, OR For noncarcinogens, exceeding the acceptable daily intake value is required.
3. Sediment toxicity	Greater than 50 percent response (mortality or abnormality).
4. Depressed benthic abundance	95 percent depression or greater of a major taxon (equals an EAR of 20 or greater).
5. Fish pathology	Insufficient as a single indicator. Recommended site for future monitoring. Check adjacent areas for significant contamination, toxicity, or biological effects.

[a]Combinations of significant indices are from independent data types (i.e., sediment chemistry, bioaccumulation, sediment toxicity, benthic infauna, fish pathology).

Significant indices are defined as follows: sediment chemistry = chemical concentration at study site exceeds highest values observed at any Puget Sound reference area.

Sediment toxicity, benthic abundance, bioaccumulation, and pathology = statistically significant ($P<0.05$) difference between study area and reference area.

Application of AET. During the Commencement Bay remedial investigation, AETs were generated for three categories of biological effects variables (i.e., amphipod mortality, oyster larvae abnormality, and benthic macroinvertebrate taxa abundance) for a data set of 50 to 60 stations. Following the remedial investigation, the AET data set was expanded to 334 stations, including data from other areas of Puget Sound. Table 5-4 lists AET values used to define sediment quality objectives for the cleanup of Commencement Bay sediments during the feasibility study (from U.S. EPA, 1989b).

These values represent the lowest AET for the three biological effects indicators. Toxicity or benthic AET (i.e., the concentration above which all sediments had significant toxicity or benthic effects, respectively) was exceeded by several chemicals at most, but not all, of the 29 biological stations exhibiting significant effects. A detailed discussion of the AET for each biological effect is presented in chapter 4 of the remedial investigation report (Washington Department of Ecology/U.S. EPA, 1985) and summarized in U.S. EPA (1985).

Ranking of Study Areas. Prioritization of study areas was based on average and maximum conditions in each area that exceeded the action-level guidelines. Results of the average ranking method and the maximum ranking method were then compared. The spatial extent and general priority of all problem areas for evaluation of sources and remedial action in the Commencement Bay Superfund project are illustrated in figure 5-5. Problem areas defined only by mid-channel stations in the current study were assumed to extend from shoreline to shoreline, unless historical data indicated otherwise. Fourteen of the 21 problem areas identified were recommended for priority source evaluation. Six areas were defined as the lowest priority areas for source evaluation because they contained stations where contaminant concentrations exceeded AET, but no confirming biological data were available. The seventh area not recommended for high-priority source evaluation (Milwaukee Waterway) contained no chemicals measured above their AET.

Based on the ranking according to environmental indices, AET, spatial extent of contamination, and confidence in source identification, nine discrete areas of sediment contamination were identified (figure 5-5). These areas were designated as requiring future evaluation and response under the Superfund program. Potential problem chemicals of varying priorities for source identification were also identified in each of the 14 areas recommended for priority source evaluation.

Uncertainty Analysis. Two measures of reliability of the AET approach were evaluated with actual field data from 13 urban and nonurban embayments in Puget Sound (summarized in U.S. EPA, 1988, based on Commencement Bay data; also see Barrick and Beller, 1989, for an evaluation based on an expanded AET data base): (1) sensitivity in detecting environmental problems (i.e., are all biologically affected sediments identified?) and (2) efficiency in screening environmental problems (i.e., are only biologically affected sediments identified?).

These measures of reliability were applied to a range of sediment criteria generated by the Equilibrium Partitioning (EP) and AET approaches. Overall reliability ranged from 44 to 64 percent for the EP approach and from 42 to 85 percent for the AET approach, depending on the particular criterion and biological indicator tested. A higher percentage of correct predictions was made using a combination of the two approaches than by using either approach alone.

Table 5-4. Sediment Cleanup Objectives Related to Environmental Risks (U.S. EPA, 1989b)

Chemical	Sediment Cleanup Objective[a]
Metals (mg/kg dry weight)	
Antimony	150[B]
Arsenic	57[B]
Cadmium	5.1[B]
Copper	390[L]
Lead	450[B]
Mercury	0.59[L]
Nickel	>140[AB]
Silver	6.1[A]
Zinc	410[B]
Organic compounds (μg/kg dry weight)	
Low-molecular-weight PAHs	5,200[L]
Naphthalene	2,100[L]
Acenaphthylene	1,300[AB]
Acenaphthene	500[L]
Fluorene	540[L]
Phenanthrene	1,500[L]
Anthracene	960[L]
2-Methylnaphthalene	670[L]
High-molecular-weight PAHs	17,000[L]
Fluoranthene	2,500[L]
Pyrene	3,300[L]
Benz(a)anthracene	1,600[L]
Chrysene	2,800[L]
Benzofluoranthenes	3,600[L]
Benzo(a)pyrene	1,600[L]
Indeno(1,2,3-c,d)pyrene	690[L]
Benzo(a,h)anthracene	230[L]
Benzo(g,h,i)perylene	720[L]
Chlorinated organic compounds	
1,3-Dichlorobenzene	170[ALB]
1,4-Dichlorobenzene	110[B]
1,2-Dichlorobenzene	50[LB]
1,2,4-Trichlorobenzene	51[A]
Hexachlorobenzene (HCB)	22[B]
Total PCBs	1,000[B]

[a]Option 2 - Lowest AET among amphipod, oyster, and benthic infauna:

A - Amphipod mortality bioassay;
L - Oyster larvae abnormality bioassay;
B - Benthic infauna

Table 5-4. Sediment Cleanup Objectives Related to Environmental Risks (continued)

Chemical	Sediment Cleanup Objective[a]
Phthalates	
Dimethyl phthalate	160[L]
Diethyl phthalate	200[B]
Di-n-butyl phthalate	1,400[AL]
Butyl benzyl phthalate	900[AB]
Bis(2-ethylhexyl)phthalate	1,300[B]
Di-n-octyl phthalate	6,200[B]
Phenols	
Phenol	420[L]
2-Methylphenol	63[AL]
4-Methylphenol	670[L]
2,4-Dimethylphenol	29[L]
Pentachlorophenol	360[A]
Miscellaneous extractables	
Benzyl alcohol	73[L]
Benzoic acid	650[LB]
Dibenzofuran	540[L]
Hexachlorobutadiene	11[B]
N-nitrosodiphenylamine	28[B]
Volatile organics	
Tetrachloroethene	57[B]
Ethylbenzene	10[B]
Total xylenes	40[B]
Pesticides	
p,p'-DDE	9[B]
p,p'-DDD	16[B]
p,p'-DDT	34[B]

[a]Option 2 - Lowest AET among amphipod, oyster, and benthic infauna:

A - Amphipod mortality bioassay;
L - Oyster larvae abnormality bioassay;
B - Benthic infauna

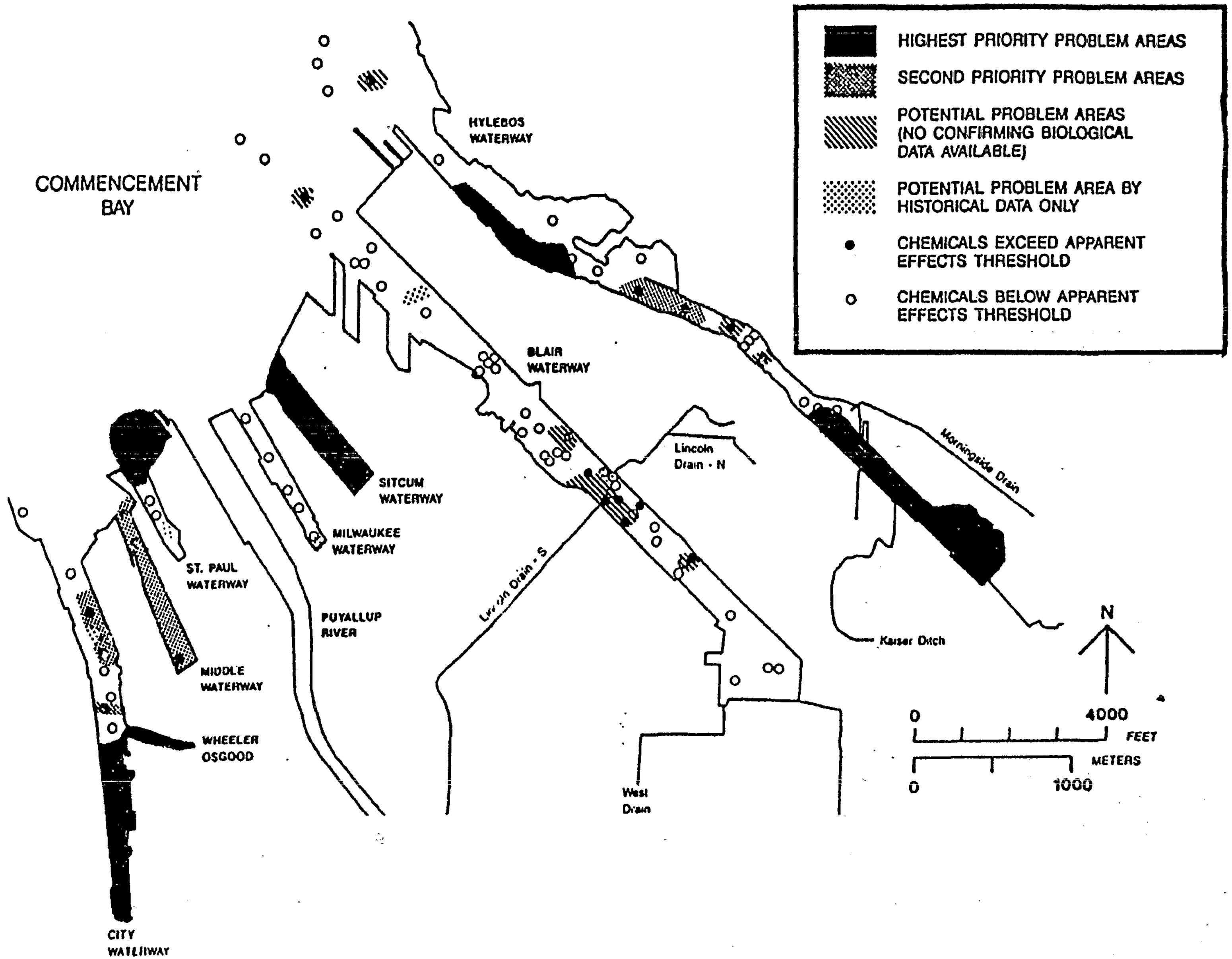

Figure 5-5. Definition and prioritization of Commencement Bay problem areas (U.S. EPA, 1989b)

The EPA Science Advisory Board (U.S. EPA, 1989a) evaluated the AET approach for assessing sediment quality. The conclusions of the Subcommittee on Sediment Criteria are:

> The method has major strengths in its ability to determine biological effects and assess interactive chemical effects. The method is considered by the Subcommittee to contain sufficient scientific merit that, with appropriate validation of the AET values, it could be used to establish sediment quality values for use at specific sites. In the Subcommittee's opinion, the AET approach should not be used to develop general, broadly applicable sediment quality criteria. Some major limitations drive this opinion, including the site-specific nature of the approach, its inability to describe cause-and-effect relationships, its lack of independent validation, and its inability to describe differences in bioavailability of chemicals in different sediments. The Subcommittee recommendations for strengthening the approach include building in replicate sediment samples to assessments, devising criteria for selecting reference sites, including considering physical factors and developing measures of variance.

Comments on Risk Characterization

Strengths of the case study include:

- *The use of data on multiple chemical measurements and biological endpoints (e.g., sediment chemistry, sediment toxicity, benthic macroinvertebrate assemblages, tissue residues resulting from bioaccumulation, and fish liver histopathology) provides a powerful weight-of-evidence approach to identify and rank problem areas.*

- *The AET is an innovative approach dealing with the problems created by contaminant suites and uncertain cause-effect relationships. The approach is a noteworthy attempt to use empirical relationships to sidestep currently intractable issues like bioavailability and synergistic/antagonistic effects among toxicants. The combination of field-collected sediment bioassays and the AET approach takes a step toward differentiating between effects associated with different contaminants. The predictive AET approach relies on objective statistical criteria for determining adverse effects for each biological indicator. The spatial extent of areas of high risk may be delineated using the concordance between biological response and chemical concentrations.*

- *Disparate chemical and biological data sets are integrated into the definition of problem areas. By expressing all chemical and biological measures as elevations relative to a reference site, comparisons among these measures and demonstration of concordance becomes straightforward. It should be noted, however, that this requires reducing biological endpoints to single values. This may be appropriate for some endpoints, such as percentage of mortality in the bioassays, but it becomes more problematic and results in substantial data loss when applied to a complex biological response such as benthic community change.*

Comments on Risk Characterization (continued)

Limitations include:

- *The ecological assessment of Commencement Bay does not provide a probabilistic approach to risk characterization. Also, the case study does not include a predictive assessment of contaminant transport and fate in relation to the distribution of ecological components.*

- *The ecological significance of some of the measurement endpoints is not explained, particularly with respect to individual site characteristics. For example, it is not clear how the adverse effects observed in the bioassay tests are representative of the insult to the entire ecosystem of Commencement Bay.*

General comments:

- *There are some limitations in the use of the AET method. The definition of the AET as the highest concentration at which no effect is observed (rather than the lowest concentration at which any effect is observed) is the least protective of the possible definitions for effects thresholds. Moreover, a correlation between contaminant concentrations and biological effects does not demonstrate cause and effect. The AET method assumes a consistently increasing biological response at increasing concentrations of a chemical. Unmeasured chemicals or physical conditions may alter this relationship. Species interactions and other community-level processes may also alter the assumed dose-response relationship. A large data base of synoptic chemistry and biological effects is needed to verify a particular AET value for a specific chemical.*

- *The Commencement Bay investigation was not originally conceived as a risk assessment in the sense that it was neither predictive nor probabilistic. It was an impact assessment intended to define the extent of contamination in marine sediments and quantify the magnitude of existing biological damage. Consequently, application of risk assessment terminology becomes confusing at times.*

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SECTION SIX

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

THE NATIONAL CROP LOSS ASSESSMENT NETWORK

AUTHOR AND REVIEWERS

AUTHOR

Walter W. Heck
Agricultural Research Service
U.S. Department of Agriculture
Raleigh, NC

REVIEWERS

Douglas P. Ormrod (Lead Reviewer)
Office of Graduate Studies
University of Guelph
Guelph, Ontario, Canada

Nancy A. Bryant
ENSR Consulting & Engineering
Acton, MA

Kenneth L. Dickson
Institute of Applied Sciences
University of North Texas
Denton, TX

Thomas M. Frost
Center for Limnology
University of Wisconsin
Madison, WI

Judy L. Meyer
Institute of Ecology
University of Georgia
Athens, GA

Randall S. Wentsel
Toxicology Division
U.S. Army Chemical Research, Development, and Engineering Center
Aberdeen Proving Grounds, MD

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LIST OF ACRONYMS

EPA	U.S. Environmental Protection Agency
LRT	likelihood ratio test
NAAQS	National Ambient Air Quality Standards
NCLAN	National Crop Loss Assessment Network
NF	nonfiltered
RYL	relative yield losses
SAROAD	Storage and Retrieval of Aerometric Data
USDA	U.S. Department of Agriculture

ABSTRACT

The National Crop Loss Assessment Network was initiated by the U.S. Environmental Protection Agency in 1980 to develop an approach for assessing the impact of ozone on crop production. Four primary and two secondary regional sites were established to conduct field studies from 1980 through 1986. Forty-four field experiments, using regression designs with 17 crop species (including 38 cultivars and 3 corn crosses), were run to determine the impact of ozone on growth and productivity. Physiological studies also were carried out in these designs. Plants were grown under field conditions and exposed to different ozone concentrations in open-top field chambers. Thirty-five experimental designs were factorial and included secondary stressors (i.e., low soil moisture, sulfur dioxide); cultivar testing; testing of dispensing methodology; or detailed growth studies. Results from these studies helped investigators interpret results from the ozone studies. Yield loss estimates for the economic analysis were obtained using the Weibull function and were reported as relative yield losses. An ozone statistic was identified (a seasonal mean of the 7-hr or 12-hr daily exposure period), and ozone monitoring data across the United States were interpolated by a kriging technique to give spatial-level ozone statistics. Crop yields on the same spatial basis were obtained from the U.S. Department of Agriculture. National yield losses, estimated from the models, showed reductions from 0 to 22 percent at a seasonal mean value of 0.05 ppm ozone. Results were used in an economic model to derive estimated producer and consumer benefits with increasing and decreasing ozone concentrations. The final economic assessment using data from eight crops estimated that increased yields associated with a 25 percent reduction in tropospheric ozone resulted in a $1.9 billion annual net benefit, while a 40 percent reduction gave almost $3 billion in net annual benefits. The study was limited in size and only one or two sites were used (total of six) in four regions of the country. The results of the effects of soil moisture on plant yield response were not conclusive, and very little was done to determine the effects of multiple abiotic and biotic stressors on crop yield response. Major scientific issues, in addition to those mentioned above, include: (1) effects of the field methodology on plant response to ozone; (2) the most applicable concentration statistic to use; (3) the accuracy of the kriging approach; and (4) understanding plant processes in relation to the impact of ozone so that more process-oriented models can be developed.

6.1. RISK ASSESSMENT APPROACH

The National Crop Loss Assessment Network (NCLAN) study was designed as a risk assessment and, with some modification, was fit to four parts of the ecological risk assessment framework (figure 6-1): The ecological components were characterized and endpoints were defined; the ozone exposure-plant response was studied experimentally in the field, and response models were developed; exposure characteristics were described and documented; and risks were characterized in both crop yield and economic terms. Primary limitations for a national risk assessment are in the spatial representativeness of the test sites (only six for a national assessment) and in the few experimental designs set up to assess the effects of interacting stressors. The principal reference used in this case study was Heck et al. (1991).

6.2. STATUTORY AND REGULATORY BACKGROUND

The Clean Air Act (1970) required the U.S. Environmental Protection Agency (EPA) to set National Ambient Air Quality Standards (NAAQS) for "any air pollutant which, if present in the air, may reasonably be anticipated to endanger public health or welfare and whose presence in the air results from numerous or diverse mobile and/or stationary sources" (U.S. EPA, 1986). EPA is responsible for developing and promulgating both primary (public health) and secondary (public welfare) NAAQS. The 1977 amendments to the Clean Air Act required that the criteria (scientific basis) for the standards be periodically reviewed and revised to include new information. In 1978, EPA reviewed the scientific literature to determine the impact of ozone on vegetation and published this analysis in the *Air Quality Criteria for Ozone and Other Photochemical Oxidants* (U.S. EPA, 1978). As a result of this analysis, EPA accepted the primary standard as a reasonable secondary standard. The credibility of the secondary standard suffered, however, because the analysis lacked sufficient data either to determine reliably the effects of ozone on the yield of major agronomic crops under field conditions or to determine resultant economic consequences. Hence, EPA established NCLAN to assess the impact of ozone on agricultural resources and to provide the most useful data and criteria for the next review of the ozone standard.

6.3. CASE STUDY DESCRIPTION

6.3.1. Problem Formulation

Site Description. NCLAN was initiated by EPA in 1980 to develop an approach for assessing the impact of ozone on crop production in the United States. Four primary and two secondary regional field study sites were used from 1980 through 1986 to develop exposure-response relationships between ozone exposure dose and crop yield response. The six sites represented different regions of the country (Southeast, Northeast, North-central, Southwest) and had well-established air quality research programs.

Stressors. Plant scientists have been concerned with the effects of ozone on vegetation since ozone was first identified as a phytotoxicant on grapes and tobacco in 1958 (Heck et al., 1977). Ozone is photochemically formed in the troposphere by the action of sunlight on NO_x and certain reactive hydrocarbon gases. Although both NO_x and reactive hydrocarbons are naturally produced, concern exists over human activities that produce these gases. Currently, ozone

Figure 6-1. Structure of Analysis for the National Crop Loss Assessment Network

PROBLEM FORMULATION

Stressors: ozone in ambient air.

Ecological Components: 17 crop species.

Endpoints: the primary assessment and measurement endpoint was the impact of ozone on yield of the crop part important for human use.

ANALYSIS

Characterization of Exposure

Monitoring data for ambient ozone from over 300 sites were used to calculate seasonal concentrations using a kriging model.

Characterization of Ecological Effects

In-field chambers were used to examine relationships between ozone exposure and plant yield. The Weibull model was used to describe dose-response relationship.

RISK CHARACTERIZATION

Ozone exposure dose-crop yield response functions were used to predict loss of yield in test species throughout the country based on the calculated ozone values. The estimated losses of yield were used as part of an economic assessment.

produced photochemically from gases associated with human activities probably accounts for between one- and two-thirds of the tropospheric ozone found during the growing season in most parts of the United States. Control of ozone requires control of nitrogen oxides or non-methane hydrocarbons, or both (U.S. EPA, 1986).

The principal focus of the NCLAN program was to quantify the relationship between ozone concentration and reduced yield in economically important, agronomic crops (Heck et al., 1982). The primary stressor was ozone. Research on crop response to ozone and extensive field observations prior to the beginning of the NCLAN program gave clear evidence that all crop species were sensitive to ozone (Heck et al., 1977; Heagle and Heck, 1980). Further research documented yield losses due to ozone exposure for a number of crops (Heck et al., 1986).

Ozone injury was first identified by foliar symptoms, often described as necrotic flecking or stipple on leaves. All early studies used foliar injury as an indication of the severity of plant response. Ozone has been shown to reduce photosynthetic efficiency, alter carbon allocation to various plant parts, affect many metabolic plant processes, reduce yield, and change food quality. There are differences in both species and cultivar sensitivities to ozone. Injury to many crop species (i.e., bean, watermelon, cotton, peanut, soybean, oat, clover, potato) has been observed in the field under ambient conditions of ozone (Heck et al., 1977; U.S. EPA, 1986).

Ecological Components. Because its focus was primarily economic, the NCLAN program studied crops that represented the major production crops in the country. The degree of sensitivity played no role in crop selection. The growing season ranged from April through September in most areas of the country, with a longer season available in the southern areas. Seed crops were most sensitive during flowering and fruit development.

The crop species were chosen because they are economically valuable to the country (together they represent about 85 percent of current crop acreage), they are of regional importance, and they represent a number of crop types. The 17 species studied are shown in table 6-1. This table also summarizes the number of studies, the number of cultivars, and the interacting factors in different designs (Heck et al., 1991).

Endpoint Selection. Agroecosystems are relatively simple systems that are highly managed to provide maximum crop yield under any given set of field conditions. The NCLAN program was designed to permit a national assessment of the impact of ozone on crop yield (Heck et al., 1982, 1983). The yield information was then used to determine the economic losses related to ozone impact on crop production systems in the United States.

The primary measurement and assessment endpoint was yield of the crop part important directly or indirectly for human use (Heck et al., 1982, 1983). Although yield is one of the primary assessment endpoints, the economic loss/gain in dollars was also considered an assessment endpoint in this study. Additional measurement endpoints used in the study but not addressed in this summary were a number of physiological (net photosynthesis, water use efficiency, water use) and biochemical (oxidative enzymes, metabolic systems, and crop quality) parameters that could eventually be used for process-level model development. From an ecological risk assessment approach, the economic analysis was an extra step in the study. Additional measurement endpoints

Table 6-1. Summary of Crop Studies in the NCLAN Program (Heck et al., 1991)[a]

Crop	No. of Studies 1 yr.	No. of Studies 2 yr.	No. of Cultivars	Ozone and Other Factors (No. of Studies)
Alfalfa	1	1	2	Moisture (1), SO_2 (1)
Barley	2		2	Moisture (1)
Corn	2		5	SO_2 (1)
Cotton	5		5	Moisture (4), SO_2 (1)
Forage				
Timothy/Red Clover		1	1/1	
Fescue/Ladino Clover		1	1/1	Moisture (1)
Bean	2		1	---
Lettuce	2		1	---
Peanut	1		1	---
Sorghum	1		1	---
Soybean	14		9	Moisture (7), SO_2 (4)
Tobacco	1		1	---
Tomato	2		1	SO_2 (2)
Turnip	1		4	---
Wheat	4		4	SO_2 (1)
Total (17)	38	3	41	Moisture (14), SO_2 (10)

[a]Cultivars were exposed to 4 to 6 ozone concentrations in each study.

that could have affected crop yield response to ozone for which data were not gathered in the study included susceptibility to pests (disease and insect) and other stressors.

The U.S. Department of Agriculture (USDA) crop inventory is done on a regular basis, and the procedures are well established (USDA, 1981; Heck et al., 1983). The program did not collect these data but used the data available from the crop inventory to estimate national yield losses (USDA, 1981).

Comments on Problem Formulation

Strengths of the case study include:

- *All reviewers felt that the problem formulation was thorough and well supported.*

Limitations include:

- *Agroecosystems are simple in comparison with natural ecosystems. Therefore, caution should be exercised in extrapolating from this assessment to natural plant communities.*

6.3.2. Analysis: Characterization of Ecological Effects

Design of Field Studies on Effects of Ozone. EPA decided that long-term field studies should be initiated to determine the impact of ozone on the yield of major agronomic crops. Although the focus of NCLAN on agronomic crops restricted the understanding of effects on other important plant species, it was decided that, for the funds available, an economic assessment would give the most useful data to EPA for the next review of the ozone standard. Thus, EPA initiated NCLAN in 1980 (Heck et al., 1982). Although the field test sites selected provided a limited basis for a national assessment, the sites did include six major agricultural regions in the country with varying soils and climatic conditions. Despite these variations, the study was able to compare relative yield losses across regions, suggesting that the results obtained gave a reasonable estimate of impact.

Research undertaken between 1958 and 1977 (Heck et al., 1977; U.S. EPA, 1978) using controlled environmental or greenhouse exposure facilities clearly showed the cause-effect relationship between ozone and injury or damage to plant species. Effects studied included foliar injury, growth and reproduction, yield, and physiological responses. During this time, selected field studies were undertaken, but only a few of these used exposure concentration-plant response designs that would permit the development of response functions (Heagle and Heck, 1980; U.S. EPA, 1986). These functions are necessary for extrapolation to other locations. These studies clearly showed that the ambient concentrations of ozone present during the growing season were sufficient to cause yield reductions in wheat, corn, soybean, and peanut.

The NCLAN experimental designs used a charcoal-filtered air treatment (ca. 0.025 ppm seasonal 7-hr/day ozone concentration) as the lowest experimental concentration. This treatment was assumed to represent an overall natural (not related to human activity) background ozone concentration for the country. The second treatment was a nonfiltered (NF) air chamber that had a seasonal average ozone concentration slightly below the ambient level. Two to four higher ozone concentration treatment chambers also were used in the experiments for a total of four to six experimental chambers for each replication. Crops were planted following recommended farming practices. After the plants were several inches tall, field plots of uniform plant material were identified, and the plots were covered with open-top chambers to control the gaseous environment around the plants (figure 6-2). Experiments were designed to expose plants to a series of ozone concentrations so that ozone exposure concentration-crop yield response relationships could be determined. Growth, yield, and physiological parameters also were measured on the agronomic crops of primary economic concern. The primary impacts of interest were crop yield on a regional and national basis and a translation of the yield changes to economic values (Heck et al., 1982). Additional experimental detail can be found in Heck et al. (1982, 1983, 1984a,b, 1991).

Exposure Concentration-Plant Yield Response Functions. The NCLAN program was specifically designed to permit the development of ozone exposure concentration-plant yield response functions. A number of linear and nonlinear functions were tested in this program (Heck et al., 1982, 1983). The Weibull nonlinear function was finally chosen for several reasons: (1) it has a flexible form that covers the range of responses observed; (2) its form is biologically realistic; (3) its parameters have straightforward interpretations; (4) it provides direct estimates of proportional yields; (5) tests of homogeneity of proportional yield responses over data sets are readily accomplished; and (6) where homogeneity is found, the common proportional response models can be used to represent the response of the crop as a species.

The Weibull model is given as:

$$Y = \alpha \exp [-(x/\omega)\lambda]+\varepsilon$$

where Y is the yield and x is the ozone dose. The three parameters to be estimated are α, the estimated yield at zero ozone concentration; ω, the ozone concentration when yield is 0.37α; and λ, a dimensionless shape parameter ($\lambda=1$ gives the exponential loss function, whereas a larger λ [e.g., 4.5] gives a region of almost no loss [a threshold] before the curve starts to drop). The ε is the random error associated with each observation.

Individual and combined response functions were developed in three categories (Somerville et al., 1989; Lesser et al., 1990):

- Category I—Response functions from the individual experimental designs;
- Category II—A combination of individual experimental designs within a species. Here the functions showed homogeneity and a homogeneous response function could be developed; and

Figure 6-2. Field exposure design (Miller et al., 1989)

- Category III—A combination of all experimental designs for a given species where all functions did not show homogeneity; thus, a heterogeneous response function was generated. The response functions (across sites and years) for all cultivars within a species were tested for homogeneity and the homogeneous functions developed were used to predict relative yield loss for specific cultivar groupings within species (Somerville et al., 1989; Lesser et al., 1990).

Table 6-2 presents estimated relative yield losses in percentage for selected Category I and II models for most of the crops tested in the NCLAN program (Heck et al., 1991). Relative yield values for several Category I models (peanut, tobacco, sorghum, a clover/fescue forage) and several Category II models (cotton and soybean) are shown in figures 6-3 to 6-8 (data are from Somerville et al., 1989).

Category III response equations are summarized in Somerville et al. (1989) and Lesser et al. (1990). A Category III response curve for each species is the best-fitting common Weibull response curve where the model includes the α-terms to account for all sites, year, block, cultivar, linear SO_2, and linear moisture effects. Thus, each equation should be regarded as an approximation of the average response of the crop. Category III models were used for the crops included in the economic assessment. Figure 6-9 contains a heterogeneous Weibull response function for corn, cotton, and soybean using a 12-hr/day seasonal mean value for ozone (Heck et al., 1991).

Wald confidence interval estimates, at the 95 percent level of predicted relative yield losses (RYL), are based on the Category I and II models (Somerville et al., 1989, 1990; Lesser et al., 1990). Selected examples are shown in table 6-3. Somerville et al. (1990) compared the classical confidence intervals, obtained from first-order linear approximation theory (Wald estimates) for the NCLAN designs, with the more theoretically correct, but difficult to compute, interval estimates based on the likelihood ratio test (LRT). Nine Weibull models from nine NCLAN studies were used to compare the Wald and LRT confidence interval estimates of relative yield loss due to ozone. Results for sorghum and peanut are shown in figures 6-10 and 6-11.

From earlier discussions and NCLAN results, it is clear that crop species, cultivar, crop growth stage, environmental conditions, and the presence or absence of insect pests and disease organisms could affect the response of the crop species to ozone (U.S. EPA, 1986; Heck et al., 1986, 1988). The interaction with soil moisture was shown in several experimental designs and is considered the most important environmental variable that might affect the response of the crop species to ozone. Results from the ozone by soil moisture designs were used to develop a model (King, 1988) that was used in the economic assessment to adjust for the soil moisture stress.

Table 6-2. Estimated Relative Yield Losses (Percent) at Four Seasonal (7-hr or 12-hr/day) Mean Ozone Concentrations Using Homogeneous Weibull Models[a]

Crop (Model)[b]	Coefficient of Variation[c]	Ozone Concentration (ppm)			
		0.04	0.05	0.06	0.07
		7-hr/day seasonal means			
Bean, kidney (2)	15.5	4.3	8.9	14.9	22.3
Lettuce (1)	28.2	0.0	0.0	0.1	0.5
Peanut (1)	7.3	6.5	12.5	19.8	27.9
Sorghum (1)	5.1	0.9	1.7	2.7	3.9
Tomato (1)	11.8-12.3	3.5	6.3	9.5	12.9
Turnip (1)	33.6	7.2	14.9	19.5	35.7
Wheat (2)	10.9	2.8	5.8	9.5	14.3
		12-hr/day seasonal means			
Alfalfa (1)	7.6-8.3	3.8	7.1	10.7	14.6
Corn (1)	9.9	1.2	3.3	7.3	13.9
Cotton (3)	6.7-17.8	6.0	14.0	26.0	41.1
Forage (2)	5.6-12.1	3.8	7.7	12.5	18.2
Soybean (1)	6.6-19.8	8.0	15.3	23.8	33.0
Soybean (3)	4.1-18.0	12.3	21.5	31.0	40.2
Tobacco (1)	5.3	6.2	11.1	16.4	21.8

[a]The predicted relative yield losses come from table 13 in Somerville et al. (1989) using the homogeneous models from Lesser et al. (1990). Values shown are mean values from the 95 percent confidence limits table. Yield losses are calculated relative to yield at a seasonal O_3 mean of 0.025 ppm. Data points were adjusted to remove all fixed effects except the effects of O_3.

[b]Homogeneous model numbers refer to models from table 1 in Lesser et al. (1990).

[c]The coefficient of variation came from Heagle et al. (1988) and is shown for the study or studies from which the modeled data were obtained.

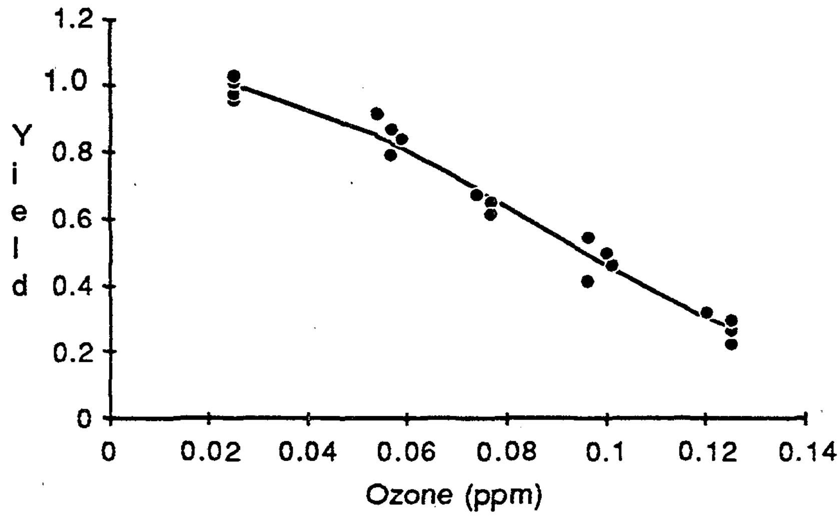

Figure 6-3. Relative yield response of peanut to O_3 using a 7-hr/day seasonal mean O_3 = 0.025 ppm (Somerville et al., 1989)

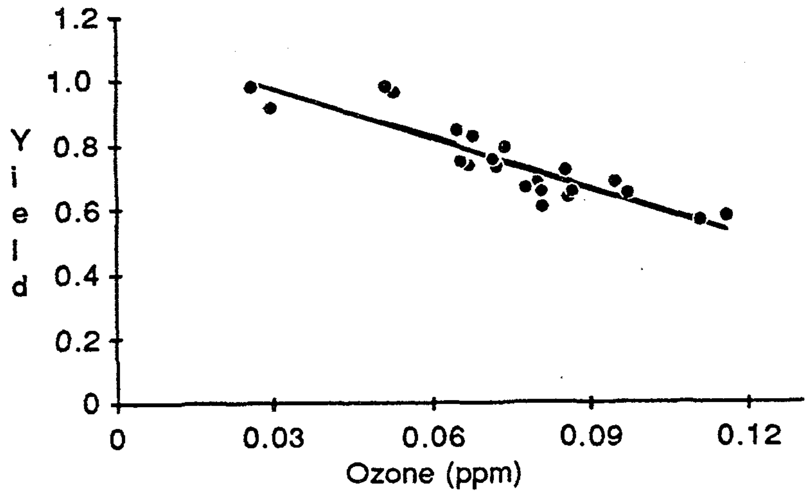

Figure 6-4. Relative yield response of tobacco to O_3 value. The relative yield value is scaled to show yield relative to yield at O_3 = 0.025 ppm (Somerville et al., 1989)

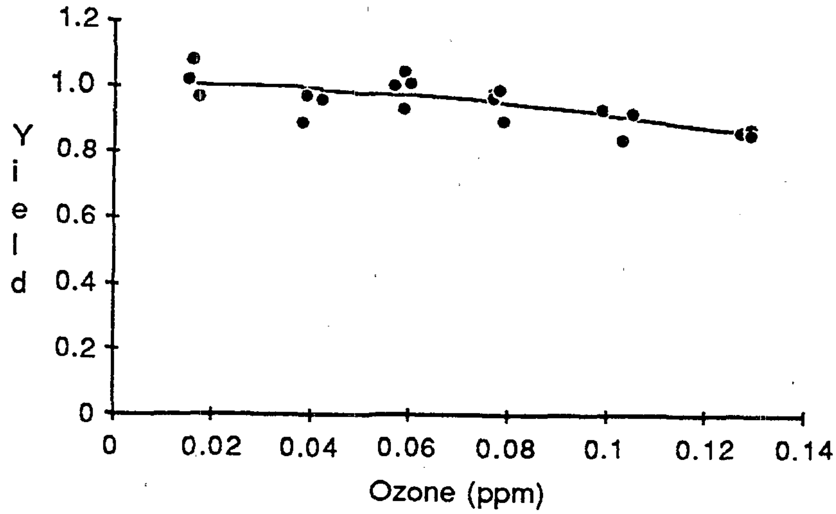

Figure 6-5. Relative yield response of sorghum to O_3 using a 7-hr/day seasonal mean O_3 value. The relative yield value is scaled to show relative yield at O_3 = 0.025 ppm (Somerville et al., 1989).

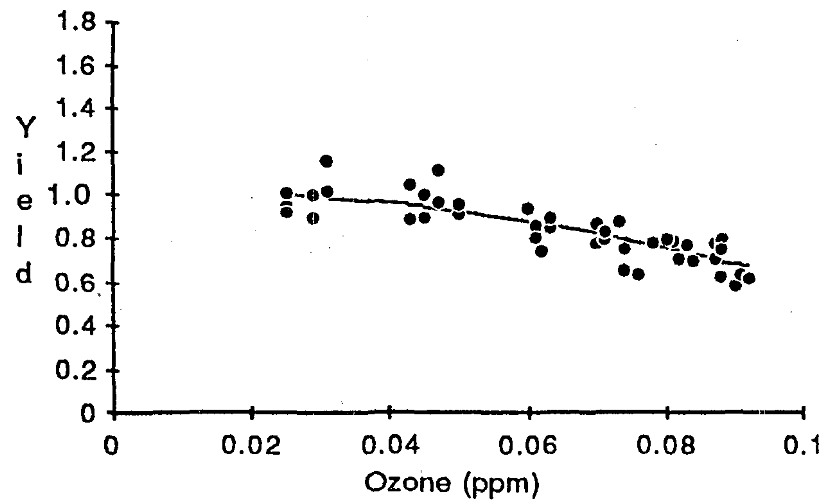

Figure 6-6. Relative yield from two clover-fescue studies (Forage Model 223) to O_3 using a 10-hr/day seasonal mean O_3 value. The relative yield value is scaled to show yield in relation to yield at O_3 = 0.025 ppm. Data points were adjusted to remove all fixed effects except the effects of O_3 (Somerville et al., 1989).

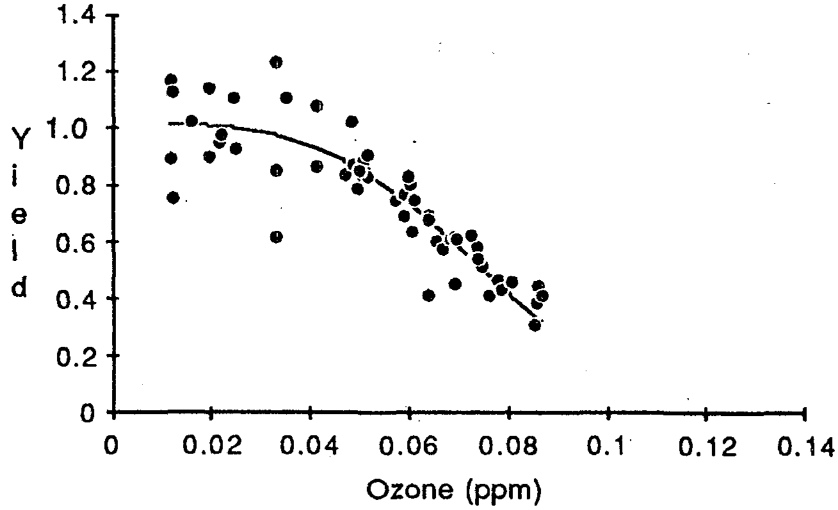

Figure 6-7. Relative yield response from three cotton studies (Model 323) to O_3 using a 12-hr/day seasonal mean O_3 value. The relative yield value is scaled to show yield relative to yield at O_3 = 0.025 ppm. Data points were adjusted to remove all fixed effects except the effects of O_3 (Somerville et al., 1989).

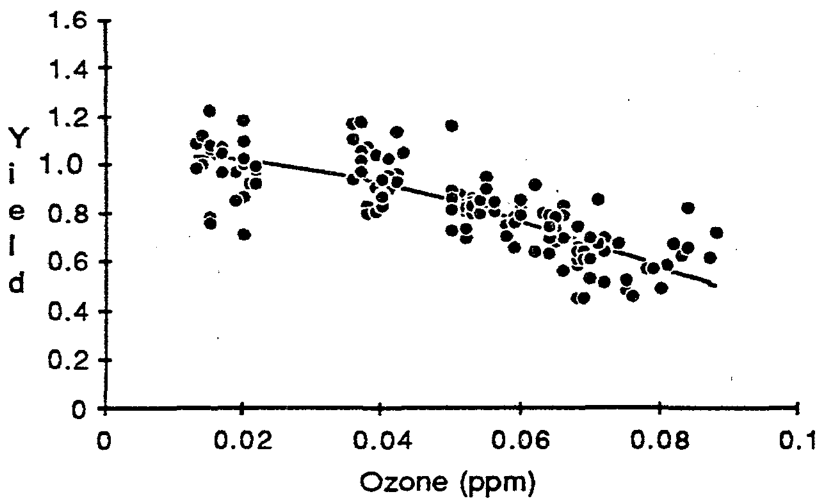

Figure 6-8. Relative yield response from six soybean studies (Model 123) to O_3 using a 12-hr/day seasonal mean O_3 value. The relative yield value is scaled to show yield relative to yield at O_3 = 0.025 ppm. Data points were adjusted to remove all fixed effects except the effects of O_3 (Somerville et al., 1989).

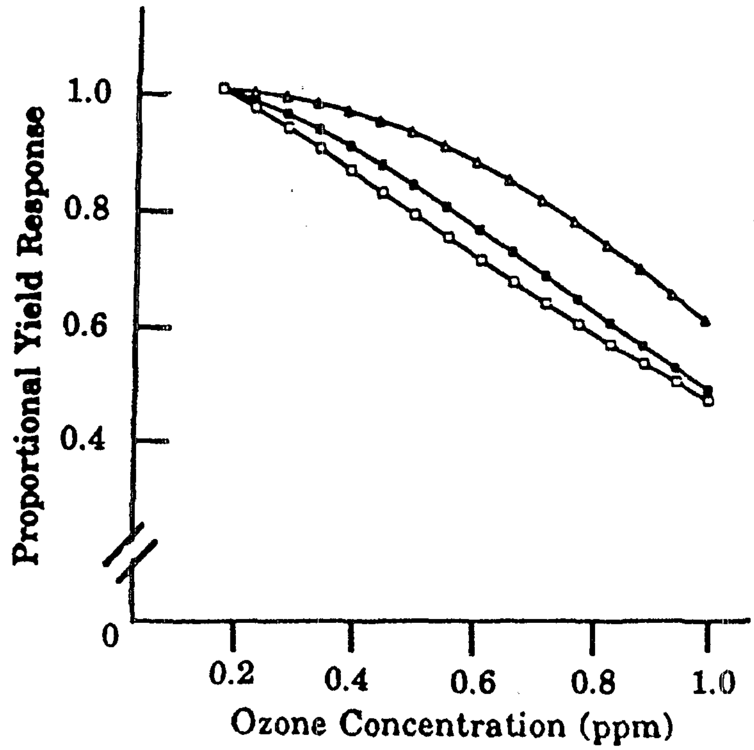

Figure 6-9. Category III models (heterogeneous) for all data from each of three species [corn (Δ), cotton (■) and soybean (□)]. Relative yield response to O_3 using a 12-hr/day seasonal mean O_3 value. The relative yield value is scaled to show yield relative to yield at O_3 = 0.025 ppm. Data points were adjusted to remove all fixed effects except the effects of O_3 (Somerville et al., 1990).

Table 6-3. Wald Confidence Interval Estimates (95 Percent) of Percentage Yield Loss From Ozone Relative to Yield at O_3 = 0.025 ppm Based on Category I or II Weibull Response Equations

Crop (Model)[a]	Estimated Relative Yield Losses (Percent)[b] Ozone Concentration (ppm) 0.03	0.05	0.07
Alfalfa (1)	(0.4, 1.9)	(4.3, 9.8)	(11.2, 17.9)
Corn (1)	(0, 0.4)	(1.0, 5.6)	(3.5, 11.1)
Cotton (3)	(0.6, 2.0)	(10.2, 17.8)	(37.3, 44.8)
Forage (2)	(-0.2, 2.3)	(2.3, 13.0)	(12.2, 24.1)
Peanut (1)[c]	(0.9, 2.6)	(8.7, 16.3)	(23.0, 32.8)
Sorghum (1)[c]	(-0.5, 1.0)	(-1.7, 5.1)	(-1.5, 9.3)
Soybean (1)	(1.3, 3.2)	(12.0, 18.6)	(29.7, 36.3)
Tobacco (1)	(0, 3.8)	(4.2, 18.0)	(13.6, 30.0)
Wheat (2)[c]	(0.1, 1.4)	(2.3, 9.2)	(8.9, 19.6)

[a]Model numbers refer to models from table 1 in Lesser et al. (1990).
[b]The estimated relative yield losses come from table 13 in Somerville et al. (1989).
[c]7-hr/day seasonal O_3 means were used in these three species; all other species used a 12-hr/day seasonal O_3 mean concentration.

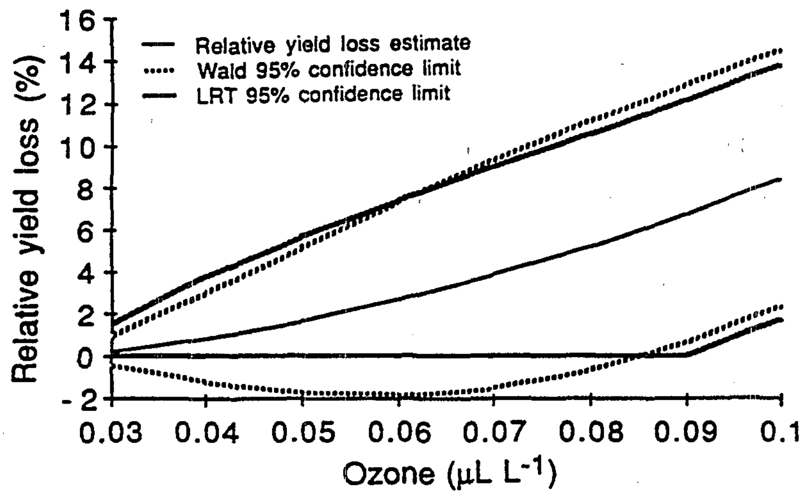

Figure 6-10. **Wald and LRT 95 percent confidence interval estimates of RYL for sorghum using the Category I concentration-response equation. Point estimates of RYL are shown with lighter solid line (Somerville et al., 1990).**

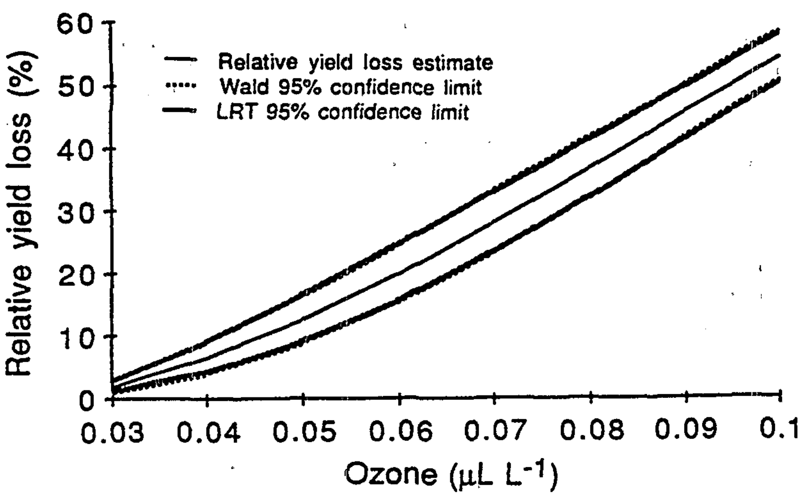

Figure 6-11. **Wald and LRT 95 percent confidence interval estimates of RYL for peanut using the Category I concentration-response equation. Point estimates of RYL are shown with lighter solid line (Somerville et al., 1990).**

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *The uncertainties for the ozone concentration-crop yield response data are clearly represented by the 95 percent confidence limits developed for each homogeneous response function for each crop species.*

6.3.3. Analysis: Characterization of Exposure

The experimental design ensured that ozone was the stressor of concern. Monitoring data from around the United States over a number of years showed that ozone was present throughout the growing season at concentrations capable of causing injury to sensitive vegetation (Heck et al., 1983). EPA maintains a data bank of ozone-monitoring data from across the country in its Storage and Retrieval of Aerometric Data (SAROAD) data base. These data, from over 300 sampling sites, were used in an interpolation process (kriging) to calculate the seasonal 7-hr/day ozone concentration on a county basis across the country (Heck et al., 1984a,b; Lefohn et al., 1987). Although the uncertainties associated with the ozone data could be calculated, this was not done. However, many data sets were dropped because of incomplete data. Data kriged to several NCLAN sites compared well with the site data, providing some verification to the kriging model. Variations in the kriged values would affect the final results, but there was no way to verify the model or to establish confidence levels. Figures 6-12 and 6-13 show kriged values for two different years across the United States; the figures are from Knudsen and Lefohn (1988).

Ozone concentrations in the experimental chambers were monitored on a continuous basis during the experimental period (Heck et al., 1982). The data were initially summarized as hourly averages. The experimental ozone data sets were then summarized as the 7-hr or 12-hr seasonal averages (Somerville et al., 1989). These averages were subsequently used with the chamber yield data to generate ozone exposure concentration-crop yield response functions (Heck et al., 1984a,b; Lesser et al., 1990; Somerville et al., 1989).

Over the 7 years of the program, the ambient seasonal 7-hr/day ozone concentrations were between 0.035 and 0.070 ppm both at the experimental sites and from the kriged county-level data. Monitored levels of ambient ozone suggested that all crops might show some indication of response in most areas of the country (Heck et al., 1991). Monitored ambient ozone data had low uncertainties, but these were not calculated.

Figure 6-12. **Location of monitoring sites used to estimate monthly maximum 7-hr O_3 for July 1984 (Knudsen and Lefohn, 1988)**

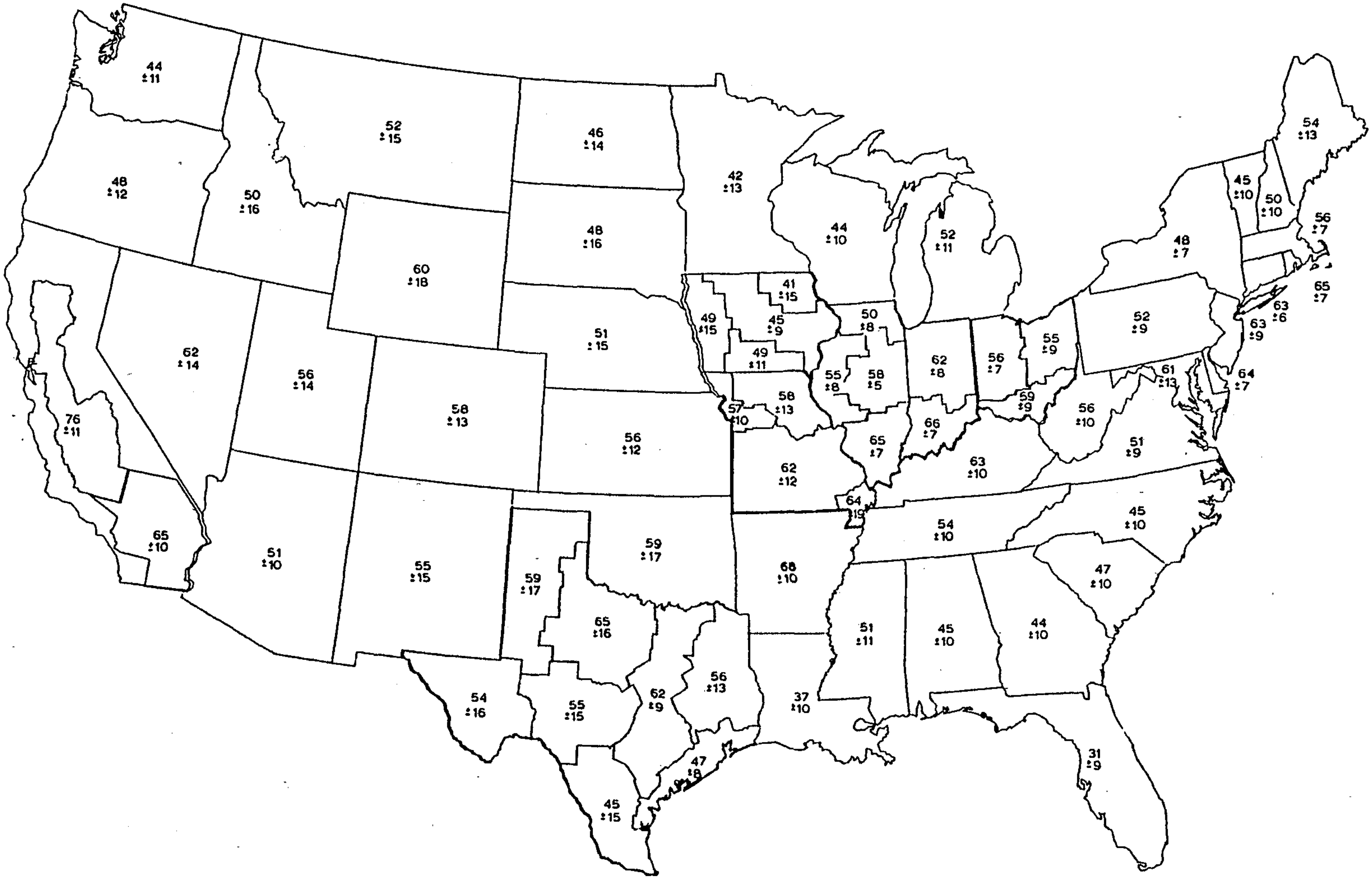

Figure 6-13. Map of kriged estimates (concentrations in ppb) of maximum 7-hr O_3 for July 1984 (Knudsen and Lefohn, 1988)

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *The use of a large data base on ozone measurements as a basis for interpolation demonstrates how large monitoring networks can be employed in a risk assessment.*

Limitations include:

- *Uncertainties associated with the ozone data were not calculated although this could be done.*

6.3.4. Risk Characterization

The ozone exposure concentration-crop yield response functions were used in a predictive fashion to determine the loss of yield in the test species across the country (Somerville et al., 1989; Lesser et al., 1990). The 95 percent confidence limits for homogeneous response functions for each species were calculated so that variation in the predicted losses could be determined (Somerville et al., 1989, 1990; Lesser et al., 1990). These response functions were the most important component in the overall analysis, because they had the greatest impact on the yield assessment endpoint.

The county seasonal ozone values and crop inventories were then used in the response functions to calculate yield losses from ozone on crop species across the country. These yield losses were then used for the national economic assessment. The summary economic assessment is shown in table 6-4 for both the 1984 and 1988 assessments (Adams et al., 1984, 1988, 1989).

A possible technique to map crop losses for cotton on a national basis is shown in figure 6-14. The county seasonal ozone values were used in a composite ozone exposure concentration-crop yield response function for cotton-growing areas in the United States. The mean cotton yield loss was estimated in a number of cotton-growing areas. The loss gradient surface was then developed to represent estimated loss on a geographical basis. Maps such as this could be developed for any of the crops studied in the NCLAN program.

Table 6-4. Comparison of the 1984 and 1988 Economic Surplus Estimates in 1982 Dollars[a]

Ozone Assumptions	Total Surplus ($ millions)
1984 Model	
25% Increase	-2,165
10% Reduction	699
25% Reduction	1,828
40% Reduction	2,637
1988 Model	
25% Increase	-2,053
10% Reduction	808
25% Reduction	1,890
40% Reduction	2,780

[a]From Adams et al. (1988). The ozone assumptions are based on the current ambient seasonal ozone concentrations as determined by the kriging interpolation of the SAROAD data base (Lefohn et al., 1987).

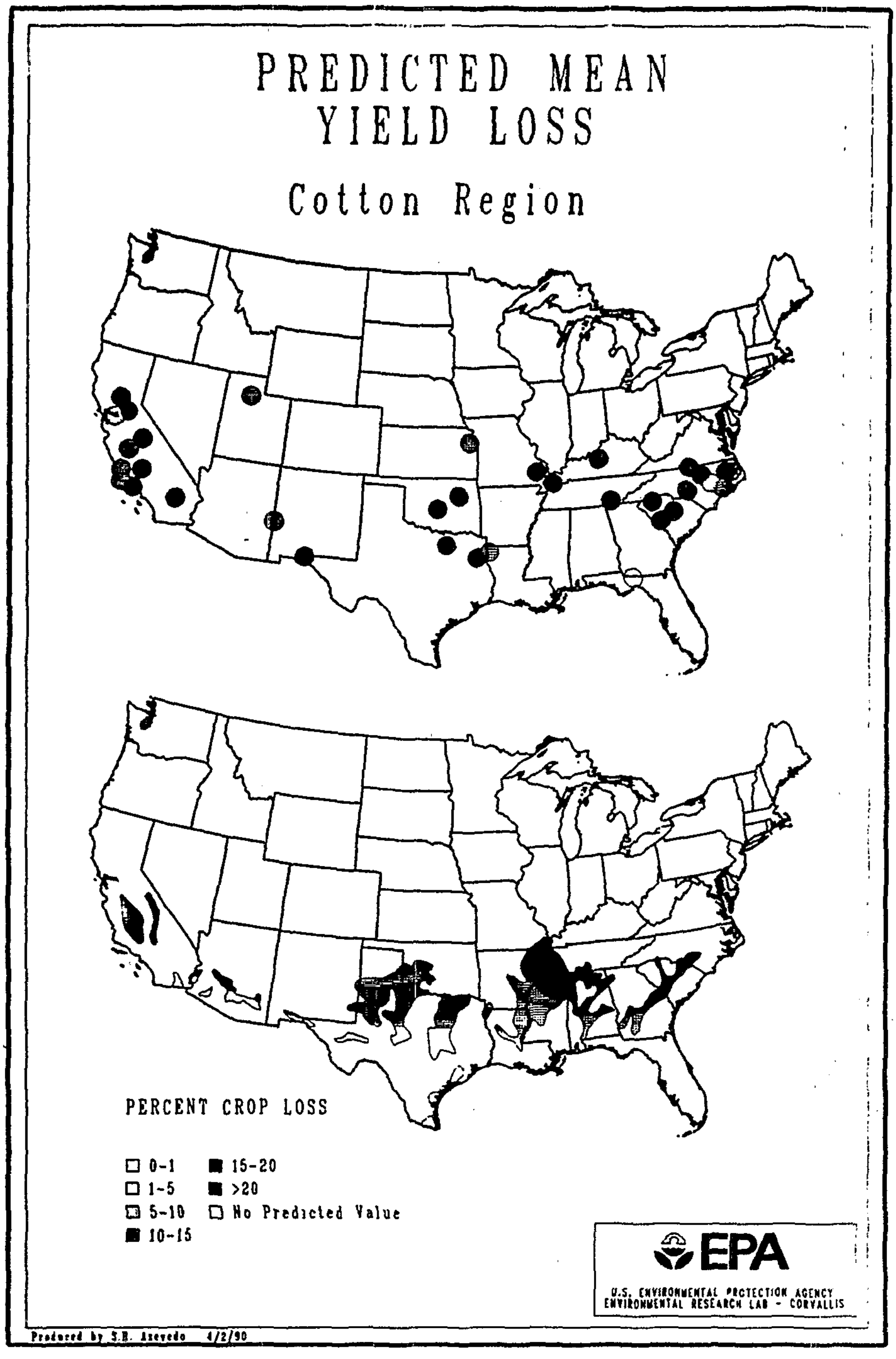

Figure 6-14. **Estimated mean percent loss of cotton yield based on measured O_3 (SUM06) and a composite O_3 exposure-response function. The upper circle map illustrates the mean cotton yield loss at each site; the sites were used to create the gradient surface in the lower map. The maps were prepared by S.H. Azevedo and permission to use them was obtained from D.T. Tingey.**

Comments on Risk Characterization

Strengths of the case study include:

- *The NCLAN data are well documented and are contained in a central data base. All data were verified using extensive cross-checking techniques. All experimental sites developed and maintained a strong quality assurance/quality control (QA/QC) program, and the sites were audited on an annual basis.*

- *The program used yield as both a measurement and an assessment endpoint of primary concern. However, many other measurements were obtained during the program that were used for a number of different purposes. Eventually, the other measurements could be used in model development that could make the predictive capabilities of the functional relationships more accurate. The economic assessment was the assessment endpoint of primary interest in the NCLAN program.*

- *The economic analysis used the experimental crop response data, the kriged county-level ozone values, and the USDA-generated county-level crop inventory data to calculate national yield losses from ozone. The data were then used to calculate economic benefits from different percentage reductions in the seasonal ozone values. The data that went into the economic analyses were carefully developed and confidence levels are known.*

- *Regular and in-depth communications were a primary factor in the success of the NCLAN program. The ability to interpolate national ozone values and to develop homogeneous yield responses across sites, years, and cultivars for a given species was another critical factor. Cooperators met annually for a week-long workshop to develop protocol and adopt consensus approaches to ensure uniformity and comparability of results.*

- *This was a valuable case study that addressed a national problem and, because it was a well-coordinated study using a standardized approach, it will be a useful model for future studies.*

Limitations include:

- *The NCLAN program outlined a number of issues that had not received sufficient attention. These issues are clearly developed in a final NCLAN paper published in an Air and Waste Management Association Transaction in the spring of 1991; areas for new research initiatives are recommended in that same publication (Heck et al., 1991). Several issues are briefly highlighted. The study was limited in size, and only one or two sites were used (total of six) in four regions of the country. The results on the effects of soil moisture on plant yield response were not conclusive and very little was done to determine the effects of multiple abiotic and biotic stressors on crop yield response. Major scientific issues remaining to be addressed, in addition to those*

Comments on Risk Characterization (continued)

mentioned above, include: (1) effects of the field methodology on plant response to ozone; (2) the most applicable concentration statistic to use; (3) the accuracy of the kriging approach; and (4) understanding plant processes in relation to the impact of ozone so that more process-oriented models can be developed.

- *The agricultural production systems under study were much simplified compared with natural ecosystems. Similarly, the final economic analysis represented an additional step that may be inappropriate in many ecological risk assessment case studies.*

General comment:

- *Based on experience with this program, it is recommended that all large-scale programs have regular researcher meetings and interactions. These activities were critical to the success of NCLAN.*

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SECTION SEVEN

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

COMPARATIVE ANALYSIS OF MINING TAILING DISPOSAL FOR THE QUARTZ HILL MOLYBDENUM MINING PROJECT

AUTHORS AND REVIEWERS

AUTHORS

Patricia Cirone
Region 10
U.S. Environmental Protection Agency
Seattle, WA

John R. Yearsley
Region 10
U.S. Environmental Protection Agency
Seattle, WA

REVIEWERS

James J. Anderson (Lead Reviewer)
School of Fisheries Research
University of Washington
Seattle, WA

Jay S. Jacobson
Boyce Thompson Institute
Cornell University
Ithaca, NY

Larry Kapustka
Ecological Planning and Toxicology, Inc.
Corvallis, OR

Wayne G. Landis
Institute of Environmental
Toxicology and Chemistry
Western Washington University
Bellingham, WA

Thomas Sibley
School of Fisheries Research
University of Washington
Seattle, WA

Frieda B. Taub
School of Fisheries Research
University of Washington
Seattle, WA

Donald P. Weston
Horn Point Environmental Laboratory
University of Maryland
Cambridge, MD

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LIST OF ACRONYMS

BPJ	best professional judgment
EC_{50}	effective concentration for 50 percent of organisms tested
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
LC_{50}	lethal concentration to 50 percent of organisms tested
MIBC	methyl isobutyl carbinol
NEPA	National Environmental Policy Act
PAH	polycyclic aromatic hydrocarbon
USDA	U.S. Department of Agriculture
USFS	U.S. Forest Service

ABSTRACT

This case study examines the relative risks of two disposal alternatives for mining tailings. Quartz Hill is a proposed molybdenum mining site located within Misty Fjords National Monument in southeast Alaska in a designated non-wilderness area. The project includes an open pit mine with ore process facilities located nearby. The process wastes (tailings) will be transported from the Tunnel Creek area by pipeline to a submarine, gravity-flow outfall in either of two basins. Two possible waste ore (tailings) disposal sites in nearby fjords (Smeaton Bay and Boca de Quadra) were evaluated for potential environmental impacts. Steady-state diffusion models were used to predict the distribution of tailings composed of metals and solids in the surface waters of two marine fjords. Total recoverable copper, suspended solids, and settled solids were the stressors. Ecological components included a variety of fish and invertebrate populations. Since the impacts were presented in probabilistic terms, a relative comparison was completed of each fjord's spatial and temporal exceedance of water quality degradation criteria and the number of benthic organisms lost per hectare of viable habitat covered by tailings. Exceedance of the water quality criterion for copper and loss of benthic habitat was predicted to be greater for the smaller of the two fjords. However, it is clear that further chemical analytical work or additional site-specific bioassays must be completed to verify the assessment. The finding from this evaluation is that disposal of tailings into Boca de Quadra would be the least environmentally damaging alternative of those considered. The permit to dispose of tailings into Smeaton Bay was denied. The purpose of this ecological risk analysis was to compare risks; therefore, no conclusion was reached regarding whether either risk was acceptable.

7.1. RISK ASSESSMENT APPROACH

A comparative risk assessment for the Quartz Hill molybdenum mining project was completed to resolve concerns about the choice of alternative locations for disposal of tailings produced during the mining process (figure 7-1). Before this assessment was prepared, it was determined that mining was an appropriate activity in this area of Alaska and that open water disposal was an acceptable method of removing waste tailings. The justification for these determinations is presented in the Environmental Impact Statement (EIS) prepared by the U.S. Forest Service (USFS).

This case study incorporates data or information from three levels: (1) experimental results from the revised draft EIS (USDA, 1987); (2) experimental results from other projects or environments with similar characteristics; and (3) professional judgment based upon experience and knowledge of physical, chemical, and biological phenomena.

All stages of the project upon which agreement was reached at earlier phases of the National Environmental Policy Act (NEPA) process are not addressed in this risk assessment. The following factors were excluded because they were not directly related to estimating the risk of adverse effects to aquatic organisms from in-water disposal of tailings:

- terrestrial impacts from construction, transportation, or accessories needed for the operation of the mine, including such things as road construction, energy use, and transport of tailings;
- discharge of marine terminal wastewater (treated sanitary wastes, runoff, and wash water) used in the mine operation exclusive of tailings process water; and
- personnel on site (e.g., impacts associated with sewers, water lines).

In this assessment, an attempt was made to quantify risks from exposure to stressors in either of two basins. However, much of the information that was collected over many years was not considered acceptable for a variety of reasons. The lack of adequate data limited the assessment of effects on the aquatic populations. Because this is a comparative risk assessment, the lack of adequate data did not limit the comparison of risk; it only limited the ability to estimate all the effects that may result from the disposal of tailings.

7.2. STATUTORY AND REGULATORY BACKGROUND

The USFS, U.S. Environmental Protection Agency (EPA), and other federal and state agencies prepared an EIS in accordance with NEPA, evaluating the potential environmental consequences of alternative mine development scenarios. Three alternatives were considered for tailings disposal: one upland and two in open water. During the initial phase of the NEPA process, the upland alternative was eliminated. The remaining two alternatives allow disposal of tailings into either of two fjords: middle-basin Boca de Quadra or Smeaton Bay/Wilson Arm. These two fjords are technically considered inland waters and therefore are not subject to the Ocean Discharge Criteria regulations (403c) of the Clean Water Act. However, EPA determined

Figure 7-1. Structure of Analysis for Comparative Analysis of Mining Tailing Disposal

PROBLEM FORMULATION

Stressors: physical impact of burial; copper in water column.

Ecological Components: benthic invertebrates and fishery resources.

Endpoints: assessment endpoint was habitat loss and potential effects on benthic invertebrate and fish populations. Measurement endpoints included estimated accumulation of tailings on seafloor and water column copper concentration.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
Models were used to predict burial and to estimate the concentration of copper in the water column.	Effects were evaluated with: - water quality criteria - toxicity testing - tissue analysis - recolonization studies.

RISK CHARACTERIZATION

A comparison of the relative risks of disposal was made for two fjords. The assessment was based on comparison of predicted water column copper concentrations with the water quality criterion for copper, the frequency of exceedences and estimates of suspended solids levels, and alteration of benthic habitat.

that for the Quartz Hill project, the Ocean Discharge Criteria provide a useful framework for evaluating the impacts of each of the alternatives and for determining National Pollutant Discharge Elimination System permit conditions. EPA's evaluation entitled *A Best Professional Judgment Evaluation Using the Ocean Discharge Criteria for Mill Tailings Disposal from the Proposed Quartz Hill Molybdenum Mine* is presented as appendix S (U.S. EPA, 1988a) of the revised draft EIS (USDA, 1987). EPA prepared an ecological risk assessment as a supplement to its Best Professional Judgment (BPJ) evaluation (U.S. EPA, 1988b).

7.3. CASE STUDY DESCRIPTION

7.3.1. Problem Formulation

Site Description. Quartz Hill is a proposed molybdenum mining site located within Misty Fjords National Monument in southeast Alaska in a designated non-wilderness area (figure 7-2). The project includes an open-pit mine with ore process facilities located in the nearby Tunnel Creek basin. The molybdenum mine tailings would be transported from the ore processing facility by pipeline to a submarine gravity flow outfall in either of two marine fjords (figure 7-3a, Smeaton Bay/Wilson Arm, and figure 7-3b, Boca de Quadra).

Wilson Arm is a small embayment at the head of Smeaton Bay and is considered a subregion of Smeaton Bay for the purposes of this evaluation. The Smeaton Bay/Wilson Arm fjord extends 20 km from the Wilson River/Blossom River estuary to Behm Canal. An underwater sill separates the 285-m-deep Smeaton Bay basin from deeper waters in Behm Canal (figure 7-4a). Wilson Arm is approximately 160 m deep. Discharge of mine tailings to Smeaton Bay/Wilson Arm is assumed to be located 1.1 km down-fjord from the Wilson River/Blossom River mudflat at a depth of 45 m. The zone of active deposition is assumed to be the bottom of the fjord, which is initially approximately 150 m deep. As the fjord fills, the zone of deposition will decrease until at 55 years it is approximately 75 m deep. The discharge of tailings to Wilson Arm eventually would affect all of Smeaton Bay; thus, tailings discharge to Wilson Arm is considered synonymous with discharge to Smeaton Bay.

Boca de Quadra is a fjord that extends from the Keta River estuary westward approximately 57 km to the Revillagigedo Channel. Underwater sills divide the fjord into inner, middle, and outer basins approximately 8, 27, and 33 km long and 170, 400, and 375 m deep, respectively (figure 7-4b). The tailings outfall for middle-basin Boca de Quadra will be at 45 m depth at a distance of 6.7 km down-fjord from the mudflat at the mouth of the Keta River. For purposes of this analysis, it is assumed that the zone of active deposition will be the bottom of the fjord, which is approximately 150 m deep. Detailed site descriptions are included in USDA (1987) and in U.S. EPA (1988a).

Stressors. Mine tailings discharge will average 36,290 metric tons/day (mt/d) for the first 4 to 6 years, and approximately 72,570 mt/d for the remaining 49 to 51 years of project life. This presents approximately 99 percent of the mine materials or approximately 0.84 billion m^3. To obtain the molybdenum ore, the host rock must be crushed, the ore particles physically separated using a flotation process, and the ore concentrated and sent to a processing plant outside the project

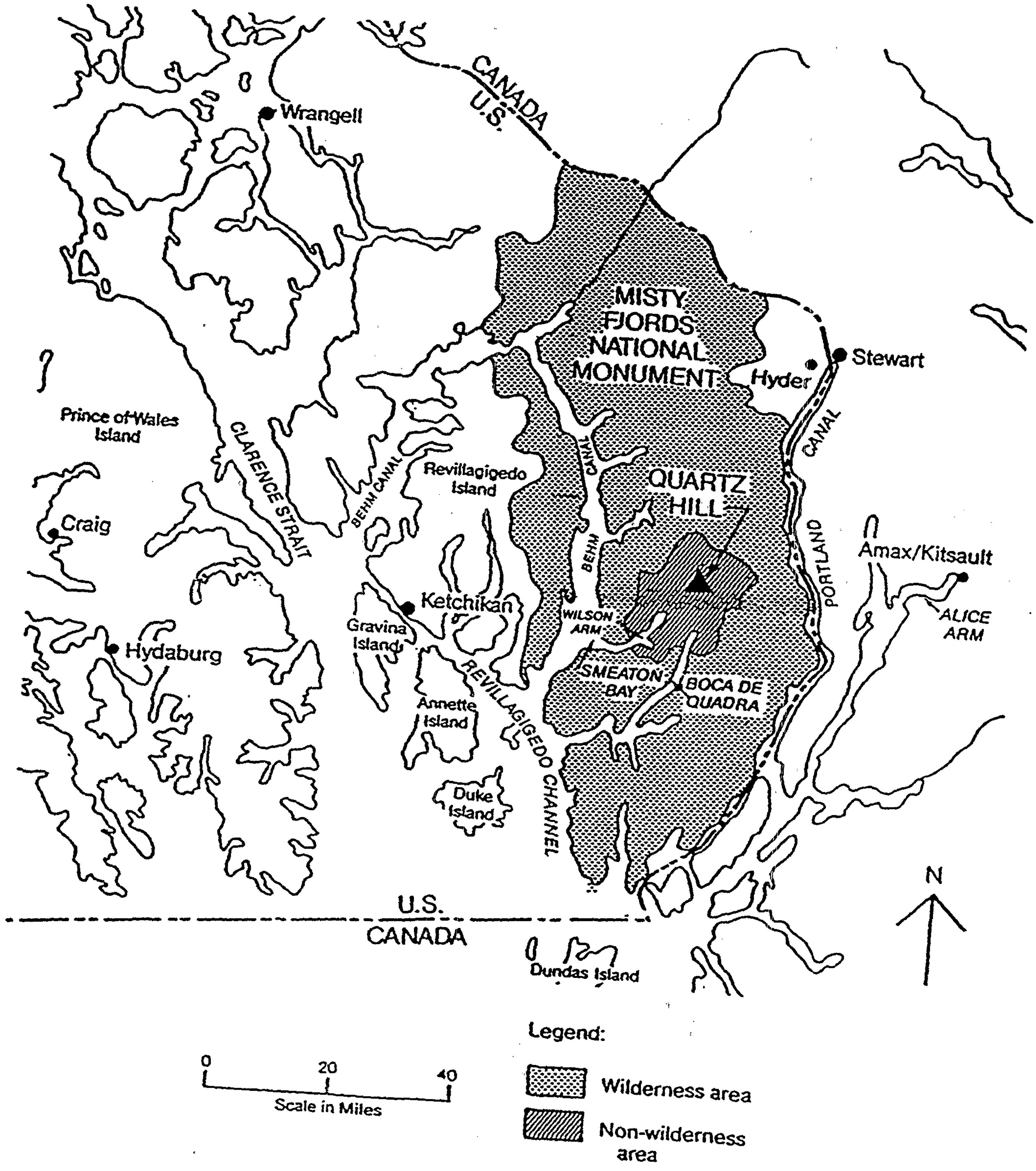


Figure 7-2. Location of Quartz Hill project (USDA, 1987)

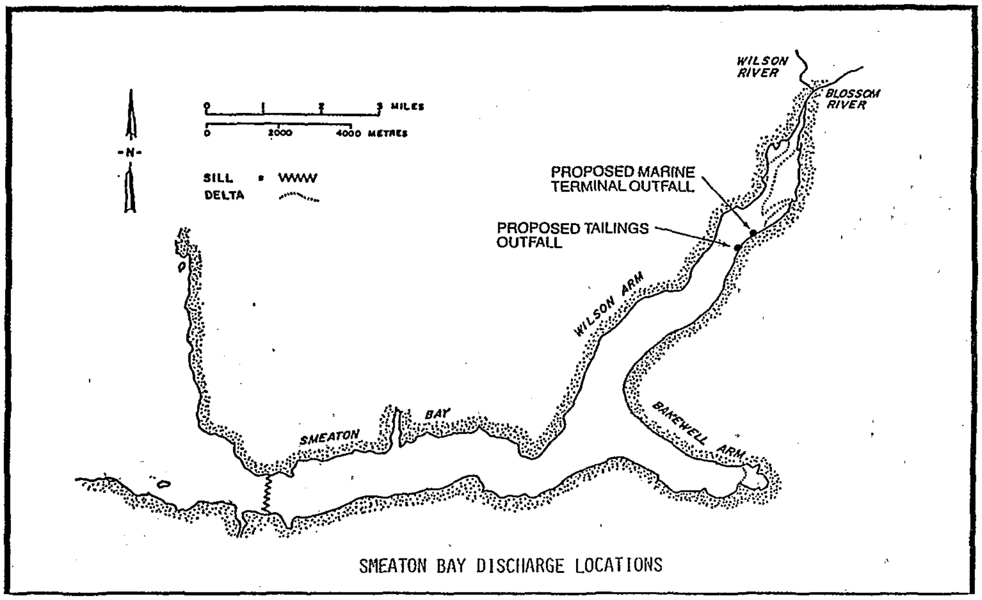


Figure 7-3a. Map of Smeaton Bay alternative for tailing disposal from the proposed Quartz Hill mining project, Alaska (USDA, 1987)

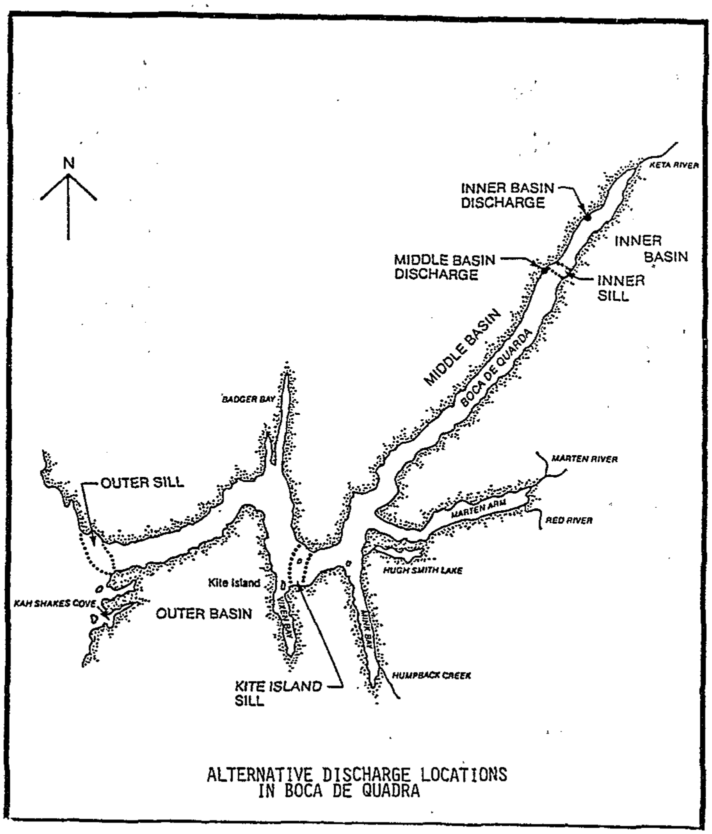


Figure 7-3b. Map of Boca de Quadra alternative for tailing disposal (USDA, 1987)

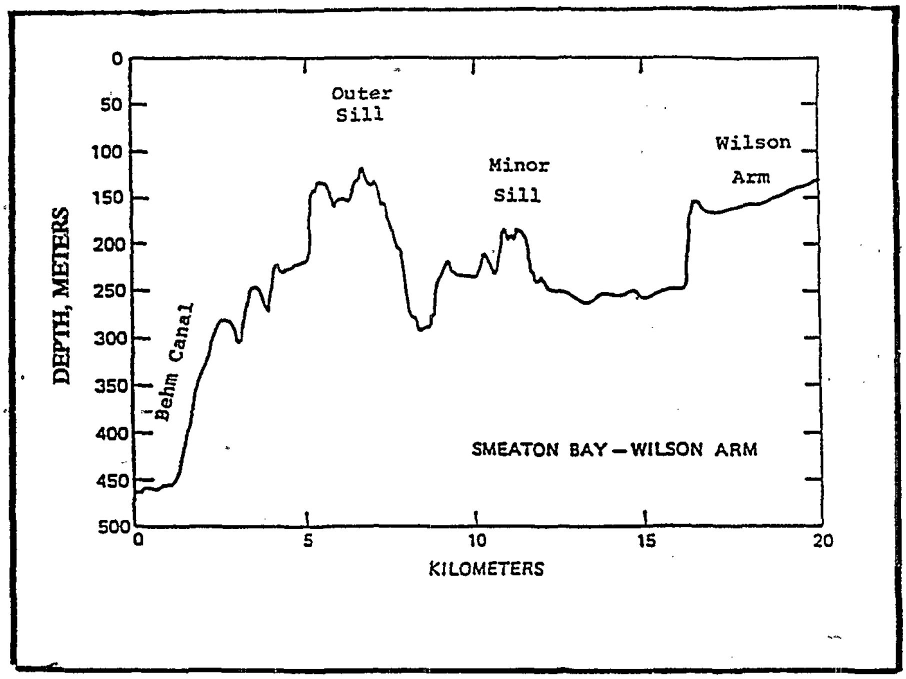


Figure 7-4a. Smeaton Bay longitudinal bathymetric section (USDA, 1987)

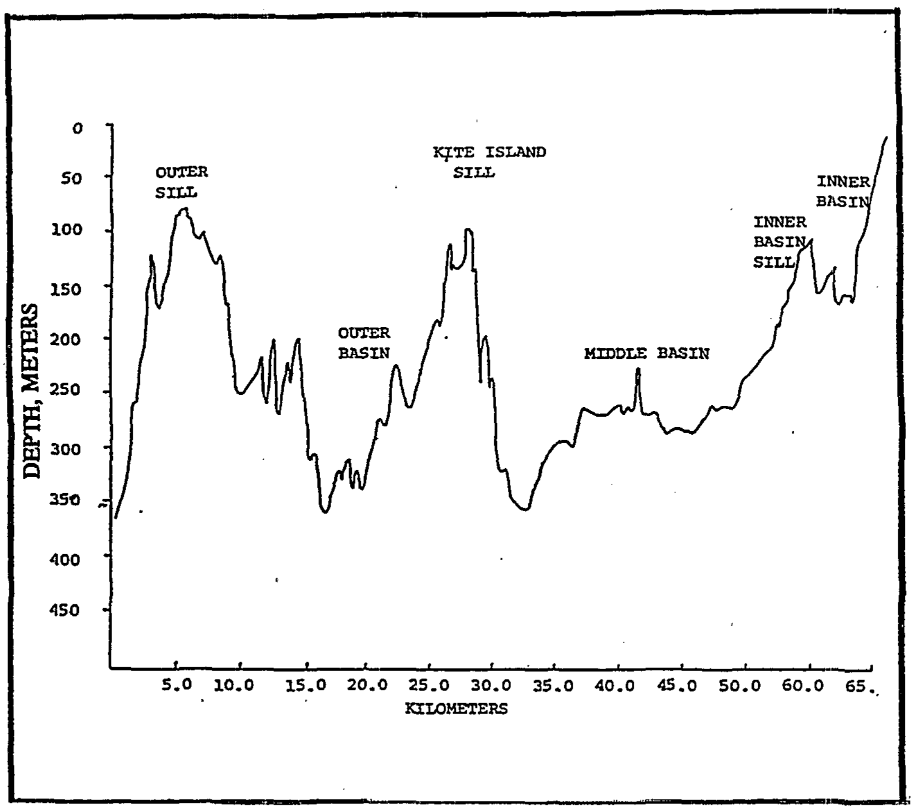


Figure 7-4b. Boca de Quadra longitudinal bathymetric section (USDA, 1987)

area. The mill tailings effluent will be composed of waste rock particles (median grain size 63 microns), freshwater, seawater, and residual milling chemicals.

The 72,570 mt/d of solids will be mixed with 98,000 tons of seawater (1:1 by weight) prior to discharge to the fjord. The total mill tailings discharge rate will be 1.35 m^3/sec prior to predilution. For comparison, the mean annual discharge of the Keta River to Boca de Quadra is 23 m^3/sec; the combined discharge of the Blossom and Wilson Rivers to Wilson Arm is 53 m^3/sec. A number of types of stressors are present, including metals and organic compounds. A discussion of each type of stressor is provided below.

Metals. Approximately 94 percent of the waste rock particles will be quartz and feldspar minerals. Minerals include antimony, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, vanadium, and zinc (table 7-1).

Two chemical fractions that remain after the ore is processed were selected for this evaluation: (1) the dissolved metal fraction and (2) the particulate extractable fraction. The dissolved metal fraction is that concentration of metal that is measured in the water after mixing ore particles with process water and seawater. Concentrations of dissolved metals in the tailings are expected to be at least one to two orders of magnitude higher than concentrations now observed in either Boca de Quadra or Smeaton Bay. The particulate extractable fraction is the concentration of metal that is leached during an acidification procedure described as a total recoverable method in *Ambient Water Quality Criteria for Copper* (U.S. EPA, 1985). The extractable fraction is an estimate of what additional leaching of dissolved metal may occur after the tailings are released into the fjord. The dissolved and extractable fractions are important in assessing the risks from metal toxicity.

Of the metals identified in the ore and tailings, silver, copper, and mercury are the most toxic metals. Silver, copper, and lead were found in the highest concentrations. The toxicological characteristics of the metals are described in *Ambient Water Quality Criteria for Copper* (U.S. EPA, 1985). Based on the water quality criteria (table 7-1) and concentration in the tailings, copper was selected as the metal that would pose the highest risk to aquatic organisms. In addition, high copper concentrations were observed in the water at another mining operation in British Columbia (Island Copper in U.S. EPA, 1988a).

Reagents. A number of reagents must be added to the milling process to control the separation of the molybdenite from the waste minerals and diesel fuel. Ranked in order of quantity used, the reagents are diesel fuel, M-502, methyl isobutyl carbinol (MIBC), lime, sodium silicate, Nokes reagent, and (tied for seventh place) CMC-7 and ALFOL-6. Each reagent (table 7-2) has been arrayed from two standpoints: (1) predicted mass loading and (2) probable aquatic toxicity.

Because of many confounding factors (often no information on persistence or degradation; mixtures of multiple chemicals in the final tailing product; occasional presence of more than one chemical in a given reagent, especially in diesel oil; little or no information on aquatic toxicity, etc.), all assumptions are intentionally biased toward conservatism. The concentrations and toxicities assumed to be present in the tailings are thus more likely to represent the "worst case,"

Table 7-1. **Ore and Tailings Inorganic Content and Associated Toxicity (U.S. EPA, 1988b)**

	Ore (mg/kg)	Solid Tailings (mg/kg)	Liquid[a] Tailings (μg/L)	EPA Acute Water Quality Criteria (μg/L)
Antimony		0.002		
Arsenic		10.9	6.8	36.0
Cadmium		2.4	15.0	9.3
Chromium		10.0	34.0	50.0
Cobalt		3.3		
Copper	90	69.0	35.0	2.9
Iron		16,900	1,790	
Lead	60	47.0	120.0	5.6
Manganese		462.0		
Mercury		0.05	1.2	0.025
Molybdenum	2,170	120	1,080	
Nickel		17.7	290	2.9
Selenium		0.1	6.6	54
Silver		0.13	7.0	2.3
Vanadium		17.6		
Zinc	40	46.0	77	86.0

[a]Effluent concentration (USDA, 1987; appendix F, table F-2).

Table 7-2. Reagents Used in Quartz Hill Mining Project (U.S. EPA, 1988a)

Reagent	Use per 80,000 Tons/Day (lb/day)	Application per Ton Ore (lb)	Approximate LC_{50} (mg/L)
Diesel #2 fuel oil	50,720	0.634	0.1-5.0[a]
M-502	15,920	0.199	1.0[b]
MIBC	12,800	0.160	1.0[b]
Lime	10,720	0.134	1.0[b]
Sodium silicate	5,040	0.063	5.0[b]
Nokes reagent	4,320	0.054	0.002[c]
CMC-7	3,600	0.045	5.0[b]
ALFOL-6	3,600	0.045	5.0[b]

[a] LC_{50} from literature values.
[b] No information on aquatic toxicity in literature; LD_{50} is assumed based on conservative assumptions.
[c] Assumes Nokes reagent disassociates to H_2S.

rather than the actual concentrations. Reagents used in the Quartz Hill mining project are described below.

Diesel fuel is the most commonly utilized reagent at the site. The diesel is used as a collector of molybdenum, which serves to make the molybdenite particles more hydrophobic and thus more floatable. Multiple ingredients in the diesel oil, such as polycyclic aromatic hydrocarbons (PAHs) and other organics, and the differing solubilities and acute and chronic toxicities of these ingredients make it difficult to estimate the effects that may occur when aquatic organisms are exposed during discharge of the tailings. Also, most of the relatively sparse aquatic toxicity data on diesel fuel deal with the water-soluble fractions having lower molecular weights. One is therefore forced to assume a duality of sorts with diesel, in that some of the material will be associated with the water column, while other portions will seek out a less hydrophilic environment (e.g., the marine microlayer, lipids in the biota). The literature indicates LC_{50} ranges from 0.01 to 5 mg/L for aquatic toxicity.

M-502 is the trade name for a cationic, quaternary ammonium salt polymer that is used as a flocculent, to facilitate the settling out of solids in the separation process. Its predicted daily use is 7,221 kg. Only 10 percent of the M-502 flocculent is predicted to be lost to the environment daily, due to a great affinity for the clay portion of the tailings. There is no specific information on aquatic toxicity of this material. However, quaternary ammonium compounds are usually highly reactive with tissue and can usually be irritants in a general sense. They also could have other effects on water chemistry, such as buffering, alterations of pH, and so forth.

MIBC may also be used as a flocculent in the ore separation process. Most of the material is predicted to remain with the liquid phase. Moreover, because of its high vapor pressure, MIBC is predicted to have a high loss rate via volatilization to the air as ketone (assuming this is in the form of methyl isobutyl ketone or MIBC). Because of the many uncertainties (e.g., whether cold water temperatures negate some of the volatile tendencies of the material), and because of its relatively high predicted use, the loading will be greater than the 10 percent that is predicted. Nothing could be found in the available literature regarding aquatic toxicity of this material.

Lime is calcium oxide and is to be used at a daily predicted rate of 4,863 kg/day. It is used as a pH modifier in the separation process. According to the literature, increasing the pH of the solution to 8.7 makes iron sulfides more floatable, aiding in the separation and purification of the molybdenum. Calcium oxide is predicted to remain with the liquid portion of the process. The portions released to the environment would be expected to be readily buffered by the receiving marine waters. The aquatic toxicity of calcium oxide is probably more indirectly related to its effect on pH than to other factors.

Sodium silicate is also known as "water glass" and is composed of a complex of SiO_2 with Na_2O. It is used as a flotation regulator and as a gangue (slime) depressant, acting to depress slime via electrostatic charge. At least 50 percent of the material can be expected to remain with the tailings. Information on the aquatic toxicity of sodium silicate is not available.

Nokes reagent consists of 43.4 percent phosphorus pentasulfide and 56.6 percent sodium hydroxide. It is extremely toxic and irritating. Nokes reagent is used in the milling process as a flotation regulator by generating sulfhydryl ion (SH^-), which acts to depress unwanted metals. The ability of this material to generate SH^- is central to its toxicity. The toxicity of the material, as SH^-, is assumed to be very significant. In the absence of other data on Nokes reagent per se, the EPA water quality criterion for sulfide/hydrogen sulfide is used for comparison purposes, assuming that the SH^- generated by the process will follow equilibria in water similar to the known dissociation reactions for H_2S-HS^- set forth in the development of the EPA criterion. The criterion is 2 μg/L (or 0.002 mg/L) as undissociated H_2S. If the concentration exceeds this criterion, there may be a resultant toxic impact.

CMC-7 is used as a gangue depressant and flotation regulator (as with sodium silicate mentioned previously). It is composed of sodium carboxyl methyl cellulose. At least 50 percent of CMC-7 will be associated with the tailings. Information on the degradation and toxicity of the material is not available.

ALFOL-6 is 1-hexanol, or 1-tetradecanol. It is expected to have a long life in the liquid phase. ALFOL-6 is intended for use as a substitute flocculent (possibly in lieu of MIBC discussed earlier), and its alcohol structure suggests that it would tend to have a reasonable affinity for water. The aquatic toxicity of ALFOL-6 is unknown.

Chemical analyses of tailings prior to dilution with seawater detected no priority pollutants above 12 μg/L (U.S. EPA, 1988a). This detection limit is higher than the toxicity levels for most chemicals of concern. Also, no reagent standards were analyzed. Thus, the evidence suggesting that reagents will not be present in the effluent is not substantiated by the preliminary analysis. A worst-case analysis would suggest that the chemicals may be present at toxic concentrations. Further analyses with lower detection limits and chemicals outside the priority pollutant category are needed to reduce the uncertainty. A detailed discussion of the potential effects of reagents is given on pages 17 to 24 of EPA's BPJ report (1988a). While it is clear that exposure to reagents may be stressful to the aquatic populations, these chemicals were not included in the quantitative risk assessment. Since this is a comparative risk assessment, an estimation of absolute risks due to exposure from all stressors was not deemed necessary. However, the limited information on toxicity of the reagents is a factor that should be included as part of the biological tests that must be completed in order to fully understand the effects of tailings disposal on these aquatic ecosystems.

Settled solids. Settled solids are assumed to be that fraction of tailings that is greater than 10 microns in diameter and settles according to theoretical predictions (U.S. EPA, 1988a). Settled solids are potentially harmful to the benthic biota of the fjords because they may smother or bury resident populations and their habitats and prevent community development. Chemical changes in the sediments may also result in long-term leaching of contaminants from the tailings.

Suspended solids. The suspended solids portion of tailings is assumed to be the 10 percent fraction with the smallest diameter (median = 5 μm) that is injected into the water column along the axis of each fjord (U.S. EPA, 1988a). The suspended solids present a potential harm to pelagic organisms due to their interference with normal metabolic processes (respiration, photosynthesis) as well as toxicity of contaminants adsorbed to the particles. Several studies have demonstrated the tolerance of juvenile salmonids to suspended solids. These studies (Noggle, 1978; Smith, 1978; Ross, 1982) measured 96-hour LC_{50}s ranging from 1,500 to 54,000 mg/L of suspended solids. One additional concern raised with respect to juvenile salmonids is the possibility of eating zooplankton that are covered with particulates associated with toxic chemicals (as opposed to those zooplankton that have ingested contaminated particles). Because of limited information on the effects of suspended solids, adverse effects are assumed to occur when the concentrations exceed background levels.

Ecological Components. A detailed discussion of the aquatic populations and their habitats is presented in EPA's BPJ evaluation (1988a). The populations that were considered as targets for environmental impacts were salmon, herring, benthic invertebrates, and plankton. Salmon and herring were selected for their high commercial and recreational value. Benthic invertebrates were selected for their importance as commercial species (crabs, shrimps) as well as their position as primary prey for the valued predators (salmon, herring). The plankton were chosen because of their importance as a primary food source for the invertebrates as well as being the early life stages of adult species (herring larvae). Marine mammals and birds and aquatic plants were excluded because of lack of data on the populations or species that may be at risk.

Phytoplankton. Phytoplankton inhabit the upper water column (0 to 25 m) where there is adequate light for photosynthesis; phytoplankton blooms generally occur from March to August. Primary production is limited to depths above 8 to 25 m in both bays. Phytoplankton abundance and primary productivity are the same for each bay.

Zooplankton. Zooplankton inhabit a depth of 0 to 150 m. Copepods are the dominant zooplankton group. There appears to be some difference in the abundance of zooplankton between bays. Herbivorous zooplankton, predatory medusae, ctenophores, and chaetognaths dominate the shallower (0 to 25 m) epipelagic water. The euphausiids and amphipods inhabit the deep (50 to 150 m) mesopelagic zones.

Fish. Seventy-five species of fish have been identified from near shore, pelagic, and benthic habitats of both fjords. Dominant species of the near-shore habitat are juvenile salmon, juvenile herring, and starry flounder. Pelagic habits are utilized by herring, salmon, and an abundance of larval fishes. Over 40 species of demersal fish have been identified from the benthic habitat in areas less than 150 m deep. The biomass (kg/km) of demersal fish was greater for Boca de Quadra than for Smeaton Bay.

Herring. Both bays are important rearing habitats for young-of-the-year and age-1 herring. During the winter, herring descend to depths of 125 to 150 m (U.S. EPA, 1988a). This appears to be their preferred depth even when the water is deeper. Approximately 10 to 15 percent of the herring in southeast Alaska spawn at the Kah Shakes spawning ground at the mouth of Boca de Quadra. Due to the proximity of Boca de Quadra to the Kah Shakes spawning grounds, there is speculation that large populations of herring may inhabit the fjord (U.S. EPA, 1988a).

Salmon. Adult salmon are abundant in both fjords. The tributaries to Smeaton Bay support a much larger salmon run (1.4 million salmon) than the tributaries to Boca de Quadra (0.4 million salmon). Juvenile salmon probably stay in the top 10 to 20 m of the water column in saltwater (U.S. EPA, 1988a). Other studies (Straty, 1974) have shown that juvenile sockeye salmon are captured within the top 5 m of the water column during daylight and evening hours. In the sampling from Boca de Quadra and Smeaton Bay/Wilson Arm, it appears that the salmon were located in the 0 to 20 m depths as expected. The numbers also appear to be greatest during the summer migration period (March to August). The winter populations are unknown.

Benthic invertebrates. Benthic invertebrates inhabit a wide variety of niches in both fjords from 0 to 300 m (table 7-3). Rocky intertidal, rocky subtidal, soft-bottom intertidal, and soft-bottom subtidal benthic habitats occur in both bays. The dominant benthic habitat is the subtidal soft-bottom habitat, which accounts for the entire bottom below 35 m. The subtidal biological communities are characterized by distinct shallow (20 to 100 m), mid-depth (100 to 200 m), and deep (greater than 200 m) benthic assemblages. Shallow and mid-depth communities are more productive (a higher number of individuals) than deep communities. Mid-depth and deep communities can be more diverse (Shannon-Weiner species diversity index) than shallow communities because of a more even distribution of species' abundances. Total infaunal biomass is greatest in deep communities because of the presence of large deposit-feeding heart urchins. Subtidal epifaunal assemblages were distributed according to depth. In general, shallow and mid-depth epifaunal assemblages were richer in number of taxa than deep assemblages. Dungeness crab and pandilid shrimp were most abundant in trawl catches from inner-basin Boca de Quadra than the shallower depths of Smeaton Bay. Shrimp caught in pots were most abundant along the sides of fjords at shallow and mid-depths (0 to 150 m). Total epifaunal biomass was greatest in the middle basin of Boca de Quadra and the deep areas of Smeaton Bay because of heart urchin and mud star populations.

The infaunal and epifaunal benthic assemblages and species composition were similar for both basins. Limited sampling (two consecutive sampling periods per year) suggests that shallow and mid-depth infaunal communities in Smeaton Bay may be more productive (number of individuals) than comparable communities in Boca de Quadra's inner basin.

<u>Food Webs</u>. The major sources of energy to the epipelagic, near-shore, and estuarine habitats are solar radiation for photosynthesis and detritus from terrestrial and aquatic sources. Most energy flow (detritus and prey organisms) is downward through the water column. Deep benthic habitats contribute relatively little to habitats in the upper 100 m. Benthic infauna, epibenthic species of commercial value, and demersal fishes are significantly more abundant in shallow (20 to 100 m) and mid-depth (100 to 200 m) soft-bottom habitats of both basins.

<u>Critical Habitats</u>. Abiotic factors affecting population distributions are important measurements for predicting the likelihood of exposure. Abiotic factors that are important in the development of biological communities include substrate types, depths, riverine loading of sediments, nutrients, organic carbon, freshwater intrusion, tidal fluctuations, depth of solar radiation, temperature, and development of a pycnocline during spring and summer.

All pelagic and benthic habitats less than 100 m deep are important to the functioning of the marine community above this depth. The most densely populated and stable part of the mesopelagic habitat is between 50 and 150 m depths. Rocky intertidal and rocky subtidal habitats down to 10 m are an important habitat and source of food for many benthic species. Salmon occur in the upper 20 m of the water column and herring penetrate down to 150 m. The shallow (to

Table 7-3. General Characteristics of Benthic Habitats in Boca de Quadra and Smeaton Bay (VTN Environmental Consulting, 1983b, table 4.2.1., as reported in EPA, 1988a)

Habitat	Zone	Location	Characteristic Organisms[a]
Rocky intertidal	High intertidal and gradual slopes	Throughout fjords	Rockweed
	Low intertidal and all slopes	Throughout fjords	Barnacles, mussels
Soft-bottom intertidal	High intertidal	Keta and Wilson River mud flats	Sedge, insects
	Middle intertidal	Keta and Wilson River mud flats	Rockweed, amphipods
	Low intertidal	Keta and Wilson River mud flats	Polychaetes, bivalves, harpacticoids, eelgrass (Wilson mud flats only)
Rocky subtidal	Vertical walls 0-3 m	Throughout fjords	Red algae, barnacles, sea urchins, sea stars
	3-7 m		Kelps, red and brown crustose algae, gastropods
	7-10 m		Brachiopods, tunicates
	Gradual slopes 0-2 m	Throughout fjords	Eelgrass
	2-10 m		Sea stars, bivalves
Soft-bottom subtidal	20-100 m	BQ inner basin and Smeaton Bay	Polychaetes, bivalves, Dungeness crabs, Tanner crabs, pandalid shrimps, pinch bug crabs
	100-200 m	BQ inner basin and Smeaton Bay	Polychaetes, bivalves, Tanner crabs, pandalid shrimps
	200-330 m	BQ middle and outer basins	Polychaetes, bivalves, sidestripe shrimp, Tanner crabs, heart urchins, mud stars

[a]Larger organisms are listed here.

100 m) subtidal soft-bottom benthic assemblage is the most productive of the subtidal areas. This habitat type is used more by commercially valuable crabs than are deeper subtidal soft-bottom areas.

Endpoints. The assessment endpoint was the loss of critical habitat for benthic invertebrates and fish and the potential for population effects due to discharge of toxicants. Measurement endpoints included model estimates of benthic habitat loss due to tailings deposition, predicted exceedance of the water quality criterion for copper, and predicted exceedance of background suspended solid concentrations.

Comments on Problem Formulation

Strengths of the case study include:

- *The case study identified as stressors the discharge of mine tailings and associated reagents used in the ore separation process.*

- *The case study lists a wide range of important species including salmon, herring, benthic invertebrates, and plankton. The habitats of the ecological components are generally sufficiently characterized. Special emphasis is given to quantifying the potential loss of benthic organisms in the two fjords.*

- *The measurement endpoints are clearly defined and are sufficient for a relative comparison of risk between the two fjords.*

Limitations include:

- *The case study does not adequately explain the risk assessment setting, where it is located, and whether or not the fjords are unique or contain special populations. The assessment only considers potential copper toxicity and the covering of benthic habitat. Effects due to other metals and reagents used in the extraction process are not considered because of lack of data. The exclusion of factors because of lack of data or poor data quality has the potential effect of underestimating risk. Poor data are in effect "rewarded" by being dropped from the study. Since the case study focuses only on identifying which basin is at least risk, stressors are not described completely or evaluated throughout the case study. Catastrophic release of reagents and long-term effects of the project are not addressed.*

- *While water column ecological components are identified, they are not carried through to the later analysis.*

- *The measurement endpoints are insufficient to determine absolute risk, which would have to be addressed if tailing discharges are to be allowed at all.*

7.3.2. Analysis: Characterization of Ecological Effects

The relationship between a given water concentration and a predicted biological response is determined from a series of laboratory tests with specific chemicals and selected organisms. The EPA water quality criteria are based on such a series of tests. The criteria were developed as a means of protecting aquatic communities in any given environment without requiring detailed knowledge of the responses of individual species inhabiting the area. Thus, while concentration-response curves, effective concentration (EC_{50}), lethal concentration (LC_{50}), and acute or chronic studies are not available for all species and chemical forms, the criteria are reasonable guides for a water quality analysis.

Bioassays. Bioassays on a variety of organisms to evaluate acute, chronic, and sublethal effects were completed at this site and with mine tailings from similar projects. A detailed discussion of all bioassays performed on tailings materials is provided in EPA's BPJ report (1988a). Initial tests with juvenile coho salmon, mussel larvae, amphipods, and euphausiids indicated that the acute toxicity at exposure periods from 96 hours to 10 days for these species was low. LC_{50} and EC_{50} concentrations ranged from 109,000 mg/L for the euphausiids to 208,000 mg/L for coho salmon (U.S. EPA, 1988a). Subsequent tests with Dungeness crab zoea, mussel larvae, and amphipods also indicated relatively low toxicity. The LC_{50} or EC_{50} concentrations observed in the studies were 170,000 mg/L for crab zoea, 142,500 mg/L for mussel larvae, and 86,000 mg/L for amphipods (U.S. EPA, 1988a). These tests represented a reasonable preliminary effort to describe some of the possible impacts of the proposed tailings disposal plan. No consideration was given in this first series of tests to the possible physical effects of suspended solids, grain size, or chemical interactions (total organic carbon). Therefore, the results of these preliminary studies are not considered a definite statement of toxicity. In addition, the studies were done in static water. Tailings concentrations were only measured at the beginning of the test; therefore, there was no measure of exposure after day 1.

Acute toxicity studies (U.S. EPA, 1988a) of zooplankton and euphausiids from other mining sites with similar processing procedures indicate that suspended solid concentrations of 560 mg/L over a 40-day period are necessary before ecologically important effects are noted. It should be noted (U.S. EPA, 1988a) that the effects were most likely due to physical stress rather than toxicity.

Chronic and sublethal tests were completed on clam burrowing behavior and phytoplankton growth. Estimates of absolute numbers of organisms observed colonizing an area (table 4-13, USDA, 1987) are subject to considerable uncertainty due to sampling procedures as well as assumptions regarding colonization. VTN Environmental Consulting (1983), as reported by EPA (1988a), observed colonization rates of up to 25 months with test plots in Boca de Quadra. Phytoplankton growth did not appear to be affected by exposure to tailings.

Bioaccumulation. Bioaccumulation of metals (cadmium, copper, manganese, molybdenum, lead, and iron) from tailings was investigated over a 4-month exposure period in the laboratory with crabs, clams, mussels, and sanddabs. Elevated tissue metal concentrations were not observed in test organisms. No behavioral or morphological aberrations were noted. These tests did not address the natural process of chemical uptake that would take place during feeding. Substantial uptake of copper, zinc, cadmium, and lead from tailings by the bivalve *Yoldia thraciaeformis* was noted at another mine site (U.S. EPA, 1988a).

Recolonization. Studies of tailings deposition (VTN Environmental Consulting, 1983, as reported in U.S. EPA, 1988a) for the Quartz Hill project indicate that most species would not survive burial. However, colonization from either larval settlement or lateral migration may replace the lost benthic communities. Few studies have been completed on the colonization of tailings material by aquatic organisms.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *For a comparison of the two fjords, the characterization of ecological effects is not highly significant. The peer reviewers accepted the assumption that all habitats buried would be lost. The use of the water quality criteria for copper in the water column precludes the need for developing an exposure-effects relationship for this endpoint.*

Limitations include:

- *Although the characterization of effects was attempted for other endpoints, data were judged inadequate for use. The most serious inadequacy is the lack of tailings toxicity data.*

7.3.3. Analysis: Characterization of Exposure

Oceanographic Processes. A steady-state model, using the distribution of natural and humanmade conditions, provided the basic framework for estimating the distribution of stressors. Exposure of ecological components was extrapolated from observations of population abundances and habitat preferences.

The site descriptions included oceanographic data for existing conditions, collected and analyzed by the University of Alaska (USDA, 1987). This information has been used to characterize the flow and density structure in each fjord as a function of time. For each fjord, two hydrodynamic seasons were identified (USDA, 1987): (1) summer renewal and (2) winter nonrenewal. For the density structure, six 2-month seasonal periods were identified, based on the analysis of the hydrography in each fjord (U.S. EPA, 1988b). The fjords were compartmentalized spatially as well as seasonally. Based on their hydrographic and hydrodynamic characteristics as well as their biological habitats, each fjord was divided into 12 subregions. The hydrodynamic and hydrographic information was then used to construct a model of the oceanographic processes that could lead to changes in the physical, chemical, and biological states of the two fjords and the likelihood of exposure of organisms to stressors.

Magnitude, Frequency, and Duration of Exposure. Environmental factors that may result in variability in the distribution of settled solids include average and maximum short slope stability, episodic slumping, and *in situ* compacted density of tailings.

Suspended solid distributions are affected by vertical mixing and upwelling, rate of exchange of water, breaking of internal waves, formation of an upper-level plume, and down-fjord turbidity currents. Statistical uncertainty analysis (Monte Carlo method, U.S. EPA, 1988b) was performed on the interaction of each of these factors, resulting in probabilities, rather than discrete estimates of concentrations. Simulations that included system variability were performed to represent the seasonal conditions throughout 1 year, for each of 3 years in the life cycle of the project. These three periods—year 5, year 20, and year 55—were chosen to represent the initial, intermediate, and final stages of the project. Two distinct seasons (winter and summer) were chosen to keep the problem as simple as possible. The maximum suspended solid concentrations (table 7-4) indicate that the higher concentrations reaching the upper water column (above 100 m) are greater for Smeaton Bay/Wilson Arm than for middle-basin Boca de Quadra.

The dissolved and total copper concentrations were measured in several laboratory tests (USDA, 1987). Most studies of environmental toxicity are based on dissolved metal concentrations. However, in developing the documentation on water quality criteria, EPA attempted to account for the possibility that some of the metal content of particulate matter may leach when solids are exposed to ambient water conditions. It is an assumption, therefore, that when applying the water quality criteria in risk analysis, the appropriate form of metal content should be the acid-soluble form. For purposes of evaluating the potential of copper toxicity, the total recoverable fraction as well as the dissolved fraction should be analyzed. The total recoverable metal fraction was not measured according to EPA protocols. However, an alternate method specified as an "extractable" tailings characterization was presented in the USDA revised draft EIS (USDA, 1987, appendix F, table F-6). The "extractable" portion of tailings may be converted to a water column concentration by the following equation:

$$\begin{array}{l} \text{Tailings Metal Extractable} \\ \text{Portion (mg/mg)} \\ \end{array} \times \begin{array}{l} \text{Water Column} \\ \text{Suspended Solid} \\ \text{Concentration (mg/L)} \end{array} = \begin{array}{l} \text{Water Column Metal} \\ \text{Extractable Concentration} \\ \text{(mg/L)} \end{array}$$

This "extractable" fraction and the dissolved fraction were taken in sum as the "total recoverable concentration" (USDA, 1987, appendix E).

In order to provide a basis for comparing the risk due to exposure to metals in either fjord, 100 percent of the copper content (90 μg/g) of the ore and 44 percent (40 μg/g) of the copper content of the ore were selected (table 7-5) as representative of the range of maximum copper concentrations that may be experienced during the life of the mine. While these copper concentrations may not be accurate, they represent in the first case the actual measured maximum copper content of the ore (total copper) and in the second case an estimate of the average extractable portion of total ore for all metal fractions (reported in USDA, 1987, appendix F, from

Table 7-4. Maximum Suspended Solid Concentration Predicted for Upper Water Column (above 100 Meters) of Smeaton Bay/Wilson Arm and Boca de Quadra (U.S. EPA, 1988b)

	Maximum Suspended Solid Concentrations (mg/L)	
Year	Smeaton Bay	Boca de Quadra
5	74	56
20	160	65
55	170	65

Table 7-5. Estimation of Extractable Copper Concentration (U.S. EPA, 1988b)[a]

	Smeaton Bay/Wilson Arm		Boca de Quadra	
	Extractable Portion (10^{-3} mg/mg)	Extractable Concentration (μg/L)	Extractable Portion (10^{-3} mg/mg)	Extractable Concentration (μg/L)
Total ore	0.09	13.5	0.09	5.2
Total tailings	0.069	10.4	0.069	4.0
44% of ore	0.04	6.0	0.04	2.3
44% of tailings	0.03	4.5	0.03	1.7
Tailings extraction	0.022	3.3	0.022	1.3

[a] From a maximum suspended solid concentration (150 mg/L) in the upper 100 meters of Smeaton Bay/Wilson Arm at year 55 and the maximum suspended solid concentration (58 mg/L) in the upper 100 meters of middle-basin Boca de Quadra at year 55.

Burrell, 1983). These estimates of copper content were used only for comparison with the water quality criterion and are not necessarily the true concentration of extractable copper that may only be obtained by chemical analysis of tailings using the EPA-prescribed total recoverable method. The predicted concentration of total recoverable copper in the upper water column (above 100 m) is higher for Smeaton Bay/Wilson Arm than for Boca de Quadra.

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *The dispersion of tailings on the bottom and in the water column is studied with numerical models that contain most of the relevant physical transport mechanisms. Concentrations of copper are computed at two levels representing best-guess and worst-case scenarios.*

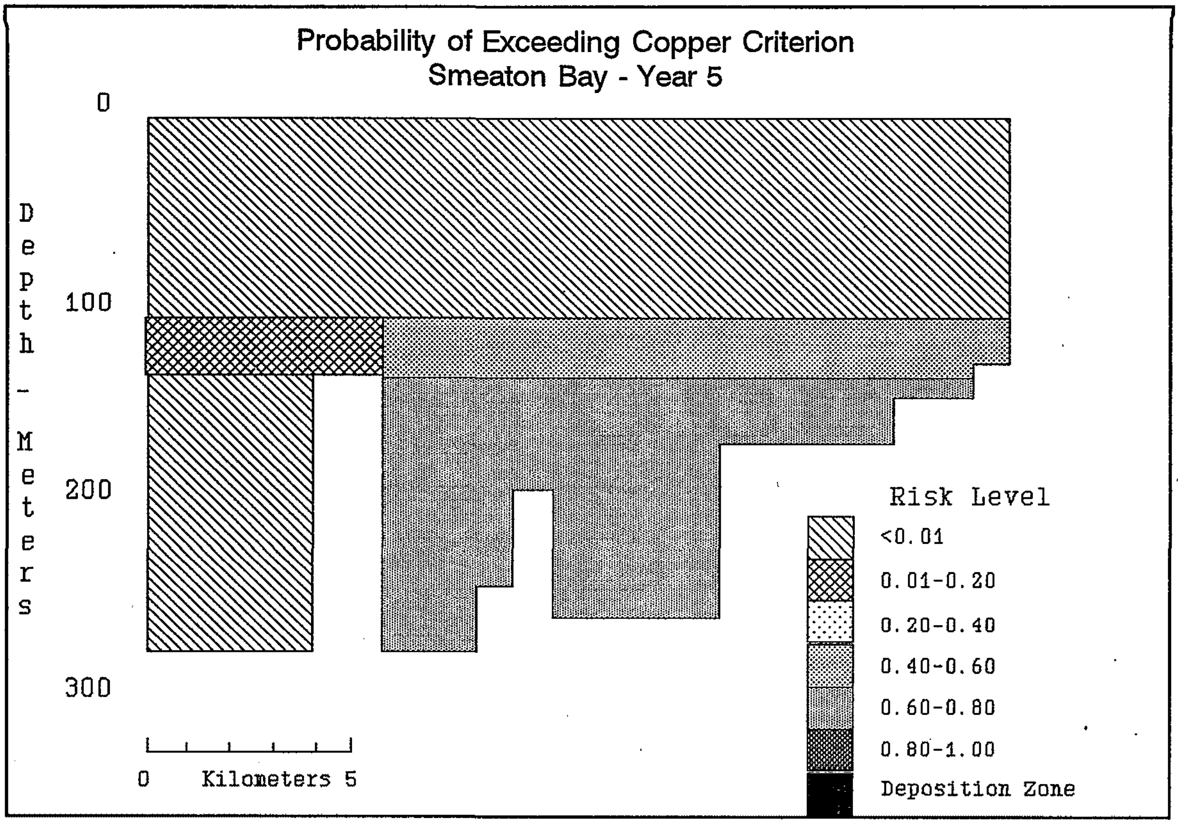
Limitations include:

- *Model verification or comparison is not presented in the case study although some comparisons are discussed in the supporting documents. The exposure assessment does not consider low-density hydrocarbons that might disperse in the surface layer. The model does not include chemical processes.*

7.3.4. Risk Characterization

A detailed discussion of the risk characterization is presented in U.S. EPA, 1988b. The approach used in this study is to define the boundary of that space in each fjord for which the estimated total recoverable copper exceeds the criterion of 2.9 μ/L. The estimated probability of water quality exceedances at any location during a given season is the total number of exceedances at that location divided by the total number of simulations performed. Results indicated that the water quality criterion for copper would be exceeded in both fjords in the deeper water (greater than 100 m) during all years of the proposed project. The likelihood of surpassing the criterion increases with higher levels of extractable copper (from 44 percent to 100 percent) and as the project proceeds to year 55 (figures 7-5a, 7-5b, and 7-5c). The probability of exceeding the criterion increases in the upper water column of Smeaton Bay/Wilson Arm as the project proceeds to year 55. This is not true for Boca de Quadra (figures 7-5a, 7-5b, and 7-5c). The prediction of lower levels of suspended solids in the upper water column of Boca de Quadra results in a decrease in the concomitant copper concentration.

The response of aquatic organisms when exposed to copper, suspended solids, and settled solids was predicted by calculating the maximum loss of organisms due to avoidance or burial (settled solids), or mortality due to exposure to copper. Since water quality criteria are derived as a single reference dose rather than a range of doses from a dose/response curve, no data are available to estimate probabilities of responses other than no effect or a 100-percent response (i.e.,



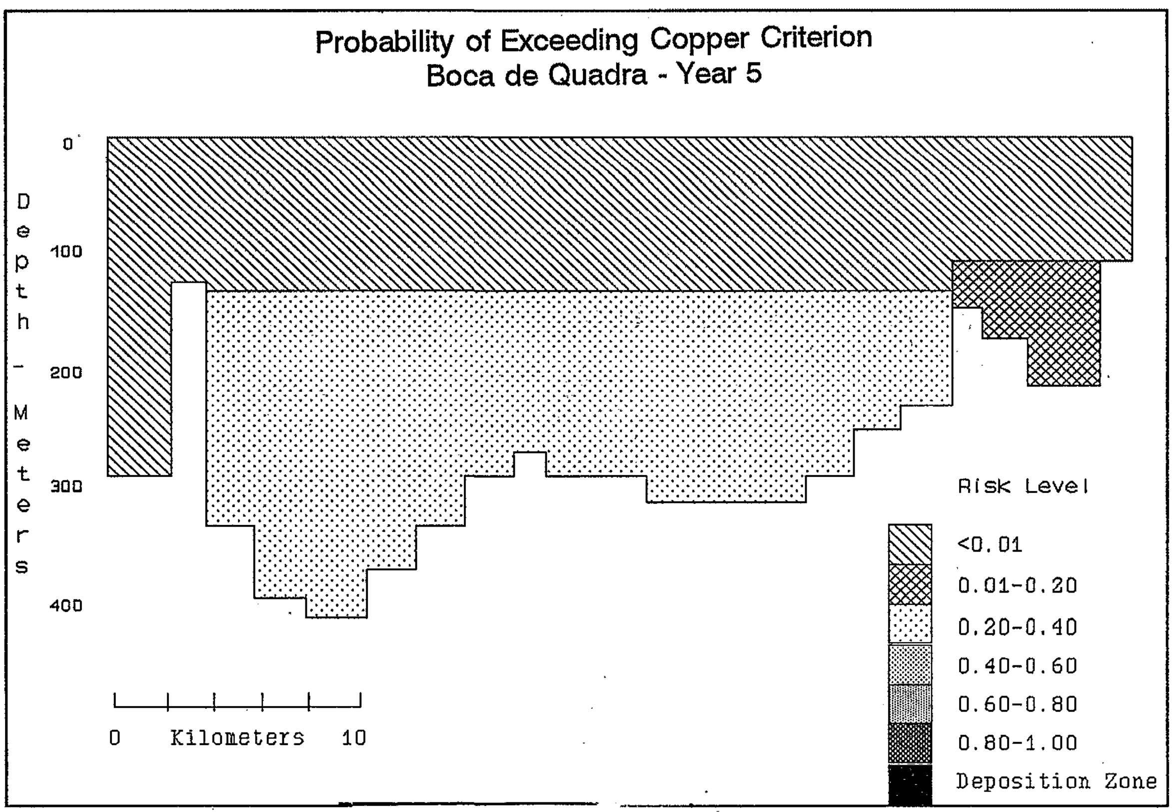
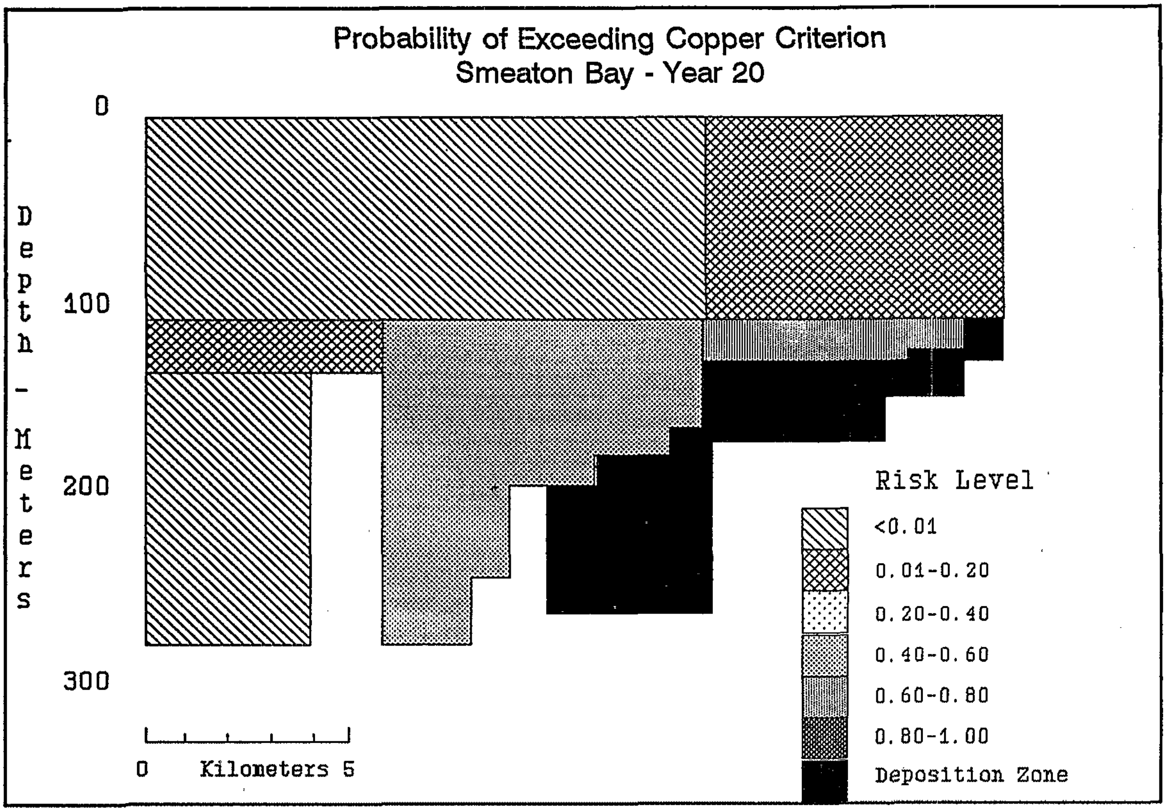


Figure 7-5a. Probability of exceeding water quality criterion—Year 5 (data from U.S. EPA, 1988b)



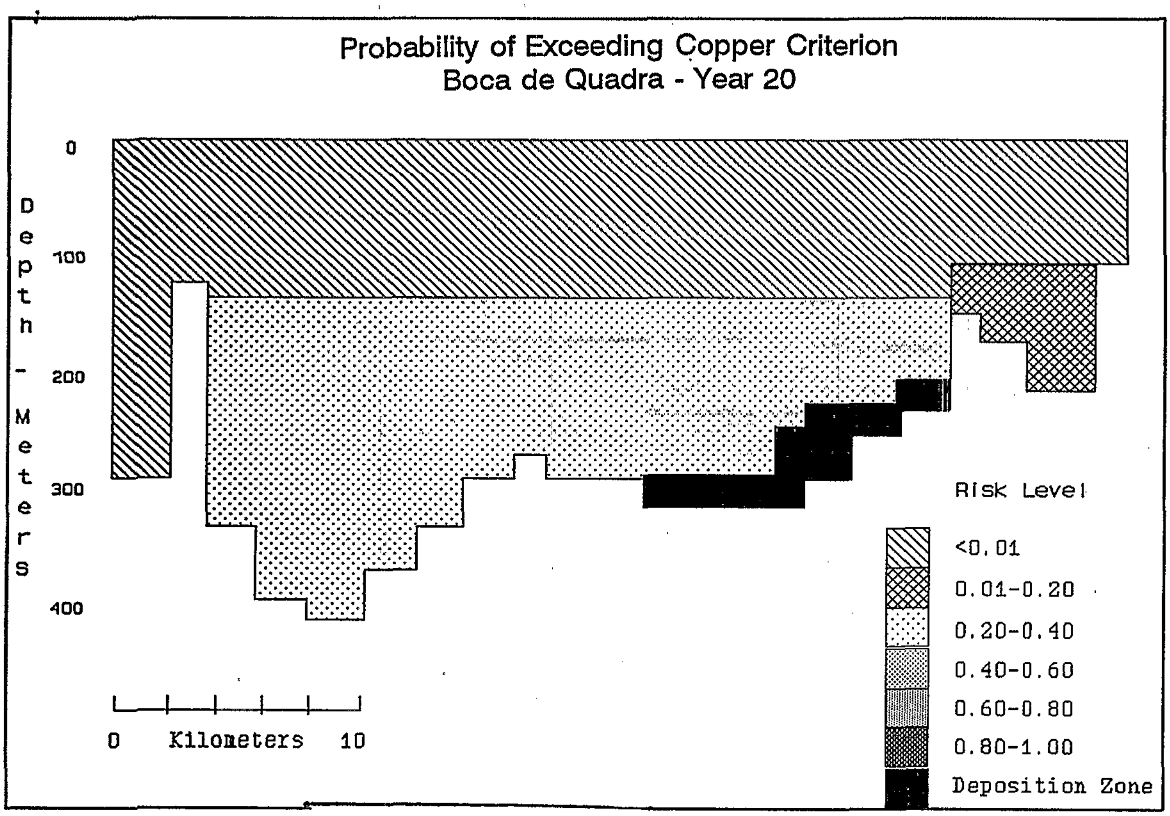
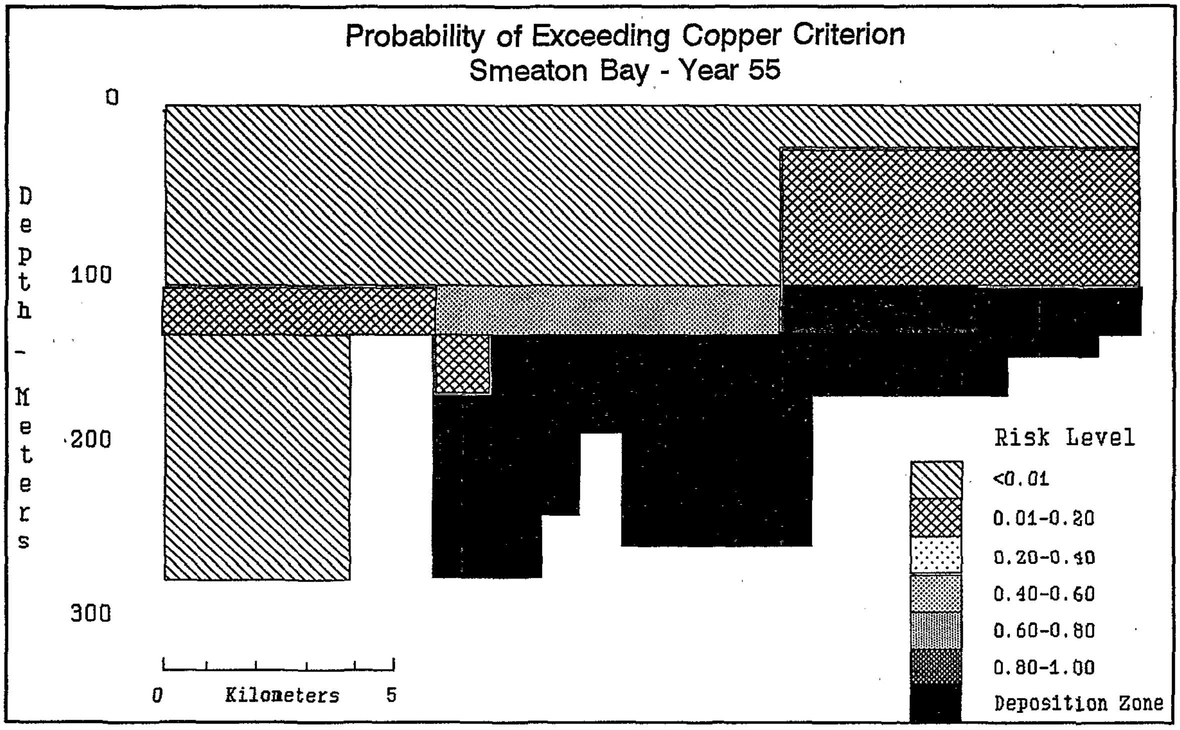


Figure 7-5b. Probability of exceeding water quality criterion—Year 20 (data from U.S. EPA, 1988b)



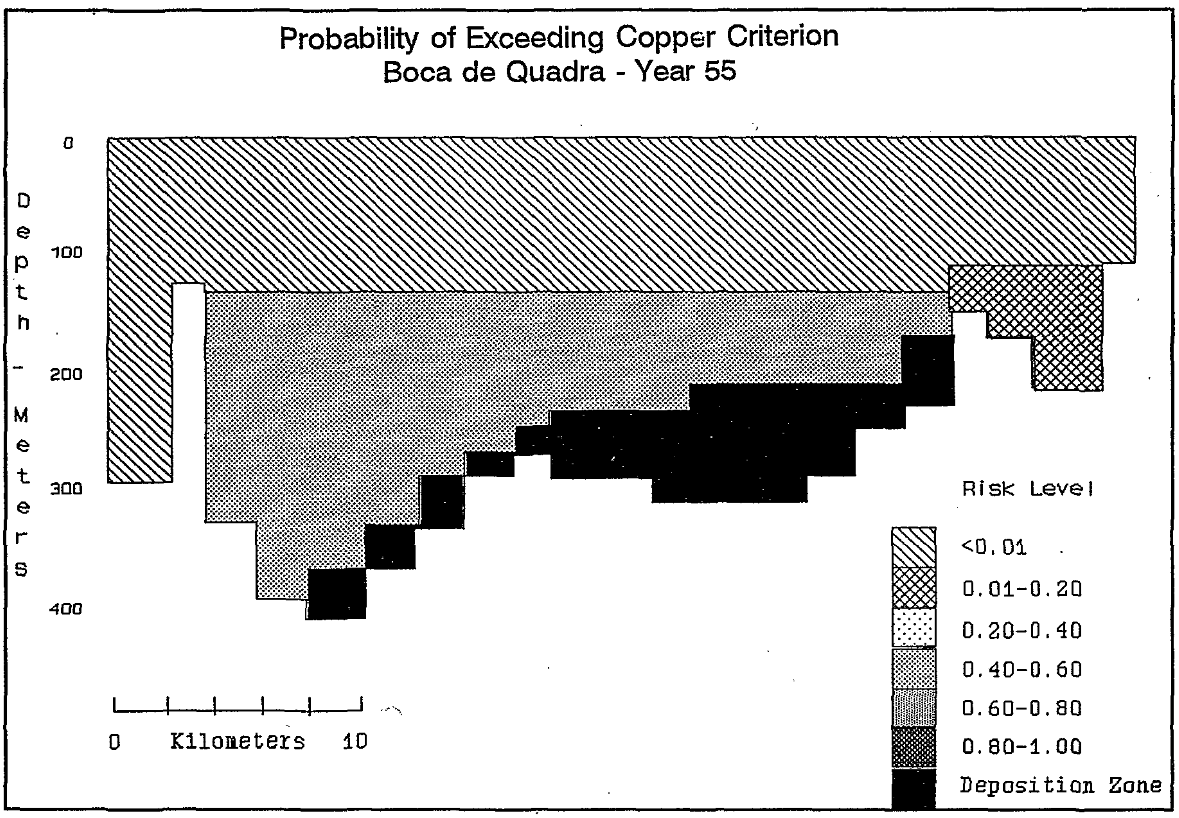


Figure 7-5c. **Probability of exceeding water quality criterion—Year 55 (data from U.S. EPA, 1988b)**

either above or below the criterion for copper). In order to estimate risk, an exposure concentration in excess of an ambient water quality criterion was assumed to cause 100-percent mortality of the exposed animals.

The critical habitat for pelagic organisms is above 100 m in both fjords. The impact of tailings disposal was evaluated with respect to the effect on this important habitat. As the project proceeds to the 55-year point, the concentration of suspended solids and concomitant copper will increase in the upper water column, presenting a higher probability of harm to pelagic organisms such as herring and salmon.

The resulting probability estimates for surpassing the water quality criterion for copper show that extractable copper concentrations will exceed water quality standards over a large part of whichever fjord is chosen for the disposal site. In Boca de Quadra, the average concentration of suspended solids and copper is higher than in Smeaton Bay/Wilson Arm, but the potential impact upon biota may be greater in Smeaton Bay/Wilson Arm due to the fact that the high concentrations occur in the upper water column where there is likely to be more biological activity. The numbers of zooplankton and euphausiids are also greater in Smeaton Bay, reinforcing the likelihood that the risks are greater for biological effects in Smeaton Bay.

The risk due to exposure to settled solids is presented in terms of number of benthic organisms lost per hectare covered with tailings. The adverse effect may be death due to smothering or burial or dislocation due to loss of suitable habitat. Loss of habitat is considered to be equal to death by burial. The loss of benthic organisms due to deposition of tailings is approximately four times greater (table 7-6) for the Smeaton Bay/Wilson Arm disposal option than for the disposal in Boca de Quadra. Smeaton Bay/Wilson Arm tailings disposal will result in adverse impacts to 1,660 ha of habitat compared with 1,600 ha adversely affected with the Boca de Quadra disposal option (U.S. EPA, 1988a).

Herring habitat affected in Smeaton Bay/Wilson Arm will be 320 ha compared with only 20 ha affected in Boca de Quadra (U.S. EPA, 1988a). If recolonization is slow, the loss will be substantial. It is likely that actual colonization will take much longer than predicted from laboratory studies, given the distance from a source of organisms, sediment slumping, sediment toxicity from metals and reagents, the constantly shifting course of undersea channels, and the continuing deposition of tailings after the organisms have adjusted to a certain distribution of settled tailings.

The two fjords also will respond differently to changes in geometry as the project evolves. Little change occurs in Boca de Quadra with time. In Smeaton Bay/Wilson Arm, there is a noticeable change in bottom geometry, specifically in the location of the discharge point as the project progresses. In Smeaton Bay/Wilson Arm, the discharge point is within 75 to 100 m of the surface during the project's final stage. It is, therefore, not surprising that the model predicts increasing impacts in the upper water column as time progresses.

It may be assumed that impacts (reagent toxicity, other metal toxicity) that are not addressed in this quantitative statement of risk due to oceanographic variation in the two fjords will also increase the risk estimate for both fjords. Since measurements of other agents were not

Table 7-6. Worst-Case Estimates of Biomass (kg) of Demersal Organisms Lost Over Life (55 Years) of Proposed Quartz Hill Mining Project (U.S. EPA, 1988b)[a]

	Dungeness Crab	Tanner Crab	Pot Shrimp	Trawl Shrimp	Walleye Pollock	Rockfish	Flatfish
Option							
Boca de Quadra	22,500	13,750	40,150	45,100	45,100	30,800	17,050
Smeaton Bay	52,800	129,800	26,400	313,500	33,000	11,002	17,250
Total Demersal Organisms Lost							
Boca de Quadra	175,400						
Smeaton Bay	773,850						

[a]The organisms are lost due to smothering, avoidance, or toxicity of settleable solids. Assumes annual colonization.

completed and expert knowledge regarding impacts is limited, the expected increase in risk estimates is not included in this analysis.

Uncertainty. Statistical uncertainty analysis (U.S. EPA, 1988b, appendix A) was used to characterize the natural variability in the oceanographic characteristics of each fjord. The result of this characterization is a statement of certain differences in the hydrodynamics and hydrography of each fjord. This oceanographic description of each fjord was used to predict the concentration and distribution of copper, settled solids, and suspended solids. Verification of the steady-state diffusion model used to predict copper and suspended solid concentrations in the basins was not completed. However, comparison with other mining operations (table 7-7) suggests that the model predictions are in fact close to realistic estimates of contaminant dispersion.

Comparison with Natural Background. Concentrations of dissolved metals in the tailings are expected to be at least one to two orders of magnitude higher than concentrations now observed in either Boca de Quadra or Smeaton Bay.

Predicted levels of suspended solids have been compared with the ambient concentrations measured in earlier studies (U.S. EPA, 1988b). These studies found a maximum of 5 mg/L, with average concentrations less than 1 mg/L for Boca de Quadra. Due to limited data on the "natural condition," the risk estimate for suspended solids is presented only as a comparison to available ambient concentrations rather than as a probability statement of potential harm to aquatic organisms. The average suspended solid concentrations that are predicted for tailings discharges to middle-basin Boca de Quadra and Smeaton Bay/Wilson Arm exceed natural ambient conditions throughout both fjords (U.S. EPA, 1988b, figures 9 and 10 in appendix A).

Table 7-7. Comparison of Projects in North America That Discharge Mining Tailings to Marine Waters (U.S. EPA, 1988b)

Project	Location	Duration (years)	Maximum Discharge (tons/day)
Island Copper	Rupert Inlet, B.C.	16	60,000
Kitsault Molybdenum	Alice Arm, B.C.	1.5	15,000
Quartz Hill	Alaska	55	80,000

Comments on Risk Characterization

Strengths of the case study include:

- *For a comparison of relative risks in two basins, the risk is adequately characterized, and Boca de Quadra appears to be less at risk than Smeaton Bay. Risk of exposure to solids is presented in terms of number of benthic organisms lost per unit area covered. Risk to the water column is presented in terms of the percentage of times the model predicts that water quality criterion for copper would be exceeded in the upper and lower layers of the fjords.*

- *The uncertainties for oceanographic characteristics are clearly identified. The modeling approach used to define the zone of greatest impact and to compare impacts on the two sites is very useful. The areal extent of impacts is an important addition to the estimate of scale for risk as well as exposure.*

Limitations include:

- *The discussion of aquatic populations is only addressed qualitatively. Uncertainty analysis could be applied to the distribution of aquatic populations as is done for the stressors. Inadequate chemical analysis of tailings and sampling of biota increases the overall uncertainty in the analysis.*

- *The assessment is clearly not written as a stand-alone document and should be considered in light of the discussions contained in the pertinent references, which give a broader explanation of the environmental impacts.*

- *The risk characterization does not assess damage to plankton or fish apart from use of water quality criteria. Risks from normal and catastrophic release of reagents are not characterized. This information would be required to assess an absolute level of risk.*

- *Uncertainty is not used consistently in the case study. This problem is common because uncertainty can have a variety of meanings. This study uses uncertainty in three different ways that might be categorized as absolute uncertainty, statistical uncertainty, and relative uncertainty.*

  — Absolute uncertainty is the comparison of the analysis with the real future state of the system. This uncertainty cannot be evaluated because the future is unknown.
  — Statistical uncertainty is assessed only in a rudimentary way by identifying the percentage of times water quality standards were exceeded.

Comments on Risk Characterization (continued)

— Relative uncertainty can be assessed by comparing a set of data against a specific or abstract reference. A high relative uncertainty is assessed if observations have known problems compared with a standard of a "good analysis." Relative uncertainty is implied in the case study. The relative uncertainty is important to the study, since it is through a comparison of one set of results with another that we gain an intuitive estimate of uncertainty. In the case study, the relative uncertainty of the modeling study could have been more clearly defined by pointing out that the results of the model are in agreement with observed distributions of tailings in a deep-water discharge mining operation on Vancouver Island.

- *The reviewers felt that this case study had problems in the way the goals are identified and in the exclusion of some elements of risk analysis on the grounds that data are lacking or of poor quality.*

- *While none of the reviewers disagreed with the conclusions that discharge into Boca de Quadra involved the least risk, they did take strong exception to the contention of the case study that only relative risk and not absolute risks are necessary to consider for the completion of the risk analysis.*

General comments:

- *The regulatory decision rested on which alternative was the least environmentally damaging; therefore, absolute risks were not necessary for completion of this risk analysis. Sampling and analysis were completed as part of the Environmental Impact Statement prior to the initiation of this risk assessment. Since the data were collected without having first developed an hypothesis or completing a planning assessment, much of the information that would be needed to do a thorough analysis of risk was not available. If one accepts risk assessment as an iterative process, this assessment should be viewed as a first-order iteration. However, decision-making under conditions of uncertainty is a fact of life in the management of environmental resources. The project evaluation was considered complete at this stage and a decision was made not to allow discharge of tailings into Smeaton Bay.*

- *Hypothetical considerations of food chain and ecosystem effects should provide some perspective on the importance of impacts at lower organizational levels.*

- *Using decision analysis prior to the initiation of the investigation would have been useful in identifying the parameters for which extensive sampling or laboratory analysis would have reduced the risk characterization uncertainty.*

Comments on Risk Characterization (continued)

- *Although the regulatory decision was dependent on relative comparisons, a more thorough discussion of other metals and reagents would have strengthened the comparisons as well as achieved a better understanding of the ecosystem level of risk.*

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SECTION EIGHT

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

ASSESSING ECOLOGICAL RISK AT ROCKY MOUNTAIN ARSENAL

AUTHORS AND REVIEWERS

AUTHORS

Carolyn L. Fordham
Terra Technologies
Golden, CO

Douglas P. Reagan
Woodward-Clyde Consultants
Denver, CO

REVIEWERS

Nancy A. Bryant (Lead Reviewer)
ENSR Consulting & Engineering
Acton, MA

Kenneth L. Dickson
Institute of Applied Sciences
University of North Texas
Denton, TX

Thomas M. Frost
Center for Limnology
University of Wisconsin
Madison, WI

Judy L. Meyer
Institute of Ecology
University of Georgia
Athens, GA

Douglas P. Ormrod
Office of Graduate Studies
University of Guelph
Guelph, Ontario, Canada

Randall S. Wentsel
Toxicology Division
U.S. Army Chemical Research,
Development, and Engineering
Center
Aberdeen Proving Grounds, MD

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LIST OF ACRONYMS

AChE	acetylcholinesterase
ARAR	applicable or relevant and appropriate requirement
AWQC	Ambient Water Quality Criteria
BAF	bioaccumulation factor
BCF	bioconcentration factor
BMF	biomagnification factor
BW	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (Superfund)
DBCP	dibromochloropropane
EPA	U.S. Environmental Protection Agency
FWQC	Federal Water Quality Criteria
LC_{50}	lethal concentration to 50 percent of organisms tested
LD_{50}	lethal dose to 50 percent of organisms tested
LOAEL	lowest observed adverse effects level
MATC	maximum acceptable tissue concentration
NCP	National Contingency Plan
NOEL	no observed effects level
NPL	National Priorities List
RMA	Rocky Mountain Arsenal
SARA	Superfund Amendments and Reauthorization Act of 1986
TBC	to be considered
USFWS	U.S. Fish and Wildlife Service
WQS	water quality standards

ABSTRACT

The Rocky Mountain Arsenal (RMA) is a 27-square-mile U.S. Army installation north of Denver, Colorado. The site was used from 1942 to 1982 by the U.S. Army and its lessees for production of chemical and incendiary munitions, pesticides, and other chemicals. In 1985, the Army initiated studies to determine the nature and extent of contamination in soil, ground water, surface water, and biota. The Biota Remedial Investigation Report (ESE, 1989) summarized the nature and extent of contamination in biota at RMA and investigated contaminant effects. By establishing a quantified relationship between adverse effects on biota and chemical concentrations in soil and water, studies also determined contaminant levels in the environment that would be likely to pose a threat to wildlife on RMA. Ecological risk was not quantified in this study because abiotic data were not fully available; thus, exposure assessment could not be completed. Studies at RMA are ongoing.

Stressors/Contaminants of Concern. Because of RMA's varied uses in the past both as a facility for the production of Army chemical and incendiary munitions and for commercial chemical manufacturing, there are many different chemicals in the environment. Seven chemicals were selected as being of major concern to biota based on consideration of parameters such as toxicity, areal extent, and persistence in the environment. These chemicals were aldrin, dieldrin, endrin, isodrin, dibromochloropropane, arsenic, and mercury. An additional 32 contaminants were also evaluated, but not to the extent of the major contaminants of concern.

Ecological Components. RMA provides important habitat for many species, including the endangered bald eagle, which winters at RMA. Other raptors frequently observed at RMA include golden eagles, ferruginous hawks, northern harriers, rough-legged hawks, great-horned owls, and burrowing owls. Resident mammals include mule deer, white-tailed deer, badger, coyote, cottontail rabbit, jackrabbit, and prairie dog. Waterfowl and wading birds occur in the lake areas.

Criteria for Evaluating Risk. Criteria based on toxicity to terrestrial and aquatic organisms were determined for surface water and sediment by a method termed "Pathways Analysis" (ESE, 1989), which provided a quantitative means of relating contaminant concentrations in sediment and water to concentrations and effects in biota. EPA's Ambient Water Quality Criteria for the Protection of Freshwater Aquatic Organisms were considered in relation to toxicity to aquatic life. Soil criteria were estimated based on toxicity only to terrestrial organisms (ESE, 1989). Bioaccumulation was also a factor in determining levels in the environment not expected to pose a threat to wildlife (ESE, 1989; Fordham and Reagan, 1991). The modeling approach included quantitative estimation of uncertainty.

Endpoints. A wide array of toxicological and ecological assessment endpoints were examined. Adverse chemical impacts and chemical contamination of ecological components at levels that might be expected to impair structure or function were considered as toxicological assessment endpoints. Decreased population success was considered as an ecological assessment endpoint.

Contaminant concentrations in plant and animal tissue were one of the measurement endpoints because of historical correlations of mortality with pesticides in tissue and because many

of the contaminants of concern were bioaccumulative. Depressed activity of acetylcholinesterase was examined in birds and mammals as another measurement endpoint for determining chemical impacts. Population density, occurrence, and age classes were evaluated for certain species at RMA as a measurement endpoint for determining population success. Reproductive success was considered a measurement endpoint for avian species population success because some of the contaminants at RMA were linked with avian reproductive effects.

8.1. RISK ASSESSMENT APPROACH

The case study described herein does not represent a complete risk assessment (figure 8-1). However, useful information is provided on problem formulation as well as on characterization of ecological effects. Ecological risk assessment methodologies are discussed. A complete characterization of exposure for chemical stressors is lacking but will be completed for the site as a result of ongoing studies. Historical information and data on the distribution and concentration of contaminants from Phase I studies of abiotic media (e.g., soil, sediment, water) were used to determine sites of contamination on Rocky Mountain Arsenal (RMA). These initial data were used to select sites for field investigation and to determine major contaminants of concern. Detailed quantitative data were available for biological media only, and exposure was inferred from measured concentrations of site-related contaminants in tissues, as opposed to measuring concentrations in tissue compared with colocated, measured concentrations in abiotic media. This additional step in evaluating exposure will be performed in ongoing risk assessment activities.

A major feature of this risk assessment was the development and use of an exposure pathway model as a "new tool" to establish a quantitative relationship between concentrations of bioaccumulative contaminants in abiotic media and concentrations at all levels in the selected food webs. The model is used to determine if contaminant levels in abiotic media present a risk to biota, and it can be used to establish ecologically based remediation criteria. The model was used for both aquatic and terrestrial ecosystems at RMA, and can be adapted for use in any ecosystem. The overall approach also involves the development of criteria for the selection of key species and for the identification of contaminants of concern to biota.

Another major aspect of this study was the use of food webs to evaluate exposure pathways. Species at RMA were organized into both a terrestrial and an aquatic food web. There was some overlap between the two food webs for species, such as the bald eagle, which preys on terrestrial mammals as well as waterfowl. The food webs aid in focusing a study design toward species that may be most affected by exposure as a result of diet. This paper focuses primarily on the terrestrial food web; for a discussion of the aquatic food web, see Fordham and Reagan (1991).

8.2. STATUTORY AND REGULATORY BACKGROUND

The Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA or Superfund) and CERCLA as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA) stipulate that environmental health should be protected as well as human health and welfare. The RMA investigation was conducted according to requirements in the National Contingency Plan (NCP) and Guidance for Remedial Investigations (U.S. EPA, 1985).

Under CERCLA, Ambient Water Quality Criteria (AWQC) can be considered an applicable or relevant and appropriate requirement (ARAR) or to be considered (TBC) for ecological risk assessments. The U.S. Environmental Protection Agency (EPA) develops Federal Water Quality Criteria (FWQC) under the authority of the Clean Water Act from the Office of Water Regulations and Standards. These are nonenforceable guidelines that can be used by the states to determine Water Quality Standards (WQS). The AWQC for the Protection of Freshwater Life and Their Uses are the only criteria currently available that are specific to the protection of nonhuman life,

Figure 8-1. Structure of Analysis for Rocky Mountain Arsenal

PROBLEM FORMULATION

Stressors: seven major and 32 other chemicals.

Ecological Components: vegetation, invertebrates, birds, and mammals.

Endpoints: assessment endpoint is reproduction and survival of species; measurement endpoints are chemical residues, population data, acetylcholinesterase activity, behavior and reproductive success.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
Limited data were available on chemicals in abiotic media; data were available for tissue residues; study is ongoing.	A Pathway Analysis Model was used to characterize exposure pathways and develop critical effect levels for abiotic media: toxicological criteria or benchmarks were used: field observations of mortality were made.

RISK CHARACTERIZATION

Risk characterization would involve comparison of exposure and intakes to benchmarks or criteria; however, this has not yet been performed because the study is not complete. Comparisons were made between control and impacted areas; Monte Carlo analysis was used.

although for some chemicals, AWQC are based on protection of humans exposed through consumption of aquatic life. AWQC were evaluated for each of the major contaminants of concern to determine that human consumption was not the basis of the criterion.

8.3. CASE STUDY DESCRIPTION

8.3.1. Problem Formulation

Site Description. RMA is a U.S. Army installation consisting of approximately 27 square miles located north of Denver, Colorado (figure 8-2). In 1942, RMA was used to produce chemical agents such as blister agent, incendiary munitions, and irritants. From 1945 to 1950, obsolete World War II ordnance was destroyed at RMA by detonation and burning. Nerve agent was produced in the 1950s. The Army leased portions of the arsenal to various manufacturers from 1947 to 1982. Organochlorine pesticides such as DDT, DDE, dieldrin, aldrin, and endrin; herbicides; adhesives; cutting oils; and other chemicals were produced during this period.

From 1970 until 1984, the primary U.S. Army activity has been demilitarization of chemical warfare agents. Currently, the sole mission at RMA is to remediate contamination; present activity at RMA to carry out this mission is limited to land management, wildlife management, security, technical investigations of contaminant distribution and effects, development of environmental remediation strategies, and interim response actions.

Disposal practices, spills, and other releases with potential adverse ecological effects have been historically documented. Some of the most important sources of contamination for biota at RMA were the wastewater basins in Sections 26, 35, and 36, as well as the four reservoirs known as the Lower Lakes (ESE, 1989). RMA is a National Priorities List (NPL) site, and certain areas have been addressed in interim response actions (EBASCO, 1988). RMA is also the site of a U.S. Fish and Wildlife Bald Eagle Management Area. Visitors are allowed onsite in uncontaminated areas under controlled conditions to view the eagles in winter at RMA. The Lower Lakes (Upper and Lower Lake Derby, Lake Ladora, and Lake Mary) have trophy-sized bass and pike and are fished on a catch-and-release basis only.

Stressors. Contaminants recorded in the RMA environment or inferred from a knowledge of historical practices at RMA included volatiles, pesticides, herbicides, inorganic salts, Army chemical agents and their degradation products, and heavy metals. The U.S. Army and U.S. Fish and Wildlife Service conducted biological monitoring studies during much of the time that RMA was an active facility. Historical studies (1949–1982) of biota in the basins and Lower Lakes have indicated death and abnormal behavior for several waterfowl species, other birds, mammals, and fish (Finley, 1959; USFWS, 1965; USA DPG, 1973; USA EHA, 1976; Linkie and Stiles, 1976; McEwen, 1981; DeWeese et al., 1982a, b; USFWS, 1983; McEwen, 1983; Thorne, 1984). Organochlorine pesticide levels in tissues were often reported to be elevated in these studies. Waterfowl losses associated with contaminated sites on RMA ranged from 2,000 to 3,000 individuals annually throughout the 1950s and early 1960s. Waterfowl mortality declined after 1965, at which time the U.S. Army drained Upper and Lower Derby and Ladora Lakes and removed contaminated sediments.

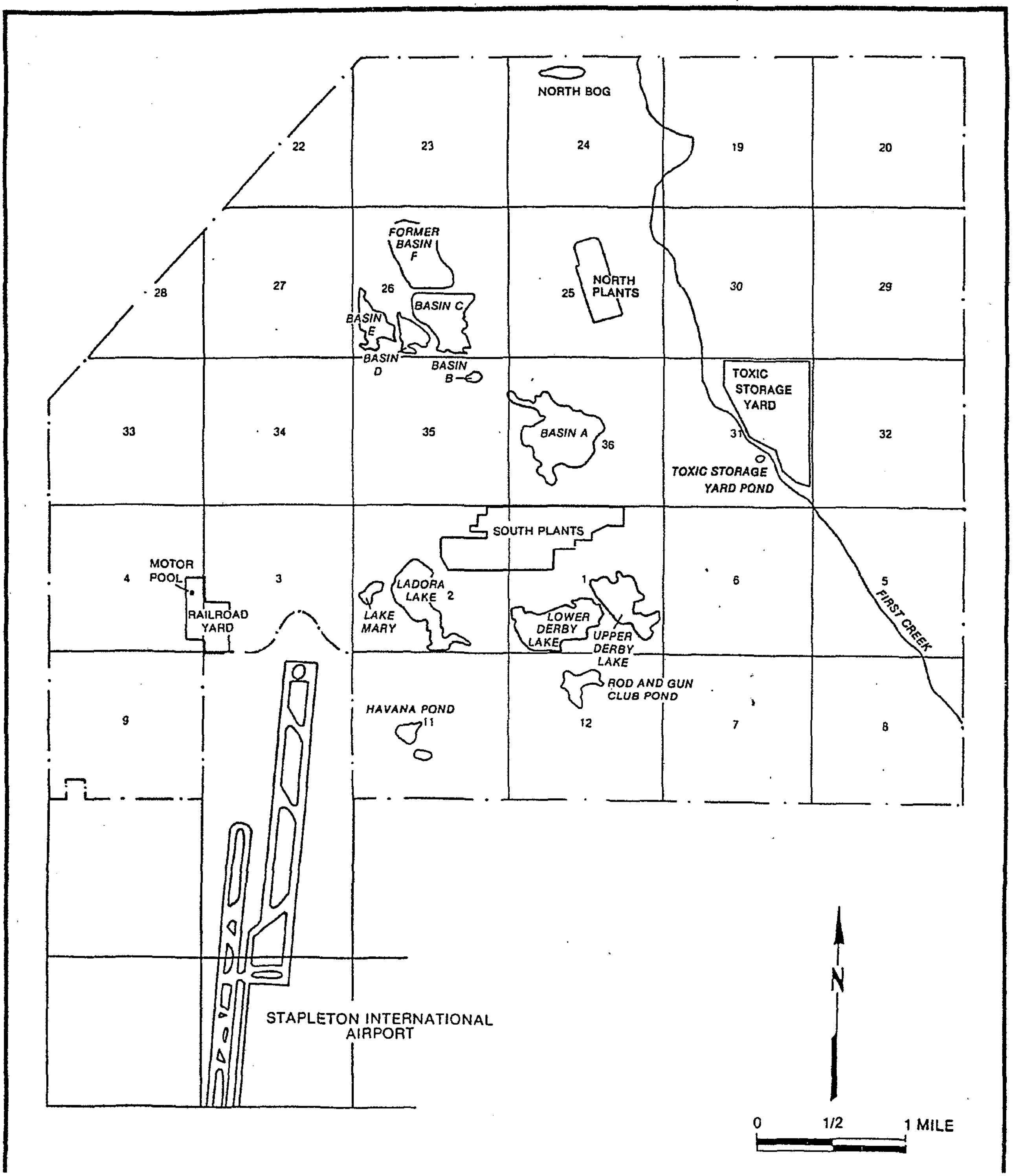


Figure 8-2. Map of Rocky Mountain Arsenal (ESE, 1989)

Data from earlier investigations indicated that organochlorine pesticides might be affecting avian reproductive success at RMA. The available literature indicates a cause-and-effect relationship between certain organochlorine pesticides and different aspects of avian reproduction (St. Omer, 1970; Haegele and Hudson, 1974; Davison et al., 1976; Blus, 1982; Spann et al., 1986).

Seven chemicals were selected as being of major concern to biota based on consideration of parameters such as toxicity, areal extent of distribution, and persistence in the environment. These chemicals were aldrin, dieldrin, endrin, isodrin, dibromochloropropane (DBCP), arsenic, and mercury. An additional 32 contaminants were also evaluated, but not to the extent of the major contaminants of concern. Isodrin and DBCP were not analyzed in tissue; isodrin is metabolized to endrin and DBCP is rapidly metabolized and is not biologically persistent. Arsenic, mercury, endrin, aldrin, and dieldrin were target analytes in tissue. DDT and DDE also were target analytes in tissue, although data available at the time did not suggest that they should be included as major contaminants of concern.

Phase I data on contaminants in abiotic media indicated that the most contaminated areas were in the areas of chemical manufacturing, storage, and disposal. Sampling of sediments and water in the lower lakes (Upper and Lower Derby Lakes and Lake Ladora) and soils in the South Plants, Basins, and Toxic Storage Yard (figure 8-2) indicated elevated levels of chemicals sufficient to provide a basis for biota field sampling. Maximum concentrations of the major contaminants of concern in the biosphere (surface water, sediments, and upper 20 ft of soil), based on 1985–1987 sampling, are presented in table 8-1.

Ecological Components. While any of the species that occur at RMA potentially could be exposed to chemical stressors, this investigation focused on those species that were the most important or sensitive in the onsite ecosystem, with the assumption that protection of the most important or sensitive species would ultimately afford protection to all species. By protecting the most important species in the ecosystem, ecological effects are minimized. Protection of the most sensitive species indicates that less sensitive species are also protected, thereby minimizing damage to ecological health. These concepts are fundamental to derivation of the EPA AWQC and are consistent with human health risk assessment procedures.

Species were classified as important or sensitive if they met the following criteria of ecological, regulatory, or economic importance: (1) classified as federally threatened or endangered under the Endangered Species Act of 1973, as amended; (2) listed as endangered or considered as a Species of Special Concern by the State of Colorado; (3) used as a game species consumed by humans; or (4) serve as a species of special ecologic value (major prey species, predator, highly sensitive, bioindicator, etc.). Species selected for tissue monitoring were chosen after deriving the list of important species with the above criteria. The following additional selection criteria were used to identify appropriate species for residue monitoring: (1) considered important components of regional ecosystems, (2) representative of the range of trophic levels in food chains/webs in regional ecosystems, (3) economically important (e.g., game species), or (4) representative of higher trophic levels in food chains/webs in regional ecosystems.

Table 8-1. Maximum Concentrations of Major Contaminants of Concern Based on 1985–87 Sampling (ESE, 1988)

	Medium		
Analyte	Soil (to 20 ft) ($\mu g/g$)	Sediment ($\mu g/g$)	Surface Water ($\mu g/L$)
Aldrin	40,000	1.38	750 (LT)
Arsenic	112,00	8.92	200,000
DBCP	31,700	1.30 (LT)	1,900 (LT)
Dieldrin	7,240	1.01	470 (LT)
Endrin	4,650	0.74 (LT)	800 (LT)
Mercury	35,000	2.30	11
Isodrin	3,200	0.30	370 (LT)

LT = Less than the Certified Reporting Limit

The ecological components investigated in this study for contaminant concentrations, population effects, or reproductive effects included: vegetation (morning glory, sunflowers, and others), invertebrates (earthworms, grasshoppers, aquatic snails), and vertebrates (mallard duck, pheasant, mourning dove, raptors [i.e., birds of prey], prairie dog, cottontail rabbit, mule deer, coyote, badger).

Aquatic species other than snails were investigated by Rosenlund et al. (1986) and other studies; these data will not be documented in detail here. The aquatic data will be referred to in this document in a manner consistent with its collection and use in the Biota Remedial Investigation. For example, as part of this investigation pathways analysis was performed for the aquatic food web leading to bald eagles and results of this analysis were used in conjunction with data collected on contaminant concentrations in the tissue of various components of the aquatic food web (e.g., aquatic plants, largemouth bass, and northern pike) to establish exposure for bald eagles.

Eagles and other raptors were studied because they fit the categories of state or federal endangered or threatened species, because they are predators and thus highly exposed to bioaccumulative pesticides, and because, as birds, they are sensitive to the effects of organochlorine pesticides. Raptor populations were surveyed throughout the year. Raptors observed at RMA include American kestrels; rough-legged, red-tailed, Swainson's, and ferruginous hawks; golden and bald eagles; northern harriers; and great-horned and burrowing owls. Bald eagle populations were monitored during the winter months because this species utilizes RMA primarily for winter roosting and feeding.

The only endangered mammal suspected of occurring at RMA was the black-footed ferret. Black-footed ferret population surveys were conducted in 5,000 acres of prairie dog towns on RMA, but no evidence of this endangered species was observed. Therefore, black-footed ferrets were not considered further.

Ring-necked pheasants and mallard populations at RMA are historically linked with contaminant-related effects. Both species are important prey species in onsite food webs and also are game species that are consumed by humans. Mourning doves are also prey species, and as ground feeders could be expected to be highly exposed to contaminants in soils.

Other mammal species were considered in this study. Mule deer fit the criteria of being a game species consumed by humans and also provide food for predators or scavengers. In addition, because they are relatively long-lived, they may be expected to accumulate certain contaminants.

Population surveys were conducted for prey species that might be expected to show adverse responses to chemical contaminants. Many of these species also form important components of aquatic and terrestrial food webs on RMA. Prairie dogs and rabbits are important prey items, abundant in the RMA ecosystem, and live in close contact with soil. Grasshoppers, earthworms, and aquatic snails are also important prey species, are abundant, and have been indicated to bioaccumulate certain metals and pesticides.

Vegetation transects were conducted to determine the dominant species of vegetation in both disturbed and undisturbed areas. Sunflower, cheatgrass, and kochia were some of the species observed in these surveys. In support of the Biota Remedial Investigation, quantitative vegetation studies conducted by Morrison Knudsen Corporation, contractors for Shell, indicated that the level of onsite physical disturbance due to management activities precluded a quantitative analysis of potential adverse effects on vegetation due to chemical contamination. However, residue analyses were conducted on two types of vegetation (morning glory and sunflowers) that often occurred in potentially contaminated areas.

The number, type, and habitat use by mammalian predators were noted, and limited quantitative studies of abundance and distribution were conducted by Morrison Knudsen, who provided these data to the Army for use in the ecological evaluation. Mammalian predators observed at RMA include red fox, gray fox, long-tailed weasel, coyote, and badger. Coyote and badger were collected as fortuitous samples when animals were found dead or dying.

Endpoint Selection. Several toxicological and ecological assessment endpoints were examined (table 8-2). Chemical contamination of ecological components at levels that might be expected to impair structure or function were considered important because many of the contaminants of concern to biota were both toxic and bioaccumulative. Adverse effects due to chronic or acute exposure were also considered as toxicological assessment endpoints. Decreased population success was considered as an assessment endpoint of ecological importance.

Several measurement endpoints were used in this study to evaluate population success. Species occurrence, population density, and age classes were evaluated as a measurement endpoint for determining population success for prairie dogs. Population density and occurrence were measured for large raptors, American kestrels, grasshoppers, and aquatic snails. Total cover, species richness, and phenology were endpoints for vegetation. Data from contaminated sites on RMA were compared with data from onsite and offsite control sites.

Reproductive success was a measurement endpoint used to evaluate population success for avian species. Reproductive success was measured for three avian species: American kestrel, mallard duck, and ring-necked pheasant. Chemical analysis of eggs and fledglings, egg measurement (volume, weight, dimensions, and shell thickness), hatching success, and observation of brood size and fledgling success were measured in samples for RMA and offsite controls. These variables were considered important in evaluating reproductive success considering the typical effects of organochlorine exposure for birds.

Bioaccumulative chemicals have the potential to contaminate tissues at concentrations that may impair the health of the organism or of associated species. Sunflower leaves and flowers were analyzed because these plants are heavily utilized by small birds and invertebrates as a food source. If chemicals are taken up by sunflowers, they may enter the food web and be translocated to different animals and birds. Invertebrate species were analyzed because they are important prey items and may provide exposure pathways from soil to mammals and birds. Invertebrates sampled for chemical residues were grasshoppers and earthworms; the sample mass for aquatic snails precluded chemical analysis.

Table 8-2. Summary of Assessment and Measurement Endpoints (ESE, 1988)

Assessment Endpoint	Measurement Endpoint	Ecological Component
Population success	Occurrence/Distribution	Carnivores, small mammals (mice, moles) mule deer, cottontail rabbit, prairie dog, raptors, waterfowl, shorebirds, wading birds, grasshoppers, aquatic snails, earthworms
	Density/Relative abundance	Cottontail rabbit, prairie dog, raptors, waterfowl, shorebirds, wading birds, grasshoppers, aquatic snails, earthworms
	Age class	Prairie dog
	Reproductive effects	
	Residues	American kestrel, mallard, ring-necked pheasant
	Egg measurements	American kestrel, mallard, ring-necked pheasant
	Hatching success	American kestrel, mallard, ring-necked pheasant
	Brood size	American kestrel, mallard, ring-necked pheasant
	Fledging success	American kestrel, mallard, ring-necked pheasant
	Male-female ratio	Ring-necked pheasant
	Phenology	Vegetation
	Total cover, height, density	Vegetation
	Species richness	Vegetation
Adverse effects due to exposure	AChE activity	Mallard, ring-necked pheasant, prairie dog, cottontail rabbit, fortuitous samples
	Behavior	Incidental observations during field activities
	Morphology	Incidental observations during field activities
	Physical condition	Incidental observations during field activities
Adverse effects due to residue concentrations	Chemical analysis	Aquatic macrophytes, sunflower, morning glory, earthworms, grasshoppers, American kestrel, ring-necked pheasant, mallard, mule deer, prairie dog, cottontail rabbit, fortuitous samples (i.e., large raptors, badger, coyote)

Contaminant concentration in tissue was chosen as one of the measurement endpoints because of historical correlations of mortality with pesticides in tissue and because many of the contaminants of concern were bioaccumulative. Concentrations of the major contaminants of concern in tissue were monitored in many of the ecological components. For each species tissues for analysis were selected based on the probable fate of the organism in the onsite food web or because of the organism's status as a game species consumed by humans. Thus, for prairie dogs and other prey species, the carcass as consumed by predators was analyzed (minus head, fur, feet, stomach, and intestines). For animals consumed by humans, muscle or muscle and liver were analyzed. Eggs, fledglings, and some adult birds were analyzed to determine contaminants in avian species. Opportunistic samples were collected from raptors and mammalian predators when dead or dying individuals were observed. Liver and brain samples were collected and analyzed from these species and necropsies were performed to determine the possible role of RMA contaminants in the death of these individuals.

Acetylcholinesterase (AChE) activity was measured because some of the chemicals historically produced at RMA were cholinesterase inhibitors. Contaminant analysis was performed on tissues from these animals as an attempt to correlate contaminant concentrations with observed effects.

Qualitative measurement endpoints to evaluate adverse effects due to exposure were determined from the organisms collected for contaminant analysis. These endpoints included behavior (e.g., impaired movement), gross morphology (internal and external), and physical condition (e.g., presence of normal body fat, emaciated appearance).

Comments on Problem Formulation

Strengths of the case study include:

- *The history of the site is documented. Chemicals of major concern are identified to provide a focus for the assessment. An effort is made to identify a causal relationship between contaminants and potential adverse responses in the environment. Criteria are presented for the selection of the major chemicals of concern.*

- *Components of the entire ecosystem are considered including individuals, populations, communities, and food webs. Criteria are presented for the selection of particular ecological components upon which to focus the assessment.*

- *A number of measurement endpoints are utilized that consider the connection between the interactions of the ecological components and the chemical properties of the stressors. A diversity of endpoints is utilized at a number of ecological levels, including tissue concentrations, biomarkers, and population surveys. This wide diversity of endpoints provides a holistic examination of the ecosystem, lending greater confidence in risk estimates.*

Comments on Problem Formulation (continued)

Limitations include:

- *Data on chemical concentrations in environmental media were only available for Phase I sampling. The potential for synergistic effects to occur among compounds is not addressed. Because the list of contaminants studied is narrow, the question is raised that some other contaminants may have been missed.*

- *Details on the life history, habitat, feeding behavior, etc., of the selected ecological components are not presented. Vegetation could not be better assessed due to physical disturbances on the site.*

8.3.2. Analysis: Characterization of Ecological Effects

No single reference (control) area was located that was comparable to RMA with respect to vegetation types, areal extent, and land use (e.g., absence of hunting and grazing pressure). As a result, different, smaller reference areas were selected to meet specific sampling needs. For example, the Wellington Wildlife Refuge, more than 50 miles north of RMA, was selected for collecting control samples of mallard, mallard eggs, cottontail rabbits, mule deer, and prairie dogs for chemical analyses, but land adjacent to Barr Lake, only 5 miles north of RMA, was used for collecting control samples of earthworms and conducting population studies of earthworms in order to sample within the same soil type. Other reference areas were selected for similar reasons of appropriateness or species availability. Site characteristics used in selecting reference areas included: (1) distance from RMA (distances varied, depending on the mobility of the taxa involved, to ensure that sample organisms from reference areas would not have been potentially exposed to RMA contamination); (2) similarity in general vegetation type(s); (3) similar land uses (e.g., protection from hunting and/or grazing); (4) similar topography (e.g., plains, not mountains or foothill terrain); and (5) similar soil type(s). Different combinations of these features were used to select reference areas, depending on the objectives of the sampling to be conducted.

Ecological effects were characterized by the following methods: (1) pathways analysis for criteria development; (2) potentially lethal tissue concentrations; (3) depression in brain AChE activity; (4) reproductive success for American kestrel, mallard, and ring-necked pheasant; and (5) population studies. Population studies (i.e., species occurrence, population density, and age-class structure) were considered qualitative or semiquantitative indicators of stress response. Many of these characteristics were not considered quantitative indicators of chemical stress because they can be influenced by so many factors other than contaminant concentrations (e.g., habitat, prey availability, noncontaminant-related human disturbance).

<u>Pathways Analysis for Criteria Development</u>. Characterization of ecological effects for the major contaminants of concern was based on comparison of observed concentrations in abiotic media and biota with estimated site-specific soil, water, and sediment criteria and the chemical-

specific AWQC. Site-specific soil, water, and sediment criteria were estimated by examining potential exposure pathways, a method termed "Pathways Analysis" (ESE, 1989; Fordham and Reagan, 1991). The criteria can be applied in risk characterization by comparing acceptable concentrations developed from the model with observed concentrations in the tissues of species within selected food webs.

The pathways model was developed as a new tool to establish quantitative relationships between contaminant concentrations in abiotic source media (e.g., soil, sediment) and in biota. The model was modified from a single food chain model proposed by Thomann (1981) to include multiple food chains in the food web of key species. The model incorporates estimates of exposure of various organisms to specific chemicals in the environment through the mechanisms of bioconcentration (concentration from direct exposure to water in an aquatic environment), bioaccumulation (concentration from diet plus bioconcentration), and biomagnification (the increase in concentration as chemicals move along food chains to higher trophic levels). Direct exposure pathways are also considered. Contaminant concentrations in soil or sediment are linked with water by the soil-water partition coefficient normalized for organic carbon.

The pathways approach looked at potential lethal and nonlethal forms of biological injury associated with residue concentrations of the contaminants of concern, as well as direct exposure. Some of these nonlethal effects were known at the level of the individual organism (e.g., abnormal behavior, reduced fledgling success in waterfowl) and evaluated at the population level as described in subsequent sections. Thus, the pathways approach was able to establish and quantify the exposure pathway, and field sampling for adverse effects was conducted to verify the effect. The model approach for quantifying exposure pathways was applied to the terrestrial and aquatic ecosystems on RMA and is adaptable to any ecosystem where basic information is available both on the species present and the trophic relationships among organisms.

Each potential exposure pathway was evaluated (i.e., ingestion of surface water, ingestion of prey items) and a criterion for contaminant concentration was developed protective of that exposure pathway. For a given chemical, all exposure criteria were compared, and the lowest criterion was selected as protective of sensitive species and sensitive pathways.

Criteria development served several purposes. It served to establish a relationship between observed tissue concentrations in biota and concentrations in abiotic media that were below detection and therefore unmeasurable. The criteria also served to indicate a level of contamination in the environment that would probably not cause adverse effects. This information was used to infer where adverse effects might be occurring without directly measuring them. The usefulness of this approach was confirmed by those instances where observed adverse effects (i.e., avian mortality) were correlated with tissue concentrations in affected species. Initial acceptable site-specific abiotic media concentrations developed by this method are presented in table 8-3.

Several exposure pathways were evaluated to obtain criteria; these were as follows:

1. Exposure as a result of surface water ingestion was considered an important exposure pathway. Toxicity data obtained from the available literature for terrestrial organisms were examined for LOAELs and NOELs for an oral exposure route. The most

Table 8-3. Acceptable Concentrations of Major Contaminants of Concern in Abiotic Media (ESE, 1988)

	Water (ppb)	Sediment (ppm)	Soil (ppm)
Aldrin/dieldrin	0.034	0.0055	0.10
Arsenic	100	15	52
DBCP	60	0.086	6.10
Endrin/isodrin	0.032	0.0019	9.2
Mercury	0.004	0.004	1.1

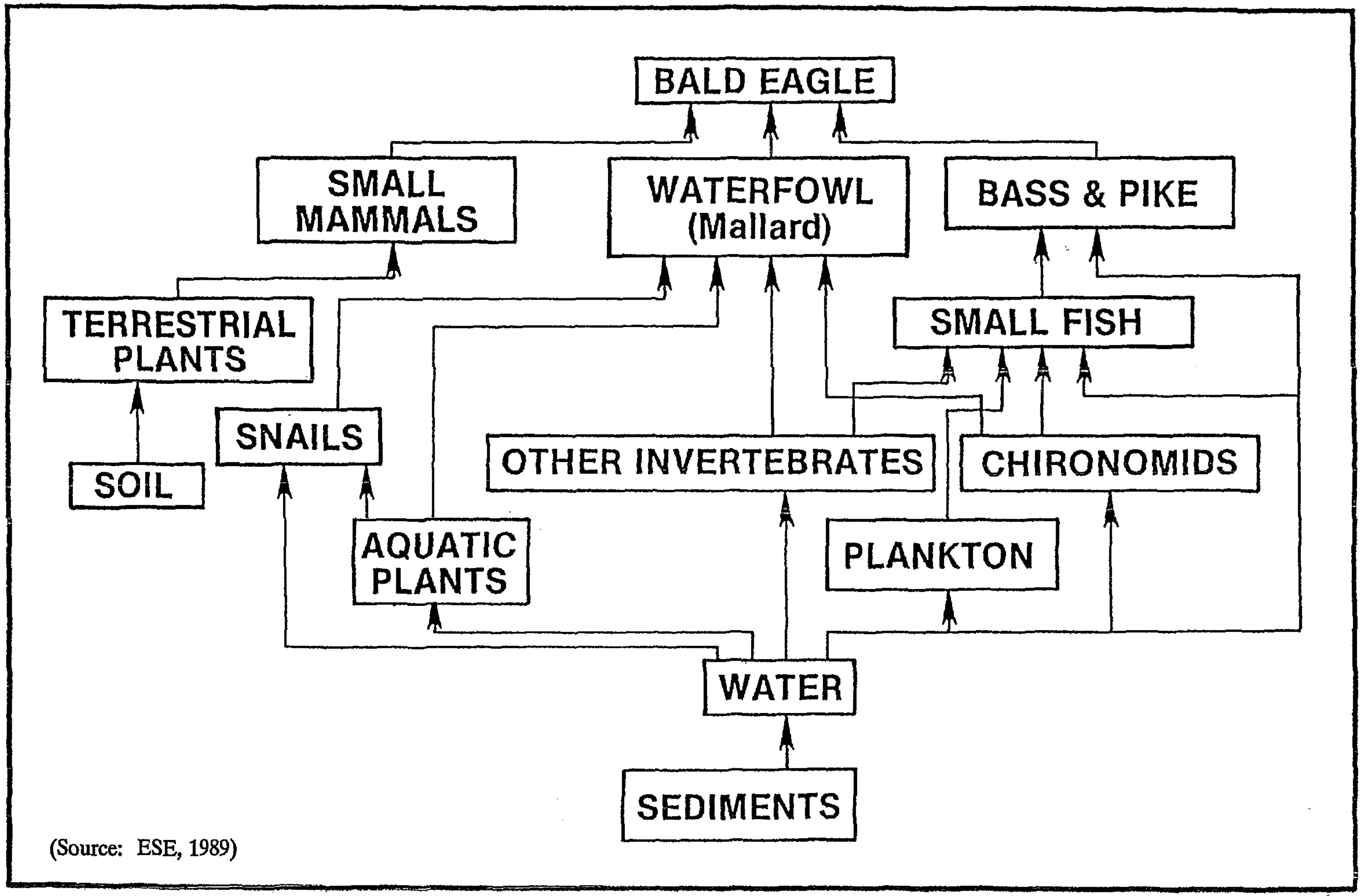
sensitive (lowest) toxicity value (in mg/kg bw/day) was converted, if necessary, to a water concentration (mg/L) by dividing by daily water intake (L/kg bw/day) for the species for which the toxicity value was available. The water concentration was then divided by uncertainty factors based on data quality (appendix A). Sediment criteria were developed from the water criteria by multiplying the chemical-specific soil-water partition coefficient (K_{oc}) and the fraction of organic carbon (f_{oc}) by estimated water criteria (appendix A). This resulted in abiotic criteria specific to toxicity as a result of ingestion of contaminated media.

2. Exposure to contaminants as a result of ingestion of diet was considered. A bioaccumulation model (Thomann, 1981) was adapted to reflect an entire aquatic "sink" food web, as opposed to single food chains. A sink food web is a subset of a community food web that includes all organisms consumed by the "sink" or top-level species (Cohen, 1978) (figure 8-3). Total bioaccumulation in the food web (total BAF) was estimated by modifying Thomann's model by applying a percent diet factor to each compartment of the food web (Fordham and Reagan, 1991). A maximum acceptable tissue concentration (MATC) for each contaminant was derived from the available literature and divided by total BAF to obtain a water concentration (the sink food web initiated with water; therefore, the BAF for the sink species reflects the amount of contaminant in water transferred through a food web) (Fordham and Reagan, 1991). Sediment criteria were obtained from water criteria by applying the K_{oc} and f_{oc}. Uncertainty factors were not applied to the resulting criteria because very conservative assumptions were made during the modeling process.

3. Toxicity to aquatic life by direct contact was determined by comparison with AWQC. When AWQC were unavailable or not applicable because they were based on human consumption of aquatic life as an endpoint, LOAELs and NOELs for aquatic life were obtained from the available literature. Uncertainty factors were applied based on the data quality (appendix A).

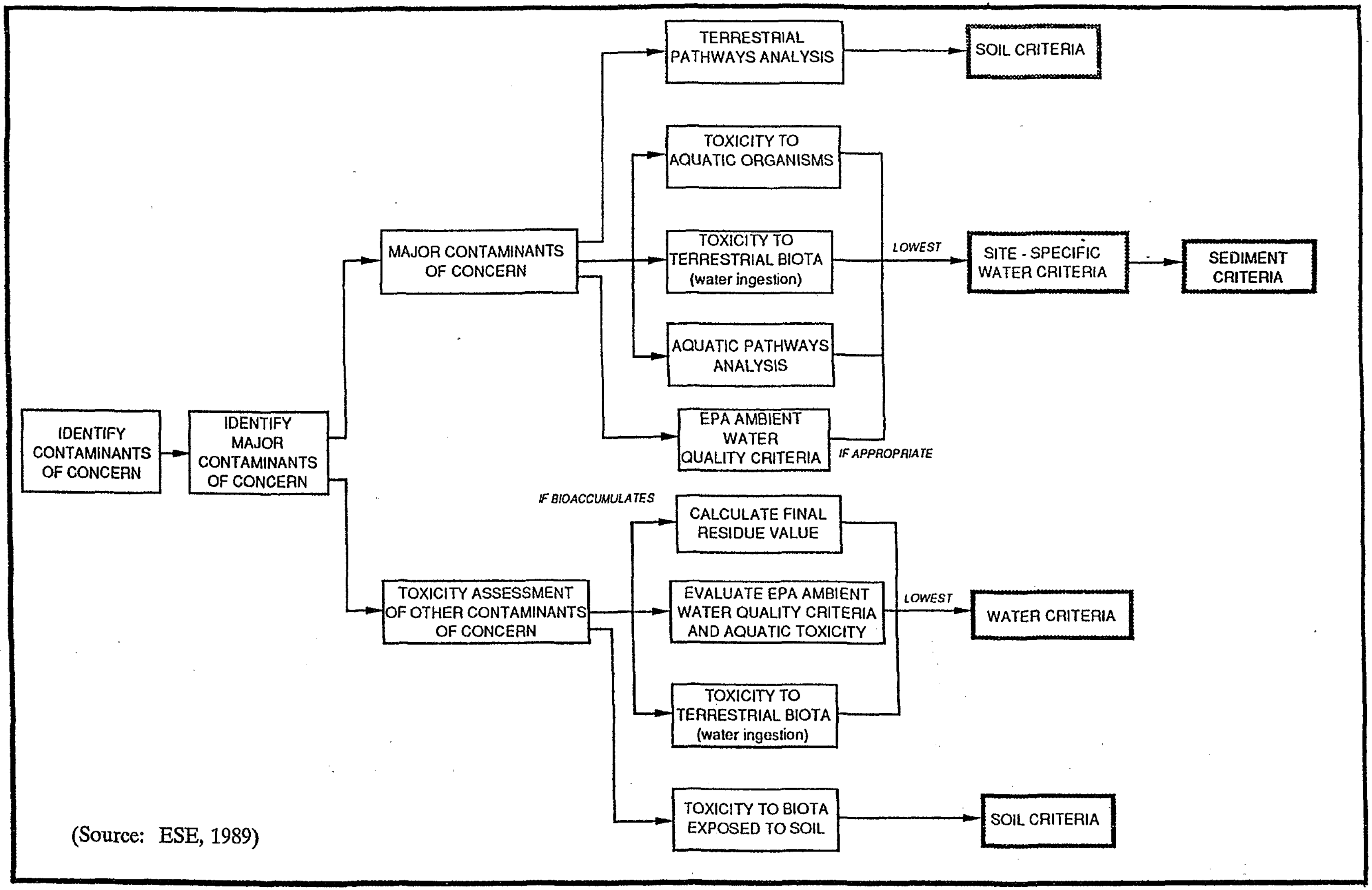
4. Soil criteria were developed by calculating bioaccumulation factors in a terrestrial food web. For example, a sink food web consists of multiple food chains. Bioaccumulation factors for species in each single food chain (i.e., plants, mammals, bald eagle) were identified from the literature. The bioaccumulation factors were multiplied to obtain a BAF for each individual food chain. A total BAF was calculated for the sink species by adjusting each pathway-specific BAF by the relative proportion in the sink species diet and summing the adjusted factors. Total BMF from all food chains was divided into an MATC for the sink species to obtain soil criteria. In addition, LOAELs and NOELs were considered for terrestrial soil fauna based on exposure by direct contact and soil ingestion.

Criteria for each of the major contaminants of concern developed for each exposure pathway above were compared to each other (figure 8-4). The lowest criterion for water, sediment, and soil was selected to represent the acceptable level in the environment that was unlikely to pose a threat to wildlife populations.



(Source: ESE, 1989)

Figure 8-3. Bald eagle food web for contaminants in soil, sediment, and surface water source



(Source: ESE, 1989)

Figure 8-4. Criteria diagram

Lethal Toxicity. While field sampling focused on selected species from predetermined areas, animals found dying or recently dead were collected as "samples of chance." Particular attention was paid to raptors because of their position at the top of site food webs and the known avian toxicity of some RMA contaminants (e.g., organochlorine pesticides). Necropsies were performed on carcasses to determine the possible cause of death and to document the condition (e.g., emaciated) of each animal. For specimens found dying, the condition and behavior of each were recorded as additional clues to the animal's death. Samples of liver and brain tissue were collected and analyzed for contaminants.

Lethal dieldrin levels in brain tissue of birds have been reported in the range of 3 to 4 mg/kg (Robinson et al., 1967; Belisle et al., 1972). Three raptors species (ferruginous hawk, red-tailed hawk, and great-horned owl) were found dead on RMA with levels of dieldrin in the brain within this range. One great-horned owl, found in an emaciated condition, had dieldrin concentrations of 9.32 mg/kg in liver tissue and 27.7 mg/kg in brain tissue.

Brain Acetylcholinesterase. Brain acetylcholinesterase (AChE) levels were determined from various species collected for contaminant analysis. Organophosphates and some metal ions are known cholinesterase inhibitors. AChE activity can be difficult to interpret as a bioindicator because this enzyme is also influenced by population variability, diurnal cycles, metabolic function, and postmortem changes. Levels of AChE activity were measured in brain tissue from opportunistically collected birds and mammals (if the specimen was considered fresh), rabbits, prairie dogs, mallards, ring-necked pheasants, and American kestrels. AChE activities in animals from contaminated areas were considered to be significantly depressed if there was 20 percent or greater depression in activity as compared to control animals (Robinson et al., 1988).

Avian species from RMA (mallard, ring-necked pheasant) had brain AChE levels that were similar to controls (ESE, 1989). One mourning dove, two golden eagles, and three red-tailed hawks found dead on RMA had activities slightly elevated with respect to normal values reported in Hill (1988). It is unknown whether postmortem changes may have influenced enzyme activity because control specimens dead at the time of collection were not available.

Prairie dogs from two areas on RMA had AChE activity significantly lower than those on onsite and offsite controls ($p<0.01$), and even onsite controls were slightly depressed with respect to levels in offsite controls ($p<0.05$). Prairie dogs from only one site, the Toxic Storage Yard (figure 8-2), had significantly depressed levels of AChE. The pattern of AChE depression suggests contaminant-related effects; however, the effect was not correlated with soil concentrations of known organic AChE inhibitors. Several inorganics were elevated in soils from the areas where depressed AChE activity was observed. Arsenic compounds (arsenite ion and to a lesser extent arsenate) are linked with cholinesterase depression in fish (Olson and Christensen, 1980). Other metal ions have also been linked to cholinesterase inhibition (Tomlinson et al., 1981). The metal ions involved appeared to be naturally occurring soil constituents and not RMA contaminants. No differences between contaminated and control areas were observed in brain AChE in cottontail rabbits.

Cholinesterase inhibition is an appropriate measurement endpoint for contaminants that are known or suspected AChE inhibitors. However, current data on chemical contaminants in abiotic media at RMA indicated that the occurrence of potential AChE inhibitors on RMA was limited.

Avian Reproductive Success. American kestrel reproductive success has been studied at RMA for several years (ESE, 1989). The 1986 survey indicated a much higher nesting success than in USFWS studies in 1982 and 1983 (38 percent nests fledged in 1982, 50 percent in 1983, and 71 percent in 1986); 1986 data for RMA were not significantly different from controls (ESE, 1989). Reproductive parameters considered were percentage nests hatched and fledged, and mean number of young hatched per nest.

Collected eggs of American kestrels, ring-necked pheasants, and mallards were measured for weight, volume, dimensions, and shell thickness. There were few differences between eggs from RMA and control sites. Kestrel eggs from RMA were slightly larger than controls, and pheasant eggs from RMA averaged smaller and lighter than controls. Kestrel mean shell thickness did not differ from controls (ESE, 1989).

Average pheasant brood counts were lower for RMA than for offsite controls (average young seen per transect: 1.1 on RMA, 3.0 offsite) (ESE, 1989). Total hens and clutch sizes were also smaller on RMA.

Population Studies. There were no detectable differences for vegetation between RMA and the offsite controls that could be attributable to contaminant-related effects. Extensive physical disturbance (e.g., mowing, herbicidal weed control, disking, reseeding, and burning) continued as part of normal maintenance activities. These activities and soil compaction occurred in both uncontaminated and contaminated areas of RMA, making it difficult to isolate effects that could be attributed to RMA contaminants.

The aquatic snail data indicated a high degree of variability among sites and between years. The covariates of vegetation (substrate) weight, temperature, and pH indicated that these factors influenced the results. The grasshopper data also did not indicate any differences that were attributable to contaminant effects. The contaminated sites had decreased species richness, but also had decreased plant diversity. Population comparisons for earthworms indicated significant differences between onsite and offsite controls.

The numbers of diving ducks and coots were higher at RMA than at control sites, but the number of dabbling ducks and geese was lower. The most striking difference was the complete lack of mallard broods at RMA. Only two mallard nests were located on RMA.

Significantly higher prairie dog adult-young ratios were observed offsite than on RMA, and density appeared lower onsite than offsite. However, colonies from contaminated areas did not differ significantly from colonies from uncontaminated areas onsite. Cyclic population fluctuations, past management practices, predation, and habitat suitability are other factors that could influence prairie dog populations. At RMA, many of the prairie dog colonies have a high percentage of cheatgrass as opposed to a diverse mix of vegetative species that provide better habitat and food sources.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *Pathways Analysis is a useful tool for quantifying the exposure and potential ecological effects.*

- *Figure 8-4 provides a helpful flowchart on the relationship between stressors, ecological components, endpoints, and acceptable criteria for chemical concentrations in the environment.*

Limitations include:

- *Results are often expressed in general terms when they should be expressed quantitatively and in tables wherever possible.*

General comment/cautionary note:

- *The criteria developed in the course of this study are not recommended for use at other sites because they were developed to be site specific, and more consideration was given to studies with data regarding species similar to those on RMA. The chemical- and media-specific criteria were developed for consideration in the absence of available EPA criteria so that overall risk to aquatic or terrestrial wildlife populations could be evaluated. At this time, the criteria have not been validated to determine if these levels are acceptable for protection of ecological health. However, the pathways modeling approach has proved to be a valuable tool in the assessment of biological risk and for the development of environmentally based remediation criteria.*

- *In addition to considering trophic level when selecting species for tissue analysis, the exposure variable of species movement in the environment should be considered. In order to relate abiotic to biotic data, the species collected for chemical analysis should preferably have a limited home range or limited feeding area. Otherwise, the exposure assessment becomes very uncertain. Migrant species can be included for contaminant analysis (i.e., American kestrel), but consideration should be given either to using life stages that are immobile or to collecting adults after they have been at the site for a sufficient length of time, where a condition approaching equilibrium may be achieved. Aquatic species may not be affected by spatial variables to the extent that terrestrial species are, due to the greater amount of spatial variation observed in a terrestrial environment as opposed to an aquatic one.*

8.3.3. Analysis: Characterization of Exposure

A full characterization of chemical exposure was not completed during the Biota Remedial Investigation because all abiotic data collected during the remedial investigations were not available. However, toxicity assessments were compiled, ecological component populations were characterized, and exposure pathways were evaluated.

Tissue Concentrations. Biota were observed to contain mercury, arsenic, dieldrin, endrin, and DDT/DDE (ESE, 1989). Dieldrin was the contaminant most frequently detected in biological tissue. Tissue concentrations were obtained for various tissues for different organisms on RMA (appendix B). Samples were collected from several known contaminated areas on RMA, as well as from areas believed to be relatively undisturbed and from offsite controls. Rosenlund et al. (1986) and data supplied by Morrison Knudsen were relied upon to determine contaminant effects in fish and other aquatic life.

Dieldrin was the primary contaminant in eggs from mallards (4.89 mg/kg), ring-necked pheasants, and American kestrels (mean of 0.504 mg/kg) (ESE, 1989) (appendix B). Dieldrin was also the primary contaminant in avian carcasses found on RMA (appendix C). Most hawks and owls found dead on RMA and analyzed for contaminant concentrations had residues of dieldrin in the brain and liver. Brain concentrations of dieldrin from raptors collected dead in an emaciated condition ranged from 0.678 to 9.98 mg/kg (appendix C) (ESE, 1989). Lowest reported lethal brain concentrations of dieldrin in raptors are approximately 3 to 4 mg/kg (Stickel et al., 1969; Belisle et al., 1972; Stickel, 1973); some raptors retrieved dead from RMA with no other known cause of death had brain concentrations within this range.

Exposure Pathways. Biota can be exposed to contaminants in many different ways: (1) direct contact with contaminated surface water or sediment by aquatic life; (2) direct contact with contaminated soil or surface ponding by soil fauna; (3) ingestion of contaminated surface water, soil, or food items by terrestrial organisms; (4) inhalation of contaminated air or fugitive dusts; and (5) dermal contact. For the major contaminants of concern, toxicological literature (Moriarity, 1985) and known distribution and concentration of contaminants in abiotic media indicated that direct contact and ingestion were the exposure routes expected to be the most important for wildlife populations. Observation of the types of species apparently most affected by contamination also indicated the ingestion and direct contact pathways as being the most significant exposure pathways. High-trophic-level birds and mammals were found dead or with high contaminant levels in tissues. Burrowing mammals and soil fauna also appeared to have high tissue levels. Waterfowl and aquatic life, both components associated with ingestion or contact with water or sediments, appeared affected by contaminants.

Inhalation was determined to be an insignificant exposure pathway based on data provided in the Air Remedial Investigation Report (ESE, 1988). In general, air concentrations were low.

Dermal contact by higher level species such as birds or mammals is difficult to evaluate because the toxicological data and exposure factors for nonhuman biota are lacking. Contaminant transfer during burrowing, grooming, and other activities across an intact dermal membrane is highly uncertain. The presence of hair or feathers may impede dermal uptake by keeping contaminants adsorbed or absorbed to soil away from skin. Chemical properties also influence dermal uptake.

Soil ingestion rate was estimated for small mammals from the available literature. The ingestion rate was compared to the estimated criteria to determine if the criteria would be protective of exposure of small mammals by soil ingestion.

Magnitude, Frequency, and Duration of Exposure. Some areas of RMA, primarily the basin areas (Section 35, 36, and 28) where wastes collected and the Lower Lakes, have been linked to extensive wildlife mortality in the past. The magnitude and frequency were probably more extensive during the time when RMA was an active industrial facility. After 1965, the U.S. Army drained Upper and Lower Derby and Ladora Lakes and removed contaminated sediments. The lakes were then refilled. Recent analyses by Rosenlund et al. (1986) indicated that sediments in the Lower Lakes were still contaminated, but that water concentrations of pesticides were below detection.

Frequency and magnitude of exposure are expected to be dependent on the species in question. For example, American kestrels do not appear to be affected as severely as in the past, but mallards and possibly pheasants continue to exhibit contaminant-related effects. This may be related to the ingestion rate of soil or sediment during pheasant or mallard feeding.

American kestrels did contain pesticides, as did many other predators collected on RMA. Predators are at risk of contaminant effects when the contaminants of concern are bioaccumulative, such as the organochlorine pesticides and mercury. For contaminants that are not bioaccumulative (e.g., arsenic and DBCP), organisms that come into direct contact with the contaminant (e.g., terrestrial plants, invertebrates, or aquatic life) are most likely to be at risk.

Some herbivores were relatively free of residues. Deer and rabbits had lower residues than prairie dogs. This may be related to the frequency and magnitude of exposure. Prairie dogs live in close contact with soil, spending a large portion of their daily time below ground or involved with burrowing activities. Grooming would be a likely source of frequent exposure due to ingestion of soil particles. Prairie dogs have vertical as well as horizontal movement and may be exposed to chemicals deep in the soil.

Consideration of a species' feeding habits and activity patterns is important in defining exposure. Species that are sedentary in their habits will have a longer duration of exposure than free-ranging species. Animals with a small home range are thus more likely to be affected by contaminant exposure than ones that may move in and out of contaminated areas for feeding. Prairie dogs, a sedentary species, thus may provide a better index of site-specific exposure than mule deer or rabbits, which may range over more territory. This is illustrated by the contaminant concentrations in prairie dogs compared with other mammals with approximately the same food habits. However, other critical variables may interact besides population movements. Prairie dogs are likely to have a higher soil ingestion intake due to the extensive burrowing activities performed by these animals. Avian species are mobile in the environment, and thus their exposure is a result of a wide range of prey and contaminant levels in prey. For this reason, it is important to consider population variables such as home range and preferred feeding areas in addition to feeding habits and activity patterns when determining risk to wildlife populations from hazardous chemicals in the environment.

Fate and Transport. Most of the major contaminants of concern to biota were environmentally persistent chemicals with bioaccumulative properties. Aldrin, dieldrin, endrin, isodrin, and mercury all accumulate in tissues. Isodrin is metabolized to endrin in tissue, and aldrin is metabolized to dieldrin. Some aldrin was detected in grasshoppers collected from Section 26 (appendix B), but this might have been a result of dermal contamination from surface soils, and not actual tissue contamination. Samples were not washed prior to chemical analysis because analysis was intended to represent a consumer organism's exposure level.

Dieldrin leaches slowly from soils; the bulk of depletion from soil is a function of volatilization rather than uptake, degradation, or runoff (Beyer and Gish, 1980). Like other organochlorine pesticides, it is relatively insoluble in water. In terrestrial species, invertebrates tended to have higher dieldrin concentrations than plants. Avian species had dieldrin levels slightly lower than those in invertebrates.

A major biological fate of DDT is metabolism to DDE; DDE but not DDT occurred in waterfowl, pheasants, and raptors (appendix B).

Endrin is persistent in the environment, but is less stable than either aldrin or dieldrin. It is subject to microbial degradation, particularly under anaerobic conditions such as occur in saturated soils. Bioaccumulation of endrin is less than for dieldrin. Endrin was not detected as often as dieldrin (appendix B).

Inorganic arsenic forms relatively insoluble complexes in soil, binding to hydrous oxides on clays or cations in the soil solution (Woolson, 1983). It occurs in several chemical forms that affect its bioavailability and transport. In general, the more water-soluble forms occur in areas that receive little rainfall (Woolson et al., 1971) and are likely to enter surface waters as runoff. In sandy soils, such as occur at some locations on RMA, arsenic can leach into ground water, although this generally occurs in shallow surface layers. In aerobic soils, arsenic occurs predominantly in the arsenate form (Woolson, 1983). Arsenic is not highly bioaccumulative and tends to be metabolized quickly and excreted. Arsenic was detected in plants and invertebrates and may have been a result of surface contamination, as samples were not washed prior to analysis. Arsenic was detected infrequently in vertebrates. Concentrations and frequency of detection in pheasants from RMA did not differ significantly from controls.

Mercury in sediments tends to be in the inorganic form (Snarski and Olson, 1982). Because microorganisms are capable of converting inorganic and organic mercury compounds into highly toxic methylmercury and dimethylmercury, any form of mercury in the environment is hazardous. The synthesis of methylmercury by bacteria in sediments and water is the major source of methylmercury in aquatic environments (Boudou and Ribeyre, 1983). Mercury, although mobile and bioaccumulative in aquatic ecosystems, occurred infrequently in terrestrial species. The occurrences in terrestrial species were in higher level avian predators, and in invertebrates and mammals in direct contact with soils. Mercury did occur in species such as mallard and other waterfowl that feed within the aquatic food web.

DBCP was not a target for analysis because it is metabolized rapidly and is not expected to occur in tissue except within hours or several days following exposure. It is slightly soluble in water and is mobile in the environment, migrating from soil to ground water. It is highly persistent and decomposes slowly by hydrolysis or microbial action (U.S. EPA, 1987). Movement of DBCP in soils is greatest in soils with a coarse texture and low organic content (Bigger et al., 1984). DBCP is metabolized to metallic bromides in plants.

> ***Comments on Analysis: Characterization of Exposure***
>
> *Strengths of the case study include:*
>
> - *The case study links the ecology of ecological components to their potential for exposure to chemicals, and considers the magnitude, frequency, and duration of exposure.*
>
> *Limitations include:*
>
> - *The observations and conclusions drawn would be more strongly substantiated by quantification of exposure.*

8.3.4. Risk Characterization

The risk characterization evaluated the data from the ecological components and compared estimated exposure concentrations with critical values from the toxicity assessments. The risk characterization could not be completed because the abiotic data base was incomplete. However, certain aspects of risk characterization that could be evaluated were the following:

- observed populational effects onsite compared to controls;
- tissue concentrations that exceeded critical values from the literature; and
- development of site-specific criteria to be evaluated and/or applied in later phases of the ongoing investigation.

These aspects were utilized in deriving a relationship between exposure and adverse effects. What could not be completed was a location-specific evaluation of ecological risk or definitive remediation goals. In order to have completed the risk assessment, abiotic media data would have needed to have been finalized.

<u>Relationship Between Exposure and Adverse Effects</u>. As part of the ecological effects characterization, assessments were made to determine the relationships between exposure concentrations and adverse effects. Toxicological effects as a result of exposure, lethal and

sublethal concentrations from laboratory studies, and data from field studies were compiled. Populational effects were initially described in the ecological characterization. Populational effects resulting from exposure were determined by comparison with controls that were unaffected by chemical hazards found at RMA.

There did not appear to be any exposure-related impacts on plant communities. There were statistically fewer aquatic snails in the RMA lakes than in offsite lakes ($p > 0.001$), although conclusions regarding a relationship with exposure could not be drawn because statistical analysis also indicated that vegetation, pH, and temperature were covariates that may have confounded results. There were no significant differences for grasshopper populations, but earthworm populations were significantly different. However, when residues were analyzed statistically, there were significant differences only for arsenic, which was higher in controls. Thus, for all lower level (plant and invertebrate) ecological data, weight of evidence fails to support contaminant-related impacts. However, mallard ducks had apparently reduced reproductive success. The reproductive success of kestrels was improved over data from previous studies. Reproductive success was compared to analytical data for mallards and kestrels collected at RMA. Organochlorine concentrations in a mallard egg exceeded critical levels from the toxicological literature of >1 mg/kg (Blus, 1982) that would be indicative of poisoning. Conversely, mean concentrations of organochlorines in American kestrel eggs (0.504 mg/kg) were less than the critical level in eggs (ESE, 1989).

Lethal and sublethal concentrations of the contaminants of concern in tissue were of particular importance because tissue concentrations were a major endpoint for this study. The relationship between exposure and adverse effects was documented for the basin areas and the Lower Lakes by the widespread occurrence of contaminant levels in the tissues of specimens from RMA. Lethal levels in brain tissue were approached or attained in raptors collected from RMA. Often, birds were in an emaciated condition, indicative of organochlorine pesticide poisoning. Arsenic levels in leaves of sunflowers were in the range of phytotoxic concentrations observed in other plants. Obvious signs of phytotoxicity were not observed in these plants, however. It is possible that surface dust may have caused the observed concentrations. Levels of the organochlorine pesticides in grasshoppers were high enough to be a potential hazard to avian consumers based on comparison with critical levels from the toxicity assessment. Arsenic in grasshoppers was also high enough to present a potential hazard to insectivores preying on grasshoppers.

Criteria were estimated for each of the potential exposure pathways (table 8-3). These pathways included dietary exposure (bioaccumulation), direct contact by soil and aquatic organisms, and surface water ingestion. Observation of the types of species apparently most affected by contamination indicated the ingestion and direct-contact pathways as being the most significant exposure pathways. The Pathways Analysis method of criteria development is under further investigation at RMA to determine levels in the RMA environment that would not pose a threat to ecological health. A calibration/validation process is being utilized to reduce model uncertainty. The criteria provide a means by which to estimate risk; they are not risk estimates. Comparison of the criteria with exposure data would result in a hazard quotient.

Depending on how the pathways model is applied, the criteria are protective of populations, or, in the case of endangered species, of individuals. For instance, if average data for populations (i.e., average BCFs for different species, average loss rates, average LOAELs) are applied, then the model will be protective of populations. The model can deliver more conservative results by altering the parameters to reflect an individual response or by applying uncertainty factors.

Evidence Linking Exposure and Adverse Effects. Comparison to controls indicated significantly higher tissue concentrations for some species collected from RMA (table 8-4). The analytes sampled were all site-related contaminants, and the significance levels were quite low for some analytes and species. Statistics were not possible for samples of chance, or for species where too few animals were collected. Nonparametric statistics were used due to heteroskedasticity in the data and small sample sizes (ESE, 1989).

Other evidence linking exposure with adverse effects is presented in appendix C. These raptors were collected dead, and chemical analysis and autopsies were performed. In some cases, the emaciated condition of the bird and the levels of organochlorine pesticides are indicative of organochlorine pesticide poisoning.

Sources of Uncertainty. There are many sources of uncertainty inherent in the analysis of the nature and extent of contamination. There is analytical uncertainty, although Program Manager for Rocky Mountain Arsenal protocols specify that the Certified Reporting Limit is the 95 percent confidence interval for the method. In the implementation of a field sampling design for a large area, additional uncertainties occur. Due to the size of the study site, there is spatial uncertainty. The extent to which the samples collected actually represent the population of potential contaminant detections is unknown. For example, prairie dogs collected from onsite control areas that were supposedly undisturbed contained dieldrin in the liver. Subsequent sampling confirmed the presence of dieldrin in surficial soils in the area. Uncertainty is also inherent in estimates of biological fate of contaminants. Bioavailability of contaminants in soil or surface water may be highly variable, as may bioconcentrations, bioaccumulations, or depuration.

Animals that are mobile in the environment can alter the actual levels of contaminants to which they are exposed, such that some populations may be highly exposed while others have minimal exposure. Therefore, determining representative and worst-case exposure concentrations over the area of the site is important to determining overall risk to wildlife populations. Uncertainty can be reduced by emphasizing in contaminant analysis those animals that are in close contact with their environment and tend to have a limited home range. Sampling avian juveniles and eggs seemed to provide a better distinction between control and contaminated sites than did the sampling of adults, although with sampling juveniles of any species, growth dilution of tissue contaminant load becomes more questionable than when sampling adults.

There is also uncertainty in the measures of ecological effects. The AChE depression observed in prairie dogs could not be explained from data available during the Biota Remedial Investigation, although the effect was statistically significant compared with control locations. Other stressor-response relationships were more clear, such as the presence of dieldrin in organisms from areas of historical dieldrin contamination. Species-to-species extrapolation (such as utilizing toxicity data from one species to address risk to others) is as uncertain as a factor of 5 or

Table 8-4. Summary of Species Where Contaminants of Concern Differed Significantly ($p \leq 0.05$) or Approached Significance Between Controls and Contaminated Areas Onsite (ESE, 1989)

Species	Analyte	Level of Significance
Earthworms	arsenic	$0.05 \geq p > 0.01$
Grasshoppers	arsenic	$0.10 \geq p > 0.05$
	dieldrin	$0.01 \geq p > 0.001$
	aldrin	$0.10 \geq p > 0.05$
Kestrels		
egg	dieldrin	$0.01 \geq p > 0.001$
juvenile	dieldrin	$0.05 \geq p > 0.01$
Mallards		
egg	dieldrin	$p \leq 0.001$
juvenile	dieldrin	$0.05 \geq p > 0.01$
	mercury	$0.05 \geq p > 0.01$
adult	dieldrin	$0.10 \geq p > 0.05$
Pheasant		
egg	dieldrin	$p \leq 0.001$
juvenile	dieldrin	$0.10 \geq p > 0.05$
Cottontail	dieldrin	$0.05 \geq p > 0.01$
Prairie dog	arsenic	$0.10 \geq p > 0.05$
	dieldrin	$p \leq 0.001$

more. Not only does toxicity differ between species (i.e., LC_{50} and LD_{50} data), but in field situations, the animals' behavior and life history can cause large differences in exposure rates.

Uncertainty in the bioaccumulation exposure pathway, which is one pathway in the Pathways Analysis method used to estimate site-specific criteria, was quantified with Monte Carlo analysis. The parameters in the bioaccumulation model were either considered to be fixed, or they were assigned a distribution, as the available data indicated. Monte Carlo analysis then performs numerous iterations by randomly selecting points from the parameter distributions. The ultimate result of the analysis is based on the variability in the parameter distributions and provides a quantitative measure of uncertainty surrounding the modeling results. The amount of uncertainty in the model to estimate criteria protective of contaminant effects resulting from bioaccumulation varied with the contaminant and the availability and quality of the data for each contaminant. For example, data were available for parameters of depuration bioconcentration (BCF) for the aquatic food web model, whereas assimilation efficiency data were frequently lacking for wildlife species. Bioaccumulation (BAF) and dietary proportion were also available from the literature.

Uncertainty in evaluation of the direct exposure pathways was quantified in a manner similar to assessment of uncertainty in human health risk assessment (appendix A). Uncertainty factors were applied to literature data in order to derive criteria. If data for acute lethality only were available, the uncertainty to derive an acceptable concentration was higher. These uncertainty factors may ultimately produce criteria that are overly conservative; criteria resulting from this process should not be considered absolute.

Additional sources of uncertainty include the possibility that sensitive species have been replaced by tolerant organisms during the decades of contamination at RMA. It is also possible that additional effects resulting from the combined actions of other contaminants have occurred, although there is currently no basis for evaluating this possibility. The evaluation of a suite of measurement endpoints at the individual, population, and ecosystem levels was an important feature of this assessment that helped reduce its associated uncertainties.

Comments on Risk Characterization

Strengths of the case study include:

- *The case study established protective criteria for the ecosystem as represented by selected ecological components. The study addressed uncertainties throughout the assessment, including discussion of potential confounding factors and use of Monte Carlo analysis.*

- *Toxicological and ecological techniques were used in an integrated manner so that a large and complex ecosystem was evaluated for chemical impacts. The methods used to evaluate those impacts can be readily adapted to other sites and*

Comments on Risk Characterization (continued)

other types of chemical impacts. The pathways methodology for relating chemical concentrations in abiotic media to concentrations and effects in biota and the logic used to select indicator species, or species for tissue analysis, can also be applied to other sites.

Limitations include:

- *The study did not complete the integration of effects and exposure data so that ecological risks could be estimated. The study is ongoing, and these activities will be completed.*

General comments:

- *Although the Biota Remedial Investigation contains many of the elements required to perform an ecological risk assessment, the focus of the study was the nature and extent of contamination in biological media and quantifying the interrelationships between contaminants in abiotic media and in biota by use of a pathways model. Abiotic data were available at the time of the Biota Remedial Investigation study report for the purpose of selecting study sites. Conclusions on the exposure assessment and assessment of overall risk to biota are continuing with calibration and validation of the model combined with additional biota sampling. A complete risk assessment for any site cannot be completed until the abiotic data are available to the risk assessor, and this is true for ecological or human health risk assessments.*

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APPENDIX A

SUMMARY OF EQUATIONS AND UNCERTAINTY FACTORS

APPENDIX A. SUMMARY OF EQUATIONS AND UNCERTAINTY FACTORS

Equations

Values such as reference doses or slope factors are unavailable for wildlife species. Instead, abiotic concentrations predicted to be safe for the most sensitive or most important species were derived, and assumed acceptable to the less sensitive or less important members of the ecosytem. The toxicological literature was surveyed for the lowest NOELs or LOAELs:

$$\frac{\text{LOAEL OR NOEL (mg/kg bw/day)}}{\text{water intake (L/kg bw/day)}} = \text{acceptable concentration in surface water (mg/L)} \qquad (1)$$

A similar process is applied to derive soil criteria for an ingestion pathway. Soil ingestion rate (kg/kg bw/day) is substituted into equation (1) above to yield soil criterion in mg/kg.

To determine dietary exposure from bioaccumulative contaminants, a model for single food chains (Thomann, 1981) was adapted for an aquatic food web (multiple food chains) by adding a term for the percentage of prey organism in the predator organism's diet, and determining equations by which the multiple food chains could be summed. The model structure is:

		Trophic Levels	
(1) $BCF = Cb/Cw$ where:	Cb = tissue Cw = water	1	(2)
(2) $BAF_2 = BCF_2 + f_2BCF_1$		2	(3)
(3) $BAF_3 = BCF_3 + f_3BCF_2 + f_3f_2BCF_1$		3	(4)
(4) $BAF_4 = BCF_4 + f_4BCF_3 + f_4f_3BCF_2 + f_4f_3f_2BCF_1$		4	(5)

Additional discussion regarding the model as developed by Thomann (1981) can be found in Spacie and Hamelink (1985).

BMF (sum of all BAFs in the food web) was calculated with variations of the following general equation:

$$BMF_i = BCF_i + \Sigma\ f_i\ BAF_{i-1} \qquad (6)$$

BMF is used to imply that results are for consideration of multiple food chains.

When BCF = 0 for organisms whose trophic level exceeds 1 (i.e., terrestrial predators feeding in the aquatic food web):

$$BMF_i = \Sigma\ f_i\ BCF_1 \qquad (7)$$

Equation (7) essentially sums the BAFs. In our food web, this was applied to mallards (trophic level 2) feeding on different components assumed to be trophic level 1. For the purposes of the model, not only plants but invertebrates were assumed to be trophic level 1; this was because the surface area to volume ratio was high, and it was assumed that BCF would make uptake from diet appear negligible for the contaminants of concern for this study. This assumption might not be applicable to other sites.

When BCF = 0 for organisms whose trophic level exceeds 1, BAFs must not be added directly because BCF for the organism is counted every time a pathway they appear in is added to another. Thus, BMF equations vary with trophic level and the structure of the food web:

$$BMF_2 = BCF_2 + \Sigma\, f_2 BCF_1 \qquad (8)$$

$$BMF_3 = BCF_3 + \Sigma\, f_3(BMF_2) \qquad (9)$$

$$BMF_4 = f_4(BMF_3) + f_3(BMF_2) \qquad (10)$$

Equation (8) was applied to bluegill (trophic level 2 in the food web). Bluegill have a BCF component, as well as uptake from the various lower trophic levels. Equation (9) was applied to pike in our food web. Pike were assumed to feed exclusively on smaller fish (represented by bluegill). This assumption is conservative, since small pike might take invertebrates as well. However, at RMA large numbers of alternative food for pike (ducklings, invertebrates) were not observed. Since only one food source was modeled, a "sum of f" term was not used. In other food webs, where trophic level 3 could be modeled with multiple food chains, the equation below is the simplest approximation:

$$BMF_3 = BCF_3 + \Sigma\, f_3 BMF_2 \qquad (11)$$

Equation (10) was used for the bald eagle. Where two food web pathways converged and BCF for the top trophic level was negligible, adding the concentration from each pathway was done by adding the total accumulation factors for the pathways. The bald eagle occurred at multiple trophic levels, reflected by the different numbers.

The food term contains variables that may often be derived from the available literature, although some of these variables must be extrapolated between species, or worst-case assumptions must be made:

$$f_i = \frac{\alpha\, R\, \%}{k_2} \qquad (12)$$

Where: α = assimilation efficiency $\left(\frac{\mu g \text{ absorbed}}{g \text{ ingested}}\right)$

R = dietary intake (g/g bw/day)

k_2 = depuration rate (day^{-1})

% = percentage of prey item in diet of high-trophic-level organism

The equations for the terrestrial food web are less complex, in part due to lack of data for loss and assimilation rates for terrestrial species. Bioaccumulation factors are derived for each species from toxicological studies; chronic, dietary data are preferred since these are more applicable to field studies. For a simple food web of the following species, hypothetical data were used as an example.

Species	Source of BAF	BAF
Plant	[Plant]/[Soil]	2
Earthworm	[Earthworm]/[Soil]	100
Songbird	[Whole Body]/[Diet]	10
Deer mouse	[Whole Body]/[Diet]	10
American kestrel	[Whole Body]/[Diet]	10

In the above example, higher trophic levels have the same laboratory BAF from diet. In a food chain/food web, the difference in dietary exposure is derived as follows:

Food Chain		BAF
(1)	Soil $\xrightarrow{2}$ Plant $\xrightarrow{10}$ Deer mouse $\xrightarrow{10}$ Kestrel	200
(2)	Soil $\xrightarrow{2}$ Plant $\xrightarrow{10}$ Songbird $\xrightarrow{10}$ Kestrel	200
(3)	Soil $\xrightarrow{100}$ Earthworm $\xrightarrow{10}$ Songbird $\xrightarrow{10}$ Kestrel	10,000

If the kestrel fed 100 percent in any food chain, the BAFs above would apply. Given the following dietary proportion, however, a BMF for kestrel from all combined food chains is derived:

Species	Prey	Proportion
Kestrel	Songbird	0.10
	Deer mouse	0.90
Songbird	Plant	0.50
	Earthworm	0.50
Deer mouse	Plant	1.0

For the model, proportions for plants and earthworms for a "diet" of soil are considered 1.0. The proportions are multiplied together and with the pathway BAF to obtain the adjusted BAF; these values can then be summed to obtain a food web BMF. Note that the songbird may provide two food chains to the kestrel.

Food Chain	BAF	Relative Food Chain Importance			Adjusted BAF
		Songbird	Deer mouse	Kestrel	
(1)	200		1.0	0.90	180
(2)	200	0.5		0.10	10
(3)	10,000	0.5		0.10	500
Total BMF					690

Several items of interest may be pointed out. Since the kestrel does not subsist on food chain (3), overall BMF is lower than BAF for food chain (3) only. The diet may have seasonal or annual fluctuations. Thus, when addressing ecological risk, future as well as current projections should be made. If food chain(s) became highly utilized, risk estimates would differ.

Also, use of chronic BAF data negates the need for loss or assimilation data. Careful selection of parameter data is important, however, depending on the food web. If the sink species is a juvenile and not an adult, BAFs for juveniles should be used. Results for growing animals must be interpreted cautiously, since growth dilution produces lower BAFs and diets change with age. Hydroponic solution data may not be applicable to use as BAF for plants, as this may differ from field data.

Site-specific variables may cause the predicted and observed values to differ. High soil moisture, pH, cation exchange capacity, or f_{oc} may alter soil to biota BAFs from literature data. Thus, calibration with site data is recommended.

Finally, bioaccumulation may occur from ingested soil as well as diet. This can be addressed by adding a soil to sink or other species pathway. The proportion of soil is incorporated as with a dietary item.

In order to relate the estimated concentration of contaminants in the food web back to an acceptable concentration in water, a health effects endpoint, the Maximum Acceptable Tissue Concentration (MATC), in mg/kg bw, must be defined from review of the toxicological literature. When divided by the Total BMF (the BMF for the target or top-level species) for the food web, which is unitless or can be considered as L/kg bw, a water concentration is obtained in mg/L:

$$\frac{\text{MATC}}{\text{Total BMF}} = C_w \tag{13}$$

Because the original food web equations began with tissue concentration in relation to water concentration (BCF), the final concentration factor also relates back to water. A sediment concentration can be estimated by assuming partitioning occurs between water and sediment as follows:

$$C_{sed} = C_w \times K_{oc} \times f_{oc} \tag{14}$$

Further details of the modeling can be obtained in Fordham and Reagan (1991) and ESE (1989).

Uncertainty Factors

Uncertainty factors were applied based on the availability and relevance of available data. Uncertainty factors were not applied to the food web model because many conservative assumptions were built into the model. Uncertainty factors were applied to the estimation of surface water criteria (equation 1) from LOAELs and NOELs and intake rates in order to make the criteria protective. The uncertainty factor approach is outlined in table 8-A1.

Table 8-A1. Uncertainty Factors Used in Establishing Acceptable Water Concentrations (ESE, 1989)

Health Effects	Factor Used to Convert Effect to a Chronic NOEL	Factor Applied for Interspecific Variation	Total Uncertainty Factor
Chronic NOEL	--	5	5
Chronic LOAEL	5	5	25
Subchronic NOEL	10	5	50
Subchronic LOAEL	50	5	250
Acute NOEL	100	5	500
Acute LOAEL, LD_{50}	1,000	5	5,000

APPENDIX B

OBSERVED TISSUE CONCENTRATIONS OF CONTAMINANTS IN TERRESTRIAL ORGANISMS

Table 8-B1. Contaminant Levels in Terrestrial Ecosystems—Terrestrial Program Samples (ESE, 1989)

Species	Tissue	Location (Section)	Contaminant Level in parts per million (mg/kg wet weight basis)(Range/mean*)						
			Arsenic (n/nt)	Mercury (n/nt)	Aldrin (n/nt)	Dieldrin (n/nt)	Endrin (n/nt)	p,p-DDE (n/nt)	p,p-DDT (n/nt)
TERRESTRIAL PLANTS									
Morning Glory	Whole Plant	RMA, (26, 36)	<0.250-5.35 (1/5)	BDL (5)	BDL (5)	<0.046-0.084 (2/5)	BDL (5)	NRQ	NRQ
	Whole Plant	RMA Control (20)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	NRQ	NRQ
Sunflower	Flowers	RMA, Basin A	BDL (6)	BDL (6)	BDL (6)	BDL (6)	BDL (6)	NRQ	NRQ
	Flowers	RMA Control (19)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	NRQ	NRQ
	Leaves	RMA Basin A	<0.250-4.51 (4/5) 1.37	BDL (5)	BDL (5)	BDL (5)	BDL (5)	NRQ	NRQ
		RMA Basin C	BDL (1)	BDL (1)	BDL (1)	>0.300 (1)	0.188 (1)	NRQ	NRQ
	Leaves	RMA (19)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	NRQ	NRQ
INVERTEBRATES									
Earthworms	Whole	RMA, South Plants	BDL (1)	<0.050->2.35 (1/2)	BDL (1)	1.93 (1)	BDL (1)	BDL (1)	BDL (1)
	Whole	RMA Control (5)	0.618-1.53 (8/8) 1.03	<0.050-0.245 (2/8)	BDL (7)	<0.062-5.30 (1/7)	<0.080-0.914 (1/7)	BDL (8)	BDL (8)
	Whole	Offpost Control	BDL (2)	BDL (2)	BDL (1)	BDL (1)	BDL (1)	BDL (1)	BDL (1)
Grasshoppers	Whole	RMA Section 26	BDL (4)	BDL (4)	0.046-5.8 (4/4) 1.59	0.496-7.2 (4/4) 2.53	<0.064-1.65 (3/4) 0.528	BDL (4)	BDL (4)
		RMA Section 36	0.905-6.60 (4/4) 3.17	<0.050-0.108 (2/4) 0.058	BDL (4)	0.271-0.446 (4/4) 0.381	BDL (4)	BDL (4)	BDL (4)
		RMA Control (7, 8)	BDL (3)	BDL (3)	BDL (3)	BDL (3)	BDL (3)	BDL (3)	BDL (3)
		Offpost Control	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)
VERTEBRATES									
Mallard	Juvenile Carcass	RMA***	NRQ	<0.050-0.066 (2/3) 0.051	BDL (3)	<0.031-0.522 (2/3) 0.201	BDL (3)	<0.094-0.507 (1/3)	BDL (3)
	Adult Carcass	RMA	NRQ	BDL (8)	BDL (8)	<0.031-4.53 (3/8)	BDL (8)	BDL-0.360 (4/8) 0.239	BDL (8)
	Juvenile Carcass	Offpost Control	NRQ	BDL (6)	BDL (6)	BDL (6)	BDL (6)	BDL (6)	BDL (6)
	Adult Carcass	Offpost Control	NRQ	<0.050-0.061 (1/8)	BDL (8)	BDL (8)	BDL (8)	<0.094-1.02 (2/8)	BDL (8)
	Egg	RMA (1)	NRQ	0.173-0.185 (2/2) 0.179	BDL (2)	3.0-4.89 (2/2) 3.94	BDL (2)	0.606-0.919 (2/2) 0.762	BDL (2)
	Egg	Offpost Control	NRQ	<0.050-0.186 (5/10) 0.068	BDL (10)	BDL (10)	BDL (10)	<0.094-1.35 (6/10) 0.302	BDL (2)
Ring-necked pheasant	Juvenile Carcass	RMA	<0.250-1.82 (3/11)	BDL (11)	BDL (12)	<0.031-1.33 (5/12)	BDL (12)	BDL (11)	BDL (11)
	Adult Carcass	RMA	BDL (4)	BDL (4)	BDL (4)	<0.031-2.92 (3/4) 0.767	BDL (4)	BDL (3)	BDL (3)
	Juvenile Carcass	Offpost Control	<0.250-1.40 (2/11)	BDL (11)	BDL (14)	<0.031-18.6 (1/14)	BDL (14)	<0.094-1.34 (1/12)	BDL (12)
	Adult Carcass	Offpost Control	BDL (2)	BDL (2)	BDL (3)	BDL (3)	BDL (3)	BDL (2)	BDL (2)
	Egg	RMA	BDL (10)	BDL (11)	BDL (11)	<0.031-5.38 (9/11) 1.12	<0.40-0.143 (1/11)	BDL (10)	BDL (10)
	Muscle**	RMA	<0.250-4.07 (2/20)	BDL (20)	BDL (20)	<0.018-0.063 (2/20)	BDL (20)	BDL (20)	BDL (20)
		Offpost Control	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)

Table 8-B1. Contaminant Levels in Terrestrial Ecosystems—Terrestrial Program Samples (continued)

Species	Tissue	Location (Section)	Contaminant Level in parts per million (mg/kg wet weight basis)(Range/Mean*) Arsenic (n/nt)	Mercury (n/nt)	Aldrin (n/nt)	Dieldrin (n/nt)	Endrin (n/nt)	p,p-DDE (n/nt)	p,p-DDT (n/nt)
Ring-necked pheasant	Liver**	RMA	NRQ	NRQ	BDL (6)	<0.018-2.3 (4/6) 0.655	BDL-0.091 (1/6)	BDL-0.44 (1/6)	BDL
		Offpost Control	NRQ	NRQ	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL
	Egg	Offpost Control	BDL (10)	BDL (11)	BDL (11)	BDL (11)	BDL (11)	BDL (10)	BDL (10)
American Kestrel	Juvenile Carcass	RMA	NRQ	BDL (10)	BDL (10)	<0.031-1.01 (5/10) 0.316	BDL (10)	<0.094-0.219 (1/10)	BDL (10)
	Juvenile Carcass	Offpost Control	NRQ	BDL (8)	BDL (8)	BDL (8)	BDL (8)	<0.094-0.733 (1/8)	BDL (8)
	Egg	RMA	NRQ	<0.050-0.405 (8/34)	BDL (33)	<0.031-3.63 (17/33) >0.512	BDL (33)	<0.094-1.25 (1/29)	BDL (29)
	Egg	Offpost Control	NRQ	<0.050-0.057 (1/11)	BDL (11)	BDL (11)	BDL (11)	<0.094-1.04 (2/11)	BDL (11)
Prairie Dog	Carcass	RMA (36) Summer	<0.250-0.741 (2/9)	BDL (9)	BDL (9)	0.233-13.4 (9/9) 2.03	BDL (9)	NRQ	NRQ
	Carcass	RMA (36) Winter	BDL (5)	BDL (5)	BDL (5)	0.119-6.18 (5/5) 1.44	BDL (5)	NRQ	NRQ
	Carcass	RMA, TSY	<0.250-4.22 (1/5)	BDL (5)	BDL (5)	0.064-0.155 (5/5) 0.114	BDL (5)	NRQ	NRQ
	Carcass	RMA Control Summer (19, 20)	BDL (9)	BDL (9)	BDL (9)	<0.031-0.346 (2/9)	BDL (9)	NRQ	NRQ
	Carcass	RMA Control Winter (20)	BDL (5)	BDL (5)	BDL (5)	<0.031-0.096 (1/5)	BDL (5)	NRQ	NRQ
	Carcass	Offpost Control Summer	BDL (9)	BDL (9)	BDL (8)	BDL (8)	BDL (8)	NRQ	NRQ
	Kidneys	RMA, (36) Winter	BDL (5)	<0.10-0.356 (3/5) 0.178	BDL (5)	<0.248-1.54 (2/5)	BDL (5)	NRQ	NRQ
Cottontail	Muscle	RMA, (36)	BDL (7)	BDL (7)	BDL (7)	<0.031-0.092 (3/7)	BDL (7)	NRQ	NRQ
	Muscle	RMA Control (19, 20)	BDL (7)	BDL (7)	BDL (7)	BDL (7)	BDL (7)	NRQ	NRQ
	Muscle	Offpost Control	BDL (7)	BDL (7)	BDL (7)	BDL (7)	BDL (7)	NRQ	NRQ
Mule Deer	Liver	RMA	BDL (14)	BDL (14)	BDL (14)	<0.031-0.187 (1/14)	BDL (14)	NRQ	NRQ
	Liver	Offpost Control	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	NRQ	NRQ
	Muscle	RMA	BDL (14)	BDL (14)	BDL (14)	BDL (14)	BDL (14)	NRQ	NRQ
	Muscle	Offpost Control	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	NRQ	NRQ

* Mean is calculated when 50 percent or more of samples have detectable contaminant levels. If less than 50 percent of samples have detectable contaminant levels, only the range of values are presented. When calculating the mean, values of ½ the detection limit are substituted for samples that are below detection limit.

BDL Below Detection Limit.

n = Number of samples analyzed that contain detectable contaminant levels, nt = total number of samples.

NRQ Not Requested.

** MKE Sample

*** For highly mobile species (mallard, pheasant, kestrel, mule deer) samples were widespread and RMA was evaluated as a whole entity.

Table 8-B2. Miscellaneous Samples: Samples of Chance and USFWS Supplemental Samples (ESE, 1989)

Species	Tissue	Location (Section)	Arsenic (n/nt)	Mercury (n/nt)	Aldrin (n/nt)	Dieldrin (n/nt)	Endrin (n/nt)	p,p-DDE (n/nt)	p,p-DDT (n/nt)
			Contaminant Level in parts per million (mg/kg wet weight basis) (Range/Mean*)						
Blue-winged teal	Liver	RMA Upper Derby	BDL (3)	0.371-1.64 (3/3) 1.07	BDL (3)	0.183-0.281 (3/3) 0.239	BDL (3)	BDL (3)	BDL (3)
	Muscle	RMA Upper Derby	BDL (3)	0.259-0.559 (3/3) 0.391	BDL (3)	0.090-0.164 (3/3) 0.127	BDL (3)	BDL (3)	BDL (3)
Redhead	Liver	RMA Upper Derby	BDL (5)	0.080-0368 (5/5) 0.211	<0.030-0.088 (1/5)	0.307-0.747 (5/5) 0.458	<0.064-0.074 (1/5)	<0.094-0.156 (1/5)	BDL (5)
	Muscle	RMA Upper Derby	BDL (5)	<0.050-0.073 (2/5)	BDL (5)	0.117-0.320 (5/5) 0.203	BDL (5)	BDL (5)	BDL (5)
American Coot	Liver	RMA Upper Derby	BDL (9)	0.300-1.77 (9/9) 1.08	BDL (9)	<0.124-0.693 (8/9) 0.291	BDL (9)	BDL (9)	BDL (9)
	Muscle	RMA Upper Derby	BDL (9)	<0.050-0.339 (8/9) 0.179	BDL (9)	<0.062-1.77 (8/9) 0.53	BDL (9)	<0.940-0.313 (2/9)	BDL (9)
Mourning Dove	Carcass	RMA (35)	BDL (2)	BDL (2)	<0.633-1.83 (2/2) 1.23	5.57-56.3 (2/2) 30.9	<0.800-3.44 (1/2) 2.0	BDL (2)	BDL (2)
	Liver	RMA (1)	BDL (1)	BDL (1)	BDL (1)	7.37 (1)	3.74 (1)	BDL (1)	BDL (1)
Bald Eagle	Egg	Barr Lake	BDL	0.099	BDL (1)	0.808 (1)	BDL (1)	6.93 (1)	BDL (1)
Golden Eagle	Liver	RMA**	BDL (1)	<0.050-0.216 (1/2) 0.120	BDL (2)	<0.031-0.221 (1/2) 0.118	BDL (2)	BDL (2)	BDL (2)
	Brain	RMA	BDL (2)	<.098-.257 (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)	BDL (2)
Ferruginous Hawk	Liver	RMA	BDL (5)	<0.050-0.293 (1/5)	BDL (5)	0.263-4.79 (5/5) 2.66	BDL (5)	BDL (5)	BDL (5)
	Brain	RMA	BDL (5)	<0.050-0.152 (1/5)	BDL (5)	<0.238-9.98 (4/5) 5.07	BDL (5)	BDL (5)	BDL (5)
Red-tailed Hawk	Liver	RMA	BDL (3)	<0.050-0.345 (1/3)	BDL (3)	0.520-6.59 (3/3) 4.10	BDL (3)	<0.313-0.759 (2/3) 0.482	BDL (3)
	Brain	RMA	BDL (3)	<0.050-0.093 (1/3)	BDL (3)	<0.751-9.44 (2/3) 6.34	BDL (3)	BDL (3)	BDL (3)
Great-horned Owl	Liver	RMA	BDL (4)	<0.050-0.086 (2/4) 0.047	BDL (4)	0.143-27.7 (4/4) 11.88	BDL (4)	<0.094-15.5 (3/4) 5.88	BDL (4)
	Brain	RMA	BDL (4)	BDL (4)	BDL (4)	<0.175-15.6 (3/4) 8.80	BDL (4)	<0.529-10.3 (3/4) 3.32	BDL (4)
Northern Harrier	Egg	RMA	BDL (2)	BDL (2)	BDL (2)	0.303-0.676 (2) 0.49	BDL (2)	BDL (2)	BDL (2)
Coyote	Liver	RMA (25)	BDL (1)	BDL (1)	BDL (1)	7.60 (1)	BDL (1)	BDL (1)	BDL (1)
Badger	Liver	RMA (25)	BDL (1)	BDL (1)	BDL (1)	1.64 (1)	BDL (1)	NRQ	NRQ
	Kidneys	RMA (25)	NRQ	NRQ	BDL (1)	0.801 (1)	BDL (1)	NRQ	NRQ

* Mean is calculated when 50 percent or more of samples have detectable contaminant levels. If less than 50 percent of samples have detectable contaminant levels, only the range of values are presented. When calculating the mean, values of ½ the detection limit are substituted for samples that are below the detection limit.
BDL Below Detection Limit.
n = Number of samples analyzed that contain detectable contaminant levels, nt = total number of samples,
NRQ Not Requested.
** For highly mobile species (mallard, pheasant, kestrel, mule deer) samples were widespread and RMA was evaluated as a whole entity.

APPENDIX C

OBSERVED TISSUE CONCENTRATIONS IN RAPTOR SAMPLES OF CHANCE AND BELIEVED CAUSE OF MORTALITY

Table 8-C1. Observed Tissue Concentrations in Raptor Samples of Chance and Believed Cause of Mortality (ESE, 1989)

Species	Age*	Physical Condition	Contaminant Levels of Brain/Liver Mercury	Dieldrin	DDE	Cause of Death
Ferruginous Hawk	A	Emaciated	BDL/BDL	0.678/0.527	BDL/BDL	Unknown
Ferruginous Hawk	A	Good	0.152/0.293	7.73/4.79	BDL/BDL	Unknown
Ferruginous Hawk	I	Emaciated/ Convulsions	BDL/BDL	9.98/3.45	BDL/BDL	Unknown
Ferruginous Hawk	A	Good	BDL/BDL	BDL/0.263	BDL/BDL	Electrocution***
Ferruginous Hawk	I	No body fat	BDL/BDL	6.85/4.26	BDL/BDL	Unknown***
Red-tailed Hawk	I	Emaciated	BDL/BDL	9.44/5.19	BDL/0.529	Unknown***
Red-tailed Hawk	A	Unknown	0.093/0.345	9.2/6.59	BDL/0.759	Unknown
Red-tailed Hawk	I	Unknown	BDL/BDL	BDL/0.52	BDL/BDL	Electrocution
Great-horned Owl	A	Unknown	BDL/0.086	15.6/10.8	10.3/15.5	Unknown
Great-horned Owl	A	Emaciated	BDL/BDL	9.32/27.7	0.475/2.47	Unknown
Great-horned Owl	A	Good	BDL/BDL	BDL/0.143	BDL/BDL	Unknown
Great-horned Owl	A	Unknown	BDL/ 0.051	10.2/8.89	2.24/5.49	Entero toxemia***
Golden Eagle		Unknown	0.257/0.216	BDL/0.221	BDL/BDL	Unknown***
Golden Eagle	I	Good	BDL/BDL	BDL/BDL	BDL/BDL	Respiratory Failure***

* A = Adult
I = Immature

** On wet weight basis

BDL = Below Detection Limit

*** Determined by Dr. Leroy Eggleston, DVM, or Dr. Terry Spraker, DVM.

SECTION NINE

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

SELENIUM EFFECTS AT KESTERSON RESERVOIR

AUTHORS AND REVIEWERS

AUTHORS

Margaret McVey
ICF International, Inc.
Fairfax, VA

Bruce A. Macler
Region 9
U.S. Environmental Protection Agency
San Francisco, CA

REVIEWERS

Wayne G. Landis (Lead Reviewer)
Institute of Environmental
Toxicology and Chemistry
Western Washington University
Bellingham, WA

James J. Anderson
School of Fisheries Research
University of Washington
Seattle, WA

Jay S. Jacobson
Boyce Thompson Institute
Cornell University
Ithaca, NY

Larry Kapustka
Ecological Planning and Toxicology, Inc.
Corvallis, OR

Thomas Sibley
School of Fisheries Research
University of Washington
Seattle, WA

Frieda B. Taub
School of Fisheries Research
University of Washington
Seattle, WA

Donald P. Weston
Horn Point Environmental
Laboratory
University of Maryland
Cambridge, MD

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LIST OF ACRONYMS

CVWQCB	Central Valley Water Quality Control Board
DFG	California Department of Fish and Game
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
FRP	Flexible Response Plan
FWS	U.S. Fish and Wildlife Service
KR	Kesterson Reservoir
LC_{50}	lethal concentration to 50 percent of organisms tested
NRDC	Natural Resources Defense Council
SLD	San Luis Drain
SWRCB	California State Water Resources Control Board
TDS	total dissolved solids
USBR	U.S. Bureau of Reclamation, Department of the Interior

ABSTRACT

Subsurface drainage of agricultural water in the San Joaquin Valley, California, to Kesterson Reservoir over several years resulted in a variety of adverse effects on waterfowl nesting in the reservoir area by 1983. Between 1983 and 1985, embryonic mortality and deformity rates at the site were significantly higher than those at control sites for several species of waterfowl.

Review of information related to past and present contamination at Kesterson Reservoir indicated that selenium was a major contaminant of concern because it had exceeded water quality guidelines and criteria, had accumulated in reservoir soils, had migrated into the ground water in some locations, and had been linked experimentally and observationally to wildlife effects. Measured tissue selenium levels at the site were elevated relative to controls for aquatic plants, invertebrates, mosquitofish, and several species of waterfowl and small mammals. In response to public pressure and the provisions of the Clean Water Act and the Federal Migratory Bird Treaty Act, the U.S. Bureau of Reclamation commissioned a study of possible remediations at the site. This ecological assessment was conducted to estimate the effectiveness of three cleanup alternatives in reducing the selenium levels in Kesterson Reservoir to levels protective of waterfowl and other wildlife.

To estimate the reduction in risk associated with the three cleanup alternatives, seven representative wildlife species in the area were selected for the risk assessment. For this assessment, food chain exposure was considered the most important pathway for exposure of fish and wildlife to selenium. The mallard, American coot, black-necked stilt, tricolored blackbird, mosquitofish, eared grebe, and San Joaquin kit fox were used to represent species that could be exposed to selenium via the midwater, benthic, and aquatic rooted plant, fish, and terrestrial pathways. Detailed food chain exposure diagrams for each of these species were developed into simplified selenium transfer models. These models were used with calculated transfer factors derived from experimental data and a Monte Carlo simulation technique to estimate the probability distribution of predictions of selenium concentration in the diets of the key species under each of the three remedial alternatives. The risk assessment indicated that each of the three remediation plans might present some risks to wildlife.

This case study represents a true risk assessment (i.e., for each remedial alternative, the probability that residual contamination of food chain organisms would be below harmful levels to key species at the refuge was calculated). Moreover, a weight-of-evidence approach was used to attribute the adverse effects of Kesterson Reservoir on waterfowl to one of the drainwater contaminants (i.e., selenium). The use of food web models was a good attempt to account for ecosystem processes in estimating exposures. The results of this risk assessment led to abandonment of the proposed remediations and adoption of site mitigation. The wetland was closed and converted to a terrestrial habitat. A larger wetland was created to provide for the lost wetland habitat at Kesterson.

9.1. RISK ASSESSMENT APPROACH

This case study represents a complete risk assessment in that it incorporates all the elements that constitute the risk assessment process (figure 9-1). The analysis included modeling of probabilities of levels of dietary selenium exposure to selected ecological components (species). Modeled exposures were based on environmental measurements at the site and incorporated data on transfer of material through food chain elements. Dietary exposures were compared with no observed adverse effect levels based on several measurement endpoints.

9.2. STATUTORY AND REGULATORY BACKGROUND

This risk assessment for Kesterson Reservoir (KR) was conducted by the U.S. Department of the Interior, Bureau of Reclamation (USBR), in response to a suit filed by the Natural Resources Defense Council (NRDC) against the California State Water Resources Control Board (SWRCB). The NRDC suit claimed violation of state water quality standards at KR and resulted in the SWRCB issuing Order WQ85-1 to the Central Valley Water Quality Control Board (CVWQCB) to define water quality standards and cleanup actions for KR (SWRCB, 1986). Under provisions of the Federal Migratory Bird Treaty Act, the USBR (Mid-Pacific Region) agreed to assist the CVWQCB to comply with the order and sponsored the assessment of remedial alternatives for KR (USBR, 1986a). This order also directed USBR to control conditions that caused any threat to wildlife or humans from operation of KR and resulted in its closure and mitigation for loss of the wetlands.

9.3. CASE STUDY DESCRIPTION

The overall objective of this ecological risk assessment was to provide, to the extent feasible with existing information, a quantitative analysis of the magnitude and uncertainty of estimates of potential adverse impacts on fish and wildlife populations that could result from implementation of KR cleanup alternatives. Three cleanup alternatives to reduce the selenium levels at KR were developed and qualitatively evaluated in the Kesterson Program Environmental Impact Statement (EIS) (USBR, 1986b). All alternatives involved closing the drains that fed KR with selenium, and all alternatives were designed to reduce food chain selenium exposure. From these, three scenarios were selected for analysis as providing high and low ends of the risk spectrum. The Flexible Response Plan (FRP) involved flooding ponds having the highest selenium concentrations with low-selenium water to dilute remaining selenium and disrupt food chains. Ponds with lower selenium content were to be dried out and disked to reduce vegetation. In Onsite Disposal Plan 1 (Plan 1), soils above 4 mg/kg selenium (342,000 m^3) would be excavated and disposed in a landfill. In Onsite Disposal Plan 2 (Plan 2), all contaminated soils (760,000 m^3) would be excavated and landfilled.

9.3.1. Problem Formulation

<u>Site Description</u>. Kesterson Reservoir is located on the Kesterson National Wildlife Refuge in Merced County in the San Joaquin Valley of California (figure 9-2). The San Joaquin Valley has historically been a region where intermittent ponding occurs from storm events and runoff. Agriculture along the western margin of this region requires irrigation, and the use of irrigation

Figure 9-1. Structure of Analysis for Selenium Effects at Kesterson Reservoir

PROBLEM FORMULATION

<u>Stressors</u>: selenium in sediments and surface waters.

<u>Ecological Components</u>: birds (five species), kit fox, and mosquitofish.

<u>Endpoints</u>: assessment endpoint was health and condition of local populations of selected fish and wildlife species; measurement endpoints included a variety of toxicological indices.

ANALYSIS

Characterization of Exposure

Selenium concentrations were measured in sediment, water, and tissues. Food chain transfer factors were developed. Food chain model was used to predict exposure.

Characterization of Ecological Effects

Effects of selenium were evaluated based on laboratory feeding studies and aquatic bioassays with fish. Field observations documented effects.

RISK CHARACTERIZATION

Monte Carlo simulations were used to estimate probability distributions of selenium in the diet under various remedial scenarios. Risk estimates were developed by combining dietary exposure and effects information. The habitat impacts associated with the physical effects of remediation were not considered.

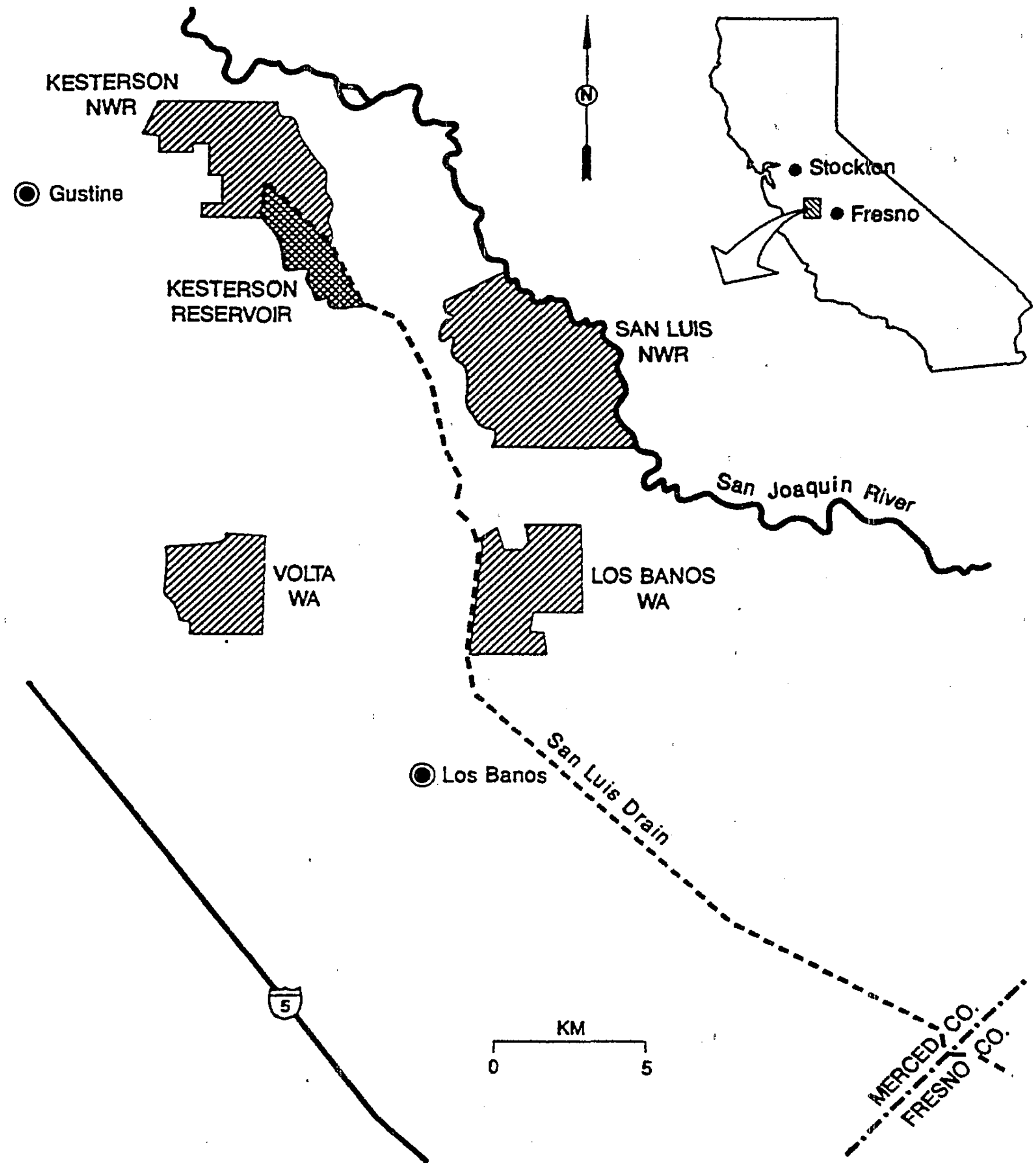

Figure 9-2. Site location of Kesterson Reservoir (Ohlendorf et al., 1989)

increased substantially during the 1960s and 1970s. Natural levels of salts in the soils of the region, coupled with water application and near-surface perched aquifers, led to accumulation of saline ground water in root zones of cropped species, which reduced productivity. It became apparent that drainage of these lands would be necessary to keep them in production. Subsurface drains were installed, flowing into the San Luis Drain (SLD), which was to discharge into the Sacramento/San Joaquin River Delta. This final link was never completed; rather, the drainwater was delivered via the SLD to KR. KR consisted of 12 shallow ponds (1 m to 1.5 m deep), totaling about 500 ha, designed to serve as evaporation and holding basins for this drainage water (Ohlendorf et al., 1988). Besides drainage to KR, more than 500 million m^3 of drainage water are discharged annually into other surface aquatic ecosystems in California, primarily the San Joaquin River and its western tributaries, the Delta-Mendota Canal, evaporation ponds in the Tulare Basin, and the Salton Sea and its principal tributaries (Skorupa and Ohlendorf, 1991).

Concurrent with agricultural and other development, more than 90 percent of the Central Valley's historic wetlands have been lost. Remnant wildlife populations have been concentrated onto the remaining wetlands, including those wetlands, such as KR, receiving drainage water. Frequently, drainage water evaporation ponds are the most common type of wetland available to wildlife during the spring. The shallow, nutrient-enriched waters of these ponds lead to high primary and secondary productivity and, therefore, are particularly attractive to breeding waterbirds. By this route, the ponds provide a pathway for wildlife exposure to contaminants in drainage water (Skorupa and Ohlendorf, 1991). KR is located on a major migratory bird flyway, and the ponds provide nesting and feeding grounds for several bird species.

<u>Stressors</u>. During 1983, it became clear that aquatic birds nesting at KR were experiencing poor reproductive success. A high frequency of both embryo mortality and developmental abnormalities occurred in most species during the 1983 to 1985 breeding seasons at KR (table 9-1) (Ohlendorf, personal communication in USBR, 1986a; Ohlendorf et al., 1986a, b). In contrast, researchers found no abnormalities in embryos from nests monitored through late stages of incubation or hatching at the Volta Wildlife Area, a control site located 10 km away in an area that did not receive agricultural drainage waters. The expected incidence of major external malformations in hatchlings of uncontaminated wild populations of birds and in embryos of laboratory-incubated mallard eggs is less than 1 percent (Pomeroy, 1962; Gilbertson et al., 1976; Hoffman, 1978; Hill and Hoffman, 1984). Table 9-2 provides estimates of avian population densities and past KR-related avian mortalities. These estimates are based on data from published literature, unpublished surveys by the U.S. Fish and Wildlife Service (FWS) and the California Department of Fish and Game (DFG), and consultation with personnel from these agencies and other local experts (USBR, 1986a).

Delivery of agricultural drainwater to KR ceased in June 1986. Contamination remaining at KR was that portion of delivered contaminants that had accumulated in soils and biota. The SWRCB evaluated several contaminants in SLD drainwater and KR surface water, ground water, soils, and biota to determine if there was any evidence for residual contamination that could result in future harmful effects to wildlife. Seven of ten drainwater constituents warranted further analysis because they exceeded water quality guidelines and criteria in historic SLD drainwater and KR surface water: boron, chromium, mercury, molybdenum, selenium, total dissolved solids (TDS), and zinc (table 9-3).

Table 9-1. Frequency of Dead or Deformed Embryos or Chicks in Nests of Aquatic Birds at Kesterson Reservoir, 1983-1985 (Ohlendorf et al., 1989)

		Frequency of Occurrence[a]					
Species		Dead		Deformed		Total	
Year	Nests[b]	No.	%	No.	%	No.	%
Coot[c]							
1983	59/92	35	(59.3)	25	(42.4)	38	(64.4)
Grebe[d]							
1983	141/163	84	(59.6)	22	(15.6)	89	(63.1)
Ducks[e]							
1983	30/42	5	(16.7)	3	(10.0)	7	(23.3)
1984	13/36	6	(46.2)	0	(0)	6	(46.2)
1985	17/27	6	(35.3)	1	(5.9)	6	(35.3)
Stilt							
1983	101/125	17	(16.8)	17	(16.8)	24	(23.8)
1984	63/189	7	(11.1)	12	(19.0)	14	(22.2)
1985	69/96	20	(29.0)	23	(33.3)	30	(43.5)
Avocet							
1983	16/16	0	(0)	0	(0)	0	(0)
1984	19/51	0	(0)	0	(0)	0	(0)
1985	22/35	4	(18.2)	4	(18.2)	5	(22.7)
Killdeer[f]							
1984	12/32	0	(0)	0	(0)	0	(0)
1985	16/25	7	(43.8)	3	(18.8)	8	(50.0)
Total	578/929	191	(33.0)	110	(19.0)	227	(39.3)

[a]Dead = number of nests (and percentage) with one or more dead embryos; deformed = nests with one or more deformed embryos or chicks; total = sum of all nests with at least one dead or deformed embryo or chick. All percentages calculated by dividing by number of monitored nests.
[b]Monitored/found: nests monitored to hatching or from which a late-stage embryo was collected/nests found during study, including those lost to predation, flooding, and desertion.
[c]No coot nests found in 1984 or 1985, although adults were present throughout the nesting season.
[d]Adult birds present throughout the nesting season only during 1983.
[e]Mallard, gadwall, cinnamon teal, and northern pintail.
[f]Species not studied in 1983.

Table 9-2. Kesterson Bird Populations and Mortalities (see also Ohlendorf et al., 1989)

	Mallard	American Coot	Black-Necked Stilt	Tricolored Blackbird	Eared Grebe	San Joaquin Kit Fox
Kesterson Reservoir Mortality	1 found dead in 1986[a] 17-22 ducks "lost"[b]	438 "lost" in 1983[b]	197 "lost" in 1985[b]	82,150 "lost" eggs and chicks in 1986[c]	411 "lost" in 1983[b]	No data
Kesterson Reservoir Population	45-100 per day[d]	9,489[e]	50-60 individuals nesting in 1986[a]	47,000[d]	17[f]	15-20[g]
San Joaquin Valley Population	89,142[h]	216,623[h]	No data	85,850[i]	No data	5,294[j]
California Population	435,421[h]	427,415[h]	~100,000[k]	133,000[i]	730,250[l]	10,000-14,800[m]
Pacific Flyway Population	1,759,800[n]	562,400[n]	No data	No data[i]	No data	Same as statewide population

[a] Personal communication from Mary Coakley, wildlife hazer, 10-2-86. This value does not reflect nestling mortalities.

[b] Unpublished memo from Dr. Harry Ohlendorf to Mr. Ken Anderson of the General Accounting Office, 5-16-86. Includes dead or deformed embryos or chicks and those presumed to have hatched but that failed to survive. Some losses were due to predation, but these cannot be accurately separated from possible mortality due to selenium toxicosis.

[c] Rough calculation assuming about 23,500 nesting pairs, a clutch size of 3.5, and mortalities of all but 100 fledglings.

[d] Unpublished FWS hazing data from Kesterson Reservoir, March 1986.

[e] Maximum average daily use total from unpublished FWS data 1984-86. See draft EIS.

[f] Paveglio (personal communication b). Average daily use for 1982, 1984, and 1985. Peak use is 125-175 during migration.

[g] Paveglio (personal communication b). Total adult population in adjacent 25,000-acre range. Up to five individuals (including pups) have been seen simultaneously at Kesterson Reservoir.

[h] Unpublished FWS data, 1973-77 average Pacific Flyway mid-winter waterfowl survey, 1986.

[i] Population estimates by Rich DeHaven (FWS) for the period 1969-72. Note that this species is largely endemic to California, so the statewide population approximately equals global population.

[j] Based on adult population estimates by Morrell (1975) for Kern, Tulare, Kings, Fresno, San Benito, Merced, Stanislaus, and San Joaquin Counties.

[k] "Ballpark" estimate by Ron Jurek, DFG, 10-2-86.

[l] Maximum extrapolated population estimate from Mono Lake, August 30, 1976 (Winkler et al., 1977); actual statewide population is probably much higher than this (Gould, personal communication).

[m] Range of adult population estimates for California by Morrell (1975).

[n] Unpublished FWS data, 1955-85 average from Pacific Flyway midwinter waterfowl survey.

Table 9-3. Summary of Contaminant Levels in KR Media (USBR, 1986a)

Constituents	Drainwater, Surface Water, or Shallow Ground Water > Standards?	KR Soils > Background?	KR Water Supply Ground Water > Standards?	KR Biota > Background?
Boron	Yes	No	Yes	Yes
Cadmium	No	--	--	--
Chromium	Yes	No	No	No
Copper	No	--	--	--
Manganese	No	--	--	--
Mercury	Yes	No	No	No
Molybdenum	Yes	No	No	No
Nickel	No	--	--	--
Selenium	Yes	Yes	No	Yes
Zinc	Yes	No	No	No
TDS	Yes	NA[a]	Yes	NA[a]

[a]NA = not applicable; TDS not applicable to water.

In 1983, researchers concluded that selenium was the most likely cause of avian deaths and deformities at KR because the types of deformities found in avian embryos and young were typical of those induced by exposure to high selenium levels, but not high levels of the other contaminants of concern. Boron, molybdenum, and strontium all were characterized as below toxic levels, but few data supported these claims. Although boron had been shown to cause mortality and teratogenic development of eggs, the dietary concentrations used in these studies were much higher than the levels in most dietary elements at KR (Hothem and Ohlendorf, 1989). Studies by Heinz et al. (1987) indicated that when mallards were fed diets containing selenomethionine, some embryos had deformities similar to those observed at KR. Studies with poultry and quail had shown several toxic responses to dietary ingestion of selenium compounds, including reduced growth; reproductive impairment; embryonic, hatchling, and adult mortality; and gross deformities (Rosenfeld and Beath, 1964; National Research Council, 1976, 1980; Shamberger, 1981, 1983). Field studies had correlated high levels of dietary selenium with low hatchability, embryonic deformity, and high mortality in wild birds at KR (Ohlendorf et al., 1986a, b; Saiki, 1986).

Selenium's potential for bioaccumulation caused additional concern. Studies by Lemly (1985) in an aquatic ecosystem indicated that plankton could concentrate selenium to 750 times the concentration in water and that fish could concentrate selenium to levels 4,000 times that in water.

Ecological Components. The wildlife species present at KR represented a variety of trophic levels and, hence, selenium exposure potential. Selection of key fish and wildlife species to represent potential exposure pathways in the risk assessment was based on several considerations. A species was selected if it was a terminus species of a major KR food chain exposure pathway, if impacts of KR on the species had been observed in the past, if it was a rare or endangered species, if it was a species with particularly sensitive life stages, or if information was available on the effects of selenium exposure for the species. Not all of the species selected satisfied all of these criteria. A weakness of this assessment was that it ignored primary producers and organisms at the lowest levels of the food chains. Descriptions of the key species and rationales for their selection are provided below.

- Mallard. Adult *Anas platyrhynchos* are omnivorous; however, during nesting and egg laying, the diet of adult females changes from primarily vegetation to primarily aquatic invertebrates.

- American coot. The adult coot (*Fulica americana*) feeds primarily on terrestrial and aquatic plants, insects, and other epiphytal fauna.

- Black-necked stilt. Adult *Himantopus mexicanus* feed while wading in the water primarily on littoral benthic epifauna.

- Tricolored blackbird. Adult *Agelaius tricolor* feed their young almost exclusively on adult insects and aquatic insect larvae. The status of the tricolored blackbird as a candidate for federal threatened and endangered species listing also was a factor in its selection.

- Eared grebe. The eared grebe (*Podiceps nigricollis*) is an invertebrate- and fish-eating bird whose population at KR had been shown previously to be adversely affected.

- Mosquitofish. The mosquitofish (*Gambusia affinis*) is the only fish that still existed in KR waters in 1986; it is highly resistent to selenium toxicosis.

- San Joaquin kit fox. The kit fox was included as the terrestrial food chain receptor because it is a federal- and state-listed endangered species.

For each of these species, a review of the scientific literature was conducted to quantify their food habits, and dietary preferences were summarized by life stage, sex, and season, as appropriate. The home range size of each species was estimated from literature values.

Endpoints. The assessment endpoint was the health and condition of local populations of selected fish and wildlife species under alternative remedial scenarios. Measurement endpoints included field, laboratory, and literature investigations of adverse effects on birds (reduced reproductive success, growth, and survival), kit fox (liver changes and heart, kidney, and spleen effects), and mosquitofish (survival).

Comments on Problem Formulation

Strengths of the case study include:

- *Detailed description of the site and profiles of metal contamination are available.*

- *The review of criteria for the inclusion of the selected ecological components (species) is sufficiently detailed. In addition, substantial data were collected from the field.*

Limitations include:

- *Boron is not included as a stressor because of the lack of toxicity data. Some of the resources of this study could have been allocated toward providing that information.*

- *Habitat alterations are not included as a stressor, although these would have occurred from the proposed remediations. These remediations would largely destroy the existing habitat and severely affect existing food chains at all levels. The lack of consideration of habitat alteration was a result of the Federal Migratory Bird Treaty Act, which focuses on chemical impacts, such as those from selenium.*

Comments on Problem Formulation (continued)

- *Ecological components include key species of social interest. However, the ecosystem as a whole and species important to ecosystem stability are not addressed. In particular, lower food chain elements are ignored. Many plants are sensitive to elevated boron, and damage to these organisms at low trophic levels could lead to large changes in the ecosystem. By focusing on the removal of selenium from incorporation into the food chains at KR, consideration of overall survival of the key species is ignored.*

General comment:

- *This study is based on a $2 to $3 million research effort to characterize the impacts of metal contamination on the waterfowl and endangered species of KR. The studies were initiated due to the discovery of malformation and other teratogenic effects in waterfowl chicks. These teratogenic impacts caused a large public outcry and the implementation of the Federal Migratory Bird Treaty Act. The emphasis on avian species within KR was driven by these factors.*

9.3.2. Analysis: Characterization of Ecological Effects

Birds. Selenium toxicity has been reviewed extensively and documented in poultry and quail in studies dating back to the 1930s (Rosenfeld and Beath, 1964; National Research Council, 1976, 1980; Shamberger, 1981, 1983). The early studies of chickens receiving selenium in their diet from cereal grains grown in seleniferous soils showed effects ranging from both reduced growth and reproductive impairment to complete failure of hatching (Moxon, 1937; Poley et al., 1937; Poley and Moxon, 1938; Moxon and Rhian, 1943). The selenium content of the grain was speculated to be as low as 10 ppm (Heinz et al., 1987). Ort and Latshaw (1978) identified 5 ppm selenium in feed (added as sodium selenite) as the threshold for reduced hatching success in chickens. Edema of the head and neck was seen at 7 to 9 ppm. Heinz et al. (1987) induced reproductive impairment and embryonic deformities in mallards at dietary levels as low as 10 ppm selenium as selenomethionine. Based on the existing information for birds, the range of harmful dietary selenium threshold concentrations was assumed to be 5 to 10 ppm. Field studies had correlated high levels of dietary selenium with low hatchability, embryonic deformity, and high mortality in wild birds at KR (Ohlendorf et al., 1986a, b, 1988; Saiki, 1986), but extreme conditions at KR provided little opportunity to assess thresholds for selenium toxicity. A summary of ecological effects reported for avian species is shown in table 9-4.

Mammals. Selenium concentrations of 8 to 30 ppm (dry weight) in the diet have been associated with chronic toxicity in mammals (Wilber, 1980). Indications of toxicity such as liver changes and heart, kidney, and spleen effects have been observed in mammals following chronic exposures to feed containing 1.4 to 3.0 ppm selenium (Anspaugh and Robinson as cited in USBR, 1986a). On the other hand, Halverson et al. (1966) observed no significant effect on growth in

Table 9-4. Summary of Dose Response Reported for Avian Species

Dose (ppm)	Chemical Form	Response	Test Organism	Reference
100	Sodium selenite	Mortality of adults; weight loss	Mallard	Heinz et al., 1987
78	Selenium selenite	Lowered egg production	Chicken	Arnold et al., 1973
40	Sodium selenite	Reduced chick survival	Chicken	Arnold et al., 1973
25	Sodium selenite	Mortality of adults	Mallard	Heinz et al., 1987
		Depressed body weight - adult	Mallard	Heinz et al., 1987
		Reduced egg laying	Mallard	Heinz et al., 1987
		Reduced duckling survival	Mallard	Heinz et al., 1987
		Lower Radcliffe index	Mallard	Heinz et al., 1987
		Depressed body weight - duckling	Mallard	Heinz et al., 1987
10	Selenomethionine	Reduced duckling survival	Mallard	Heinz et al., 1987
		Low hatching success	Mallard	Heinz et al., 1987
		18% abnormal embryos	Mallard	Heinz et al., 1987
		Multiple malformations	Mallard	Heinz et al., 1987
		Depressed body weight	Mallard	Heinz et al., 1987
12	Sodium selenite	Lower hatchability	Japanese quail	El-Begearmi et al., 1977
10	Selenomethionine	Low hatching success	Chicken	Poley and Moxon, 1938
10	Sodium selenite	Multiple malformations	Mallard	Heinz et al., 1987
7	Selenium selenite	Lowered hatching success	Chicken	Ort and Latshaw, 1978
7	Selenium selenite	Reduced egg weight	Chicken	Ort and Latshaw, 1978
6	Selenomethionine	No effect on adult or egg production	Chicken	Moksnes, 1983
5	Sodium selenite	Reduced growth	Chicken	Jensen, 1975
5	Sodium selenite	Impaired hatching success	Chicken	Ort and Latshaw, 1978
8	Sodium selenite	Lowered chick survival	Japanese quail	El-Begearmi et al., 1977

rats exposed to 1.6 to 4.8 ppm selenium in their feed. A diet of 6.4 ppm selenium in feed in the form of sodium selenite or seleniferous wheat caused significant growth depression, and death occurred in the postweanling rats after 4 weeks at levels of 8.0 to 11.2 ppm in the diet. Earlier studies had shown a toxic response in rats maintained on a diet containing 5 ppm selenium (Moxon, 1937; Franke and Painter, 1938). Based on these and other studies, the threshold range of harmful dietary selenium concentrations was estimated to be 2 to 5 ppm.

Fish. In the development of ambient water quality criteria for selenium, the U.S. Environmental Protection Agency (1980) summarized a data base of 23 studies of 8 freshwater species. The acute toxicity (96-hour LC_{50}) values ranged from 0.62 to 28.5 mg selenium/L for the bluegill (*Lepomis macrochirus*); 96-hour LC_{50} concentrations of 2.1 and 5.2 mg selenium/L were determined for fathead minnow (*Pimephales promeles*) fry and juveniles, respectively.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *Several species and toxicity evaluations are used in estimating the toxic range of selenium.*

Limitations include:

- *Effects of selenium are extrapolated from laboratory studies and are based on various forms of selenium; these studies may not reflect the conditions at KR. There also may be differences in bioavailability of selenium between the laboratory studies and the actual field conditions.*

- *No consideration is given to the physical effects of the remedial actions on fish and waterfowl habitat. (It had been determined to close KR and open an adjacent wetland habitat.) The removal or capping of the water and surrounding habitat contaminated by selenium would cause a major alteration of the habitat in which several endangered species are able to reproduce. In addition, it is important to consider that the material accumulates as succession occurs and recolonization by plants redistributes any buried selenium back into the ecosystem.*

9.3.3. Analysis: Characterization of Exposure

As described in the Kesterson Program Final EIS (USBR, 1986b), the potential existed for residual soil/sediment selenium contamination to move into terrestrial and aquatic food chains. Selenium in the environment may occur in numerous chemical forms due to the processes of oxidation and reduction and biologically mediated transformations. Selenate is the most mobile form of selenium and makes up the majority of selenium that had been delivered to KR via the

SLD. The various forms of selenium were not distinguished in this risk assessment, however, because the toxicity evaluation included all forms.

For this assessment, food chain exposure was considered the most important pathway for exposure of fish and wildlife to selenium. Detailed food chain exposure diagrams for each of the key species previously noted were developed into simplified selenium transfer models. These models were used with calculated transfer factors derived from experimental data and a Monte Carlo simulation technique to estimate the probability distribution of predictions of selenium concentration in the diets of the key species under each of the three remedial alternatives.

Measured Selenium Concentrations. Mosquitofish captured at KR in May 1982 contained high levels of selenium—about 135 ppm (Saiki, 1986). Because of this, FWS began intensive studies at KR and at Volta Wildlife Area (a control area 10 km to the southwest of the site [figure 9-2] that is not contaminated by agricultural drainwater) to further define the effects and extent of contamination resulting from drainwater delivery to KR. Of the six contaminants of concern that SWRCB identified, FWS found that only selenium exhibited elevated concentrations in KR soils compared with Volta (table 9-3). Boron, chromium, mercury, molybdenum, selenium, TDS, and zinc were elevated in Kesterson waters. Examination of the biota at KR showed that only selenium and boron were significantly higher in tissues versus biota from Volta (Ohlendorf et al., 1986b; table 9-3); therefore, only boron and selenium were considered of potential concern for KR. (Because of limited toxicological data, a boron risk assessment was not completed.) Food chain organisms and fish were sampled in detail at KR and found to have significantly higher selenium concentrations than at Volta (table 9-5).

Selenium concentrations in eggs of aquatic birds at KR were far higher than those reported elsewhere in the United States (U.S. Department of the Interior, 1984; Ohlendorf et al., 1986a), and selenium concentrations in the livers of aquatic birds sampled at KR significantly exceeded selenium concentrations in livers of birds from Volta (Ohlendorf et al., 1986b). A study of potential selenium contamination of mammals at KR was conducted in 1984. Preliminary data for the four most abundant species in this sample (California vole, harvest mouse, house mouse, and ornate shrew) indicated that selenium levels at KR were 10 to 1,000 times higher than those at Volta (USBR, 1986a).

Estimating Selenium Concentrations for the Remedial Alternatives. To estimate the reduction in selenium risk associated with the three cleanup alternatives, changes in exposure were modeled based on selenium movement through food chains to the key species. The terrestrial food chain represented the dry areas around KR. It included selenium flux from drainwater deposition through uptake into land plants, herbivores, and carnivores, ending with the kit fox. The aquatic food chains represented the ponds and seasonally wet areas of KR and included benthic, aquatic (water column), and rooted plant pathways. For the nonpiscivorous aquatic birds, selenium flux was from sediments through the water column into plants and herbivorous insects. It ended with mallards (aquatic plants and invertebrates), American coots (plants and epiphytal fauna), tricolored blackbirds (aquatic insects), and black-necked stilts (benthic epifauna). A fish/piscivorous bird pathway considered selenium flux from sediments through the water column herbivores into mosquitofish and the fish-eating eared grebe.

Table 9-5. Selenium Concentration (ppm, dry weight) in Composite Samples of Plants, Invertebrates, and Mosquitofish, May 1983 (Ohlendorf et al., 1986b)

	Volta			Kesterson		
Sample	N[a]	Mean[b]	(Range)	N	Mean	(Range)
Filamentous algae	0/4	ND[c]		6/6	35.2	(12–68)
Rooted plants	1/1	0.43		18/18	52.1	(18–79)
Net plankton	4/4	2.03	(1.4–2.9)	7/7	85.4	(58–124)
Water boatmen (*Corixidae*)	5/5	1.91	(1.1–2.5)	2/2	22.1	(20–24)
Midge larvae (*Chironomidae*)	3/3	2.09	(1.5–3.0)	3/3	139	(71–200)
Dragonfly nymphs (*Anisoptera*)	2/2	1.29	(1.2–1.4)	6/6	122	(66–179)
Damselfly nymphs (*Zygoptera*)	2/2	1.45	(1.2–1.7)	3/3	175	(118–218)
Mosquitofish (*Gambusia affinis*)	5/5	1.29	(1.2–1.4)	12/12	170	(115–283)

[a] Number with measurable concentrations/number analyzed.
[b] Geometric means computed only when selenium was measurable in at least 50 percent of samples. When only one sample was analyzed, the concentration is shown in this column.
[c] ND = not detected.

Detailed food chain exposure diagrams for each of the seven key species were abstracted into simplified selenium transfer models, as illustrated in figure 9-3 and table 9-6. The simplified pathways contain all of the basic selenium transfer pathways present in the complex transfer diagrams. The transfer of selenium through the food chain and concentrations of selenium in food groups were estimated for each cleanup alternative using transfer factors derived from the studies at Kesterson and elsewhere (tables 9-6, 9-7, and 9-8) and diet factors based on the dietary habits of the receptor species. The empirical transfer and diet factors served as the basis for the mathematical model used to predict the relationship between selenium in each trophic level and the exposure of key species to selenium.

Transfer factors were based on empirical observations (table 9-9). For example, the transfer factor between water and nonrooted plants at KR was calculated using paired observations of selenium concentrations in water and nonrooted plants. This transfer factor reflects the relationship between water and plant selenium concentrations at a water selenium concentration that was higher than is expected to result from any of the cleanup alternatives. Because the uptake and metabolism of selenium probably does not have a linear relationship with concentrations in water, a transfer factor appropriate for the predicted range (2 to 15 μg selenium/L) was derived from literature reviewed by Lemly (1985) and Ohlendorf et al. (1986b). The same procedure was followed for derivation of other transfer factors in the aquatic food chain. Transfer factors derived from the literature were generally about two to three times higher than those observed in high water selenium concentrations at KR (USBR, 1986a).

Diet factors were used to model the fraction of the whole diet of each organism that was contributed by each compartment contained in the simplified selenium transfer diagram. Diet factors were based on species food preferences as described in the literature, their home range size relative to the size of KR (see Ecological Components in section 9.3.1), and the relative abundance of different types of prey species at KR. For species that consumed aquatic invertebrates, for example, the relative abundance of herbivorous and carnivorous aquatic invertebrates measured at KR was used to specify the composition of these organisms in the diet.

Because water applied to KR in each of the three remedial alternatives would have very low selenium concentrations (i.e., <1 μg/L), the major potential source of selenium for biological uptake was the sediments. The amount of selenium present in sediments in 1986 was quite variable throughout KR, but it tended to be greater in southern ponds than in northern ponds. In the FRP alternative, where no removal of sediment was involved, a value of 7 ± 7 mg selenium/kg (standard deviation), representative of the southern ponds, was used (USBR, 1986a). The variance component of this estimate reflects the spatial heterogeneity of selenium measured at KR. Plan 1 was estimated to leave an average sediment selenium concentration of 3 ± 2 mg/kg. Plan 2 would leave a sediment selenium concentration of 1.5 ± 1 mg/kg.

A Monte Carlo model simulated the selenium concentration in each compartment along the exposure pathways from sediment to target organism by multiplying the selenium concentration in the previous compartment by the appropriate transfer factor. Key species' average dietary selenium concentration was calculated by weighting each component of the diet by the appropriate diet factor. Initial conditions for the model were those selenium concentrations estimated in KR sediments or surface water after implementation of the cleanup alternatives.

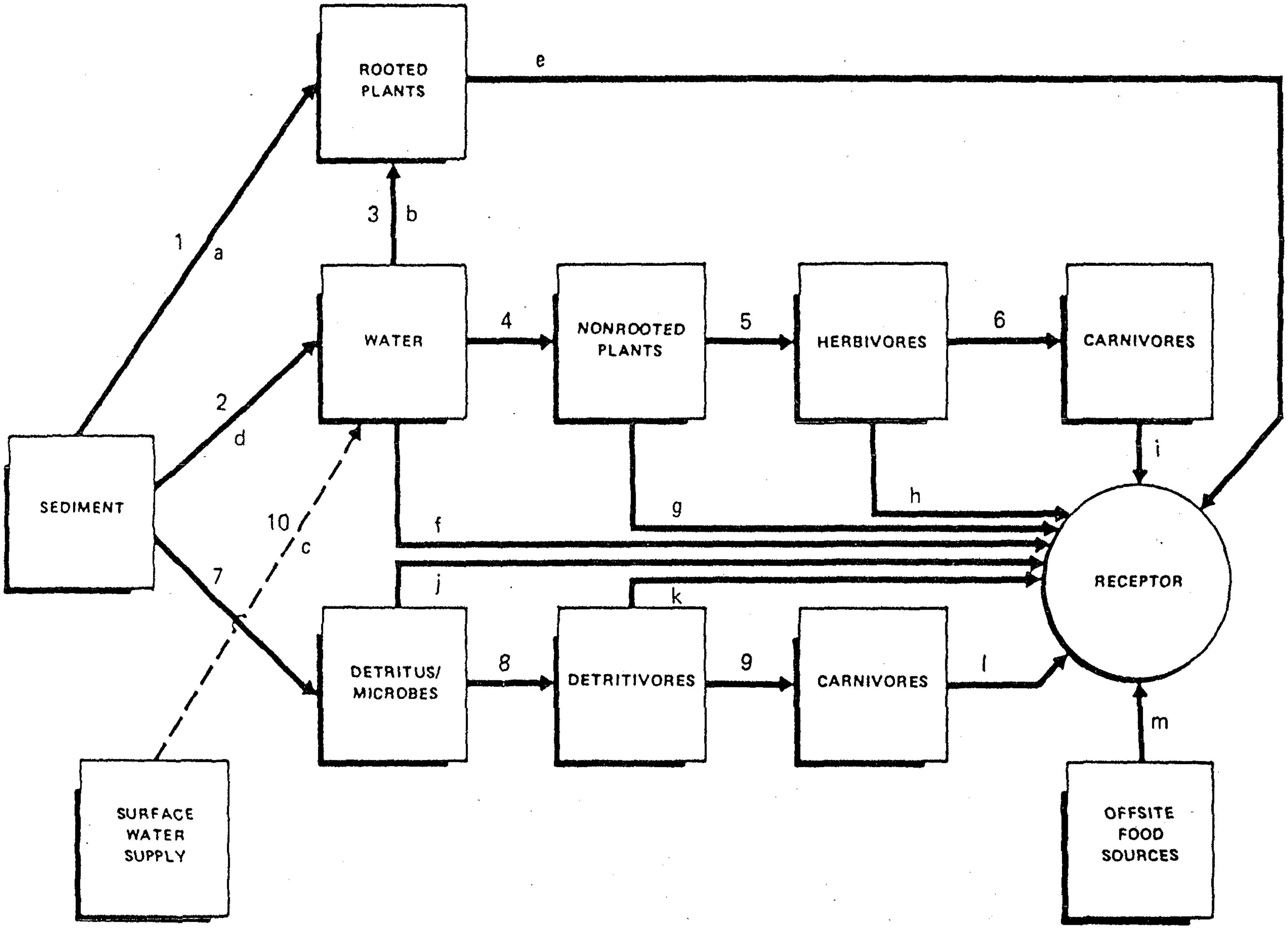

Figure 9.3. Simplified selenium transfer diagram with key to transfer and diet factors (see table 9-6) (USBR, 1986a)

Table 9-6. Transfer and Diet Factors for Simplified Selenium Transfer Diagram for Mallard, American Coot, Tricolored Blackbird, and Black-Necked Stilt (see figure 9-3) (USBR, 1986a)[a]

	Past Condition	Flexible Response	Onsite Disposal 1[b]	Onsite Disposal 2
Sediment Conc. (mg/kg d.w.)	7 (7)	7 (7)[c]	3 (2)	1.5 (1)
Surface Water Supply (mg/L)	0.3	0	0	0
TRANSFER FACTORS				
1 Sediment - Rooted Plants	2.8 (3)	2.8 (3)	2.8 (3)	2.8 (3)
2 Sediment - Water[d]	0.0003-0.002	0.0003-0.002	0.0003-0.002	0.0003-0.002
3 Water - Rooted Plants	81 (21)	81 (21)	81 (21)	81 (21)
4 Water - Nonrooted Plants	187 (22)	500 (50)	500 (50)	500 (50)
5 Nonrooted Plants - Herbivores	2.2 (0.8)	4.0 (2.0)	4.0 (2.0)	4.0 (2.0)
6 Herbivores - Carnivores	0.7 (0.3)	1.5 (0.5)	1.5 (0.5)	1.5 (0.5)
7 Sediment - Detritus/ Microbes	2.4 (2.9)	2.4 (2.9)	2.4 (2.9)	2.4 (2.9)
8 Detritus/Microbes - Detritivores	2.2 (0.8)	2.2 (0.8)	2.2 (0.8)	2.2 (0.8)
9 Detritivores - Carnivores	0.7 (0.3)	0.7 (0.3)	0.7 (0.3)	0.7 (0.3)
RELATIVE SUPPLY FACTORS[e]				
a Sediment - Rooted Plants	25	25	25	25
b Water - Rooted Plants	75	75	75	75
c Surface Water Supply - Water	100	0	0	0
d Sediment - Water	0	100	100	100

DIET FACTORS[f] FOR PAST CONDITION, FLEXIBLE RESPONSE, AND ONSITE DISPOSAL

	Adult Female Mallard Nesting	Adult American Coot	Tricolored Blackbird Nestling	Adult Black-Necked Stilt
e Rooted Plants - Receptor	14 (3)	30 (5)	3 (2)	0
f Water - Receptor	5 (2)	5 (2)	0	5 (2)
g Nonrooted Plants - Receptor	33 (7)	18 (3)	2 (1)	0
h Herbivores - Receptor	41 (6)	35 (5)	79 (8)	38 (5)
i Carnivores (1) - Receptor	7 (2)	7 (2)	16 (5)	7 (3)
j Detritus/Microbes - Receptor	0	1 (1)	0	5 (1)
k Detritivores - Receptor	0	2 (1)	0	38 (5)
l Carnivores (2) - Receptor	0	2 (1)	0	7 (3)
m Offsite Food Sources	0	0	0	0

[a] Standard deviations are in parentheses.
[b] For the seasonally wet areas. Also applicable to FRP seasonally wet areas in the northern ponds.
[c] For ponds that will be wet all year (southern ponds).
[d] Uniform distribution; therefore, range is given.
[e] In cases where two routes of selenium supply exist, their ratio of supply is defined.
[f] Percent of total diet from each compartment.

Table 9-7. Transfer and Diet Factors for Simplified Selenium Transfer Diagram for Eared Grebe and Mosquitofish (USBR, 1986a)[a]

	Past Condition	Flexible Response	Onsite Disposal 1[b]	Onsite Disposal 2
Sediment Conc. (mg/kg d.w.)	7 (7)	7 (7)[c]	3 (2)	1.5 (1)
Surface Water Supply (mg/L)	0.3	0	0	0
TRANSFER FACTORS				
1 Sediment - Water[d]	0.0003-0.002	0.0003-0.002	0.0003-0.002	0.0003-0.002
2 Water - Nonrooted Plants	187 (22)	500 (50)	500 (50)	500 (50)
3 Nonrooted Plants - Herbivores	2.2 (0.8)	4.0 (2.0)	4.0 (2.0)	4.0 (2.0)
4 Herbivores - Carnivores	0.7 (0.3)	1.5 (0.5)	1.5 (0.5)	1.5 (0.5)
RELATIVE SUPPLY FACTORS[e]				
a Surface Water Supply - Water	100	0	0	0
b Sediment - Water	0	0	0	0

DIET FACTORS[f] FOR PAST CONDITIONS, FLEXIBLE RESPONSE, AND ONSITE DISPOSAL

	Eared Grebe	Mosquitofish
c Water - Receptor	10 (5)	NA[g]
d Carnivores (1) - Receptor	90 (5)	NA[g]

[a] Standard deviations are in parentheses.
[b] For the seasonally wet areas. Also applicable to FRP seasonally wet areas in the northern ponds.
[c] For ponds that will be wet all year (southern ponds).
[d] Uniform distribution; therefore, range is given.
[e] In cases where two routes of selenium supply exist, their ratio is defined.
[f] Percent of total diet from each compartment.
[g] NA = not applicable.

Table 9-8. Transfer and Diet Factors for Simplified Selenium Transfer Diagram for San Joaquin Kit Fox (USBR, 1986a)[a]

	Past Condition	Flexible Response	Onsite Disposal 1[b]	Onsite Disposal 2
Sediment Conc. (mg/kg d.w.)	3 (2)	3 (2)[c]	3 (2)	1.5 (1)
Surface Water Supply (mg/L)	0.3	0	0	0
TRANSFER FACTORS				
1 Soil - Terrestrial Plants	10 (5)	10 (5)	10 (5)	10 (5)
2 Terrestrial Plants - Herbivores	0.3 (0.3)	0.3 (0.3)	0.3 (0.3)	0.3 (0.3)
3 Herbivores - Carnivores	4 (2)	4 (2)	4 (2)	4 (2)
DIET FACTORS[d]				
a Herbivores - Receptor	22.5 (20)	9 (5)	9 (5)	9 (5)
b Carnivores - Receptor	2.5 (2)	1 (1)	1 (1)	1 (1)
c Offsite Food Sources	75 (25)	90 (10)	90 (10)	90 (10)

[a] Standard deviations are in parentheses.
[b] For the seasonally wet areas. Also applicable to FRP seasonally wet areas in the northern ponds.
[c] For ponds that will be wet all year (southern ponds).
[d] Percent of total diet from each compartment.

Table 9-9. Summary of Data Used to Derive Transfer Factors[a]

	Nonpiscivorous Bird Pathway					
	Past		Flexible Response		Onsite Disposal	
	Value	Data Source	Value	Data Source	Value	Data Source
Sediment	7±7	1	7±7	1	3±2, 1.5±1[b]	1[c]
Rooted Plants	26±18	1	26±18	1	26±18	1
Water	300	1	2-15	2,3	2-15	2,3
Nonrooted Plants	56±5.5	2,3	0.3-20[e]	5	0.3-20[e]	5
Nonbenthic Herbivores	127±23	2,3,4	d	5	d	5
Nonbenthic Carnivores	92±19	2,3,4	2-270[e]	5	2-270[e]	5
Detritus	26±18	6	26±18	6	26±18	6
Detritivores	56±5.5	2,3,4	56±5.5	2,3,4	56±5.5	2,3,4
Benthic Carnivores	127±23	2,3,4	127±23	2,3,4	127±23	2,3,4
	Fish/Piscivorous Bird Pathway					
Sediment	7±7	1	7±7	1	3±2, 1.5±1[b]	1[c]
Water	300	1	2-15	2,3	2-15	2,3
Nonrooted Plants	56±5.5	2,3	0.3-20[e]	5	0.3-20[e]	5
Nonbenthic Herbivores	127±23	2,3,4	d	5	d	5
Mosquitofish/Carnivores	104±25	2,3	2-270[e]	5	2-270[e]	5
	Terrestrial Pathway					
Soil	3±2	1	3±2	1	3±2, 1.5±1[b]	1[c]
Terrestrial Plants	30±10	1	30±10	1	30±10	1
Herbivores	10±24	7	10±24	7	10±24	7
Carnivores	48±17	7	48±17	7	48±17	7

Data Sources:

1 USBR, 1986a. Standard QA/QC procedures, all data sources given in EIS.
2 Presser and Barnes, 1984.
3 Saiki and Lowe, 1987.
4 Ohlendorf et al., 1986a. QA/QC procedures specified.
5 Lemly, 1985. QA/QC procedures not specified.
6 No specific reference. Assumed majority of detritus composed of rooted macrophytes.
7 Clark, personal communication.

Note: All standard deviations estimated by dividing range by 6 except those from Presser and Barnes, 1984, and Saiki and Lowe, 1987, which were given in the references. This is based on the assumptions of a normal distribution and that >99 percent of values are within ±3 standard deviation.

[a] All units mg/kg (d.w.) except water, which is μg/L.
[b] Two values given for Onsite Disposal No. 1 and 2, respectively.
[c] Mean value from USBR, 1986a.
[d] Inferred from reference data.
[e] Range only was given in reference.

Each estimated transfer factor and diet factor had an associated standard distribution, either uniform or lognormal. The lognormal distribution was used to represent uncertainty in the transfer and diet factors because it is a common distribution of selenium concentration data observed in nature. In addition, the lognormal distribution has several statistically desirable properties, such as estimating only positive values. It is a skewed distribution that produces rare large values more often than does the normal distribution. The best estimate of the transfer and diet factors was taken as the mean of the distribution, and uncertainty was expressed as a standard deviation or range. Simulations were run for each cleanup alternative for each species. The model also was run under past conditions with the application of drainwater to KR as a control.

Estimates of Key Species Population Sizes. The estimates of population densities and estimates of past KR-related mortalities (table 9-2) are based on data from published literature, unpublished surveys by FWS and California DFG, and consultation with personnel from these agencies and other local experts (USBR, 1986a). These data were provided to put in perspective the relative risks of selenium exposure of each population. The population data are indices of density, and in most cases the actual values are unknown. The risks of contamination-induced mortality vary greatly between these species. The data for mallards, American coot, and black-necked stilt suggest that these species are at low risk because of their small population at KR relative to San Joaquin Valley and statewide populations, even though both the American coot and black-necked stilt suffered significant losses at KR. In contrast, the tricolored blackbird population at KR represented more than one-half of the San Joaquin Valley population and one-third of the statewide population. Tricolored blackbirds are endemic to California; thus, the statewide population approximately represents the global population. Eared grebe and kit fox populations were not known at the site but were estimated by observation to be fewer than 20 each.

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *As detailed above, extensive data exist to demonstrate that the organisms were exposed to selenium. The Monte Carlo analysis is an important contribution to the analysis of exposure. The distributions are delineated in the supporting documentation.*

- *The food chain model is well documented and maximizes use of site-specific information on selenium concentrations in various food chain compartments.*

Limitations include:

- *The assessments of exposure are dependent on the transfer factors in the elaborate food chain models. Transfer factors determining the bioaccumulation of selenium are taken as constants. However, the uncertainty in the variability of these*

Comments on Analysis: Characterization of Exposure (continued)

transfer factors is high. An attempt to calibrate the prediction of the models using measured selenium concentration in birds would have reduced the uncertainty associated with the simplified model assumptions.

•There is little technical basis for the selection of the statistical distributions used in the Monte Carlo analysis.

9.3.4. Risk Characterization

This study contained two components of risk characterization: (1) attribution of adverse effects observed in KR to selenium and (2) prediction of the effectiveness of three remedial alternatives in reducing food chain exposures to selenium to below harmful levels for key species.

Attribution. Selenium was determined to be the most likely cause of the adverse reproductive effects in waterfowl at KR for two reasons. First, the effects observed at KR were similar to those produced in several avian species in the laboratory by administering excess selenium in the diet, whereas the other five contaminants of concern generally produce other types of adverse effects or their effects are unknown. Second, measurements of drainwater constituents in ground water, soils, and biota indicated that only selenium and boron were elevated in KR samples relative to control areas. Boron was excluded from the remainder of the assessment, however, because of limited toxicological information.

Risks Associated with Remedial Alternatives. The Monte Carlo simulation of the selenium transfer models estimated the probability distribution of selenium concentrations in the diets of the key species under each of the remedial alternatives (figures 9-4, 9-5 and 9-6). As an example, the results of the simulation for the FRP are presented in figure 9-4. For any combination of cleanup alternative and key species, the 50-percent probability level represents the dietary selenium concentration predicted from the mean transfer and diet factors for that particular condition. The uncertainty of the exposure estimate is shown by the probability distribution about the mean.

Using the mallard as an example, the 50-percent probability level represents a dietary selenium concentration of about 5 ppm (5 mg/kg in figure 9-4). Therefore, based on the selenium transfer model and on the uncertainty of transfer and diet factors, the FRP has about a 50-percent chance of resulting in a mallard dietary selenium concentration of less than 5 ppm.

Estimated selenium dietary concentrations were compared with threshold selenium toxicities for the key species. Tables 9-10a and 9-10b summarize the results for each cleanup alternative as the percentage of diet selenium predictions that are below estimated harmful levels for key species (USBR, 1986a). These results indicated that predicted risks are greatest for the FRP, less for

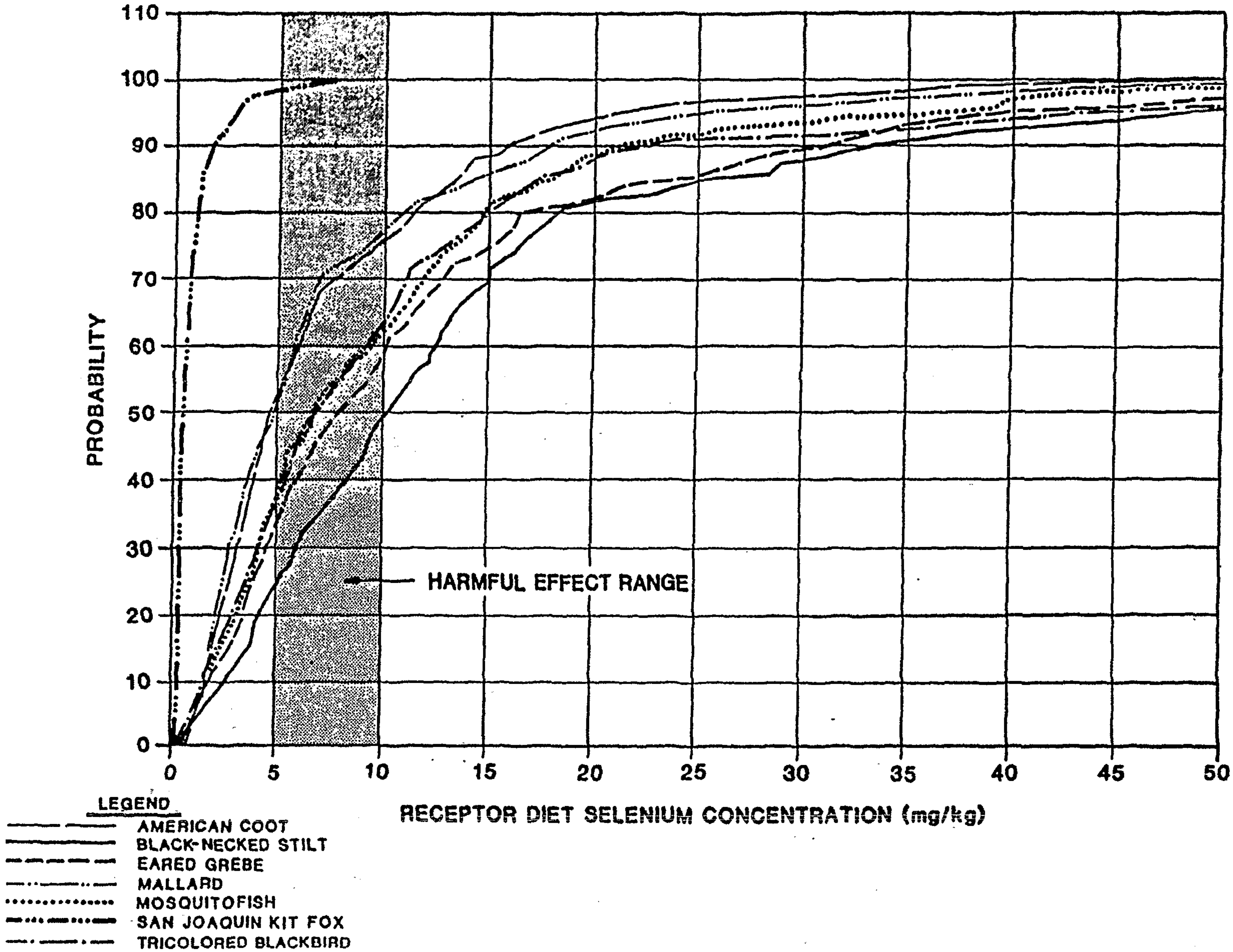

Figure 9-4. Probability distribution of predictions of selenium concentration in receptor species diet for Flexible Response Plan (USBR, 1986a)

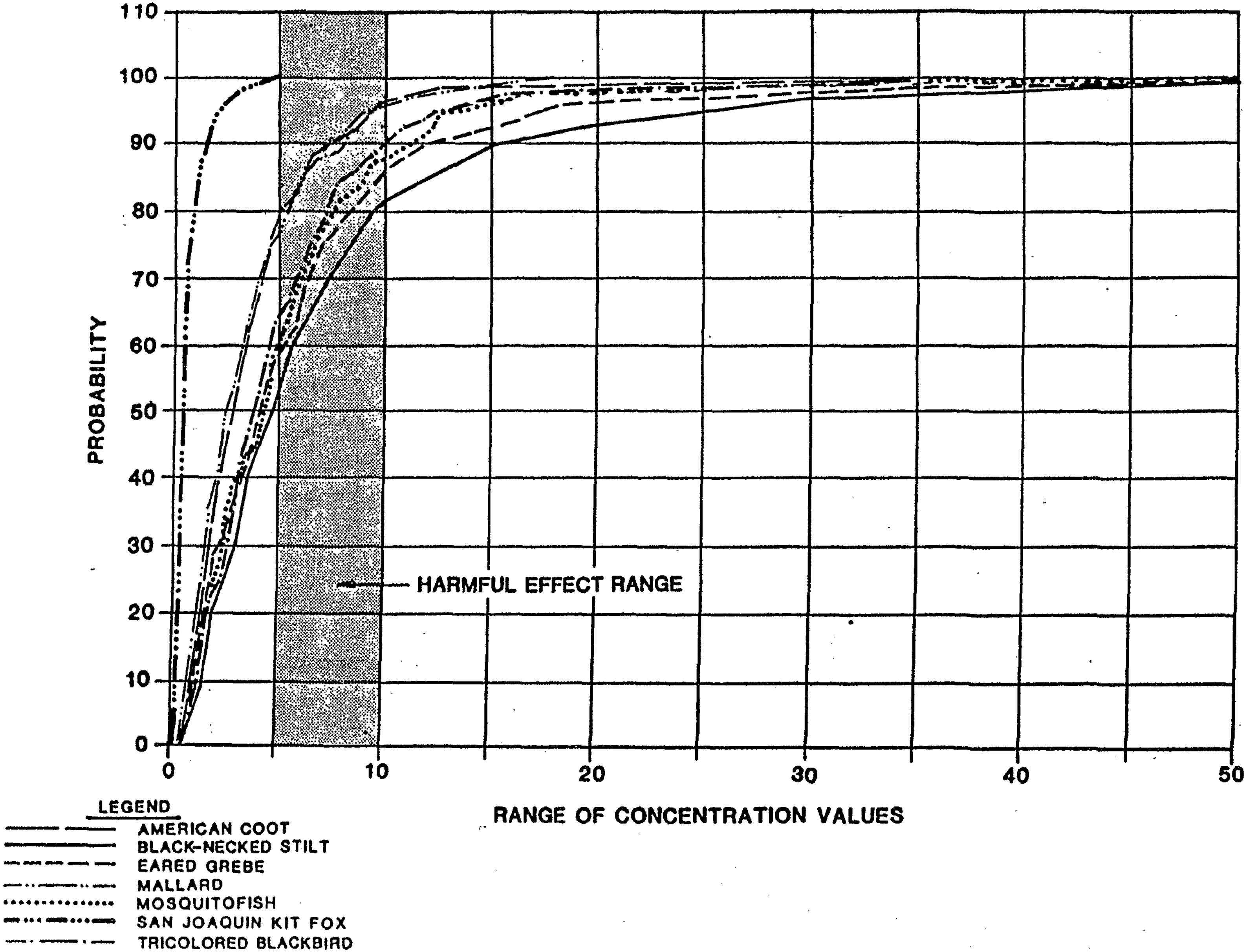

Figure 9-5. Probability distribution of predictions of selenium concentration in receptor species diet for Onsite Disposal Plan 1 (USBR, 1986a)

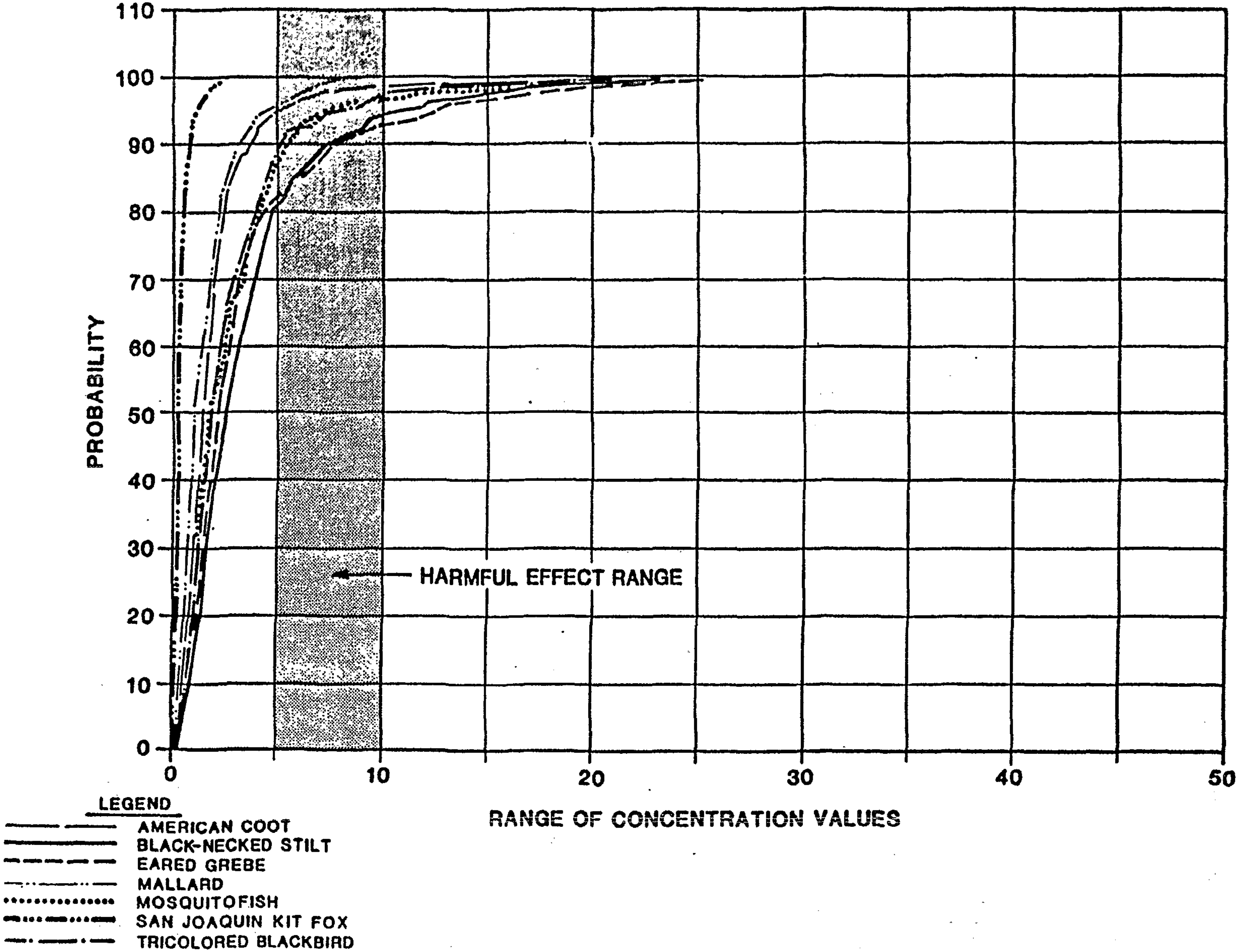

Figure 9-6. Probability distribution of predictions of selenium concentration in receptor species diet for Onsite Disposal Plan 2 (USBR, 1986a)

Table 9-10a. Percentage of Diet Selenium Predictions That Are Below the 5 mg/kg Estimated Harmful Level for Each Key Species and Each Cleanup Alternative (USBR, 1986a)

	Cleanup Alternative		
Key Species	FRP	Onsite-1[a]	Onsite-2[b]
Birds[c]	5	5	5
Mallards	50	75	95
Coots	50	80	95
Black-Necked Stilts	25	50	80
Tricolored Blackbirds	35	60	85
Eared Grebes	35	60	80
Mammals[c]	2	2	2
San Joaquin Kit Fox	90	95	~100
Fish[c,d]	3	3	3
Mosquitofish	20	35	70

Table 9-10b. Percentage of Diet Selenium Predictions That Are Below the 10 mg/kg Estimated Harmful Level for Each Key Species and Each Cleanup Alternative (USBR, 1986a)

	Cleanup Alternative		
	FRP	Onsite-1[a]	Onsite-2[b]
Birds[c]	10	10	10
Mallards	80	95	100
Coots	75	95	100
Black-	50	80	95
Necked Stilts	65	90	95
Tricolored	60	85	90
Blackbirds			
Eared			
Grebes			
Mammals[c]	5	5	5
San Joaquin	~100	~100	~100
Kit Fox			
Fish[c,d]	5	5	5
Mosquitofish	35	50	90

[a] 450,000 cubic yards.
[b] 1,000,000 cubic yards.
[c] Diet selenium concentration (mg/kg).
[d] *Mosquitofish not harmed by these selenium levels, although other species may be.*

Plan 1, and least for the Plan 2 remedial alternative. The results suggest that while none of the alternatives would clearly fail, none would clearly succeed in removing selenium risk to the key populations.

Uncertainties. The exposure estimates for each alternative are based on assumptions regarding the steady-state relationship between selenium concentration in sediments and the resulting concentration in surface water. Although this relationship is based on existing knowledge of selenium chemistry and field and laboratory experiments, the model results do not take into account the length of time necessary to achieve steady-state conditions.

The model does not describe uptake and loss rates of selenium by the components of the exposure pathways; rather, the model used transfer and diet factors observed in the laboratory and field. The uncertainty estimates of these factors are based on these observations, but they do not necessarily simulate exact conditions at KR.

Because insufficient information exists to develop quantitative dose-response relationships for diet selenium exposures specific for the key species at KR, the model results cannot be used to make quantitative estimates of the impact of cleanup alternatives to the exposed population. The toxicity profiles, however, can be used with the model results to determine the uncertainty of the relative safety of the cleanup alternatives.

Considerations for Other KR Organisms. The impact of each cleanup alternative on each species can be considered in terms of the fraction of the total population that resides at KR. Although only seven species were selected for selenium exposure evaluation, predicted impacts also may apply to other species that were not directly considered in the analysis because the trophic levels of the seven species are representative of those of a large number of species at KR. The transfer and diet factors (and associated uncertainties) and, thus, the selenium exposure estimates have broad applicability. Applicability for organisms at lower trophic levels is limited.

Habitat Changes. The model does not address the potential indirect effect of implementation of each alternative on wildlife populations as a result of physical changes in habitat. Each alternative would affect the habitat of KR to a variable degree. Furthermore, diet factors may change in the case of opportunistic organisms in response to changes in food availability brought about by implementation of a particular alternative.

Comments on Risk Characterization

Strengths of the case study include:

- *Given the constraints of the assumptions, the risk characterization is applicable. Certainly a strong point of this case study is the derivation of the probabilities as expressed in the figures.*

Comments on Risk Characterization (continued)

Limitations include:

- *Because of the narrow focus on selenium and terrestrial vertebrates in the KR study, some important factors may have been missed. Boron is not considered as a toxicant because of lack of data. The influence of habit alteration as a factor also is not properly addressed. This last factor is crucial because each of the cleanup alternatives involved major changes in habitat quantity and quality. To the extent that some habitat changes were known (e.g., drainage or covering), it may have been possible to model this into the risk assessment to provide a quantitative analysis along with uncertainty estimates.*

General comment:

- *In accordance with the Kesterson Program EIS (USBR, 1986), the Bureau implemented a phased approach to KR cleanup. During the course of the continuing research program that was conducted concurrent with the action, it became clear that an alternative remedial plan (which had not been considered in the EIS) would be most appropriate. Because that plan had not been identified before the risk assessment, it was not evaluated. In the end, the risk assessment indicated the inadequacy of alternatives and directed the responsible parties to consider site mitigation as the alternative of choice. KR was closed and converted to a terrestrial habitat. A much larger wetland was created to replace that lost at KR.*

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SECTION TEN

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

RISK CHARACTERIZATION METHODS USED IN DETERMINING THE EFFECTS OF SYNTHETIC PYRETHROIDS ON TERRESTRIAL AND AQUATIC ORGANISMS

AUTHOR AND REVIEWERS

AUTHOR

Greg R. Susanke
Office of Pesticide Programs
U.S. Environmental Protection Agency
Washington, DC

REVIEWERS

Arthur L. Buikema (Lead Reviewer)
Biology Department
Virginia Polytechnic Institute and State University
Blacksburg, VA

Joel Brown
Department of Biological Sciences
University of Illinois
Chicago, IL

Carolyn L. Fordham
Terra Technologies
Golden, CO

Christian E. Grue
Washington Cooperative Fish and Wildlife Research Unit
U.S. Fish and Wildlife Service
Seattle, WA

Clifford Hupp
Water Resources Division
U.S. Geological Survey
Reston, VA

Ronald J. Kendall
Institute of Wildlife and Environmental Toxicology
Clemson University
Clemson, SC

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LIST OF ACRONYMS

ASTM	American Society for Testing and Materials
BCF	bioconcentration factor
EEB	Ecological Effects Branch
EEC	estimated environmental concentration
EPA	U.S. Environmental Protection Agency
EXAMS	Exposure Analysis Modeling System
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
LC_{50}	lethal concentration to 50 percent of organisms tested
LD_{50}	lethal dose to 50 percent of organisms tested
LOEL	lowest observed effect level
MATC	maximum acceptable toxic concentration
NOEC	no observed effect concentration
NOEL	no observed effect level
OPP	Office of Pesticide Programs
PRZM	Pesticide Root Zone Model
SWRRB	Simulator for Water Resources in Rural Basins

ABSTRACT

The process by which the U.S. Environmental Protection Agency's Office of Pesticide Programs conducts ecological risk assessments for the registration and reregistration of pesticide products is presented in this case study, which focuses on a group of insecticides known as synthetic pyrethroids. The ecological risks associated with their application to cotton and sunflowers are assessed according to a combination of the quotient method (toxicity and exposure) and the weight of evidence (field studies). Although the case study considers ecological components in aquatic and terrestrial habitats, emphasis is placed on aquatic biota. The assessment endpoint is the health and survival of biological components or key species. Measurement endpoints include the following: (1) acute and chronic effects on aquatic invertebrates and fish, (2) system effects that adversely affect fish and aquatic invertebrate populations, and (3) potential adverse effects on breeding waterfowl and hatchlings due to reduction in the food base.

10.1. RISK ASSESSMENT APPROACH

The Ecological Effects Branch (EEB) of the Office of Pesticide Programs (OPP) has an established procedure by which it conducts risk assessments for the registration and reregistration of pesticides. The various components of these assessments are discussed in this case study. The order in which they are presented has been changed to conform to the case study format (figure 10-1). Risk assessments performed by EEB are usually pesticide specific, but for the purpose of this case study, a group of pesticides known as synthetic pyrethroids has been chosen.

To evaluate the risks associated with synthetic pyrethroid use, EEB has characterized the ecological effects and exposure for these chemicals. Ecological effects were characterized using acute and chronic laboratory toxicity testing and field studies. Exposure was evaluated based on estimates of residues in the terrestrial and aquatic environment likely to affect ecological components. Field studies were used to verify whether effects indicated by toxicity testing and exposure estimates did, in fact, manifest following applications of the chemical. Laboratory and field data were used with the exposure estimates to characterize the risk.

A strength of this case study is the broad and fairly complete toxicity data base that is obtained when data from the different generations of synthetic pyrethroids are combined. Exposure has been characterized by using measurements, estimates based on assumptions for fate and transport, and computer models. By focusing on synthetic pyrethroids as a group, this case study illustrates various methods that have been used to estimate exposure.

In addition to the laboratory studies of effects, the assessment of ecological risks due to pyrethroid application currently relies on the results of only two field studies. Other field studies have been required and are in progress or have been submitted and are under review.

Pyrethroid applications to cotton and sunflowers are used in the case study to illustrate the method by which risks have been assessed. Applications to cotton fields are discussed in some detail because they are the major site where synthetic pyrethroids are applied. Sunflowers are discussed because of the potential adverse effect on waterfowl in the prairie pothole region, the major sunflower production area.

10.2. STATUTORY AND REGULATORY BACKGROUND

A pesticide product may be sold or distributed in the United States only if it is registered or exempt from registration under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), as amended (1988, 7 U.S.C. 136 et seq.). Before a product can be registered unconditionally, it must be shown that it can be used without "unreasonable adverse effects on the environment" (FIFRA section 3[c][5]); that is, without causing "any unreasonable risk to man or the environment, taking into account the economic, social and environmental costs, and benefits of the use of the pesticide" (FIFRA section 2[bb]). The burden of proving that a pesticide meets this standard for registration is at all times on the proponent of initial or continued registration. If at any time the U.S. Environmental Protection Agency (EPA) determines that a pesticide no longer meets this standard for registration, the Administrator may initiate proceedings to cancel the registration under FIFRA section 6.

Figure 10-1. Structure of Analysis for Evaluating Risks Associated with Pyrethroids

PROBLEM FORMULATION

Stressors: several generations of pyrethroid pesticides.

Ecological Components: aquatic invertebrates, fish, terrestrial insects (bees), and waterfowl.

Endpoints: assessment endpoint is health of aquatic organisms and terrestrial wildlife. Measurement endpoints include acute and chronic toxicity tests and aquatic field studies.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
Models and simple algorithms were used to estimate a range of exposure concentrations. Measurements were made of residues.	Laboratory, mesocosm, and a few field studies were used to examine pyrethroid toxicity.

RISK CHARACTERIZATION

Estimated exposure concentrations were compared to Ecological Effects Concentrations (EECs) using the quotient method. Field and mecocosm studies were employed as part of an overall "weight-of-evidence" approach.

The special review process provides a mechanism through which the Agency gathers information about pesticides that appear to pose risks of adverse effects to human health or the environment. Risk evidence submitted to and/or gathered by the Agency must be evaluated and considered in light of benefit information. If the Agency determines that risks appear to outweigh the benefits, the Agency can initiate action under FIFRA to cancel, suspend, and/or require modification of the terms and conditions of registration.

10.3. CASE STUDY DESCRIPTION

10.3.1. Problem Formulation

Stressor. Synthetic pyrethroids may be applied to a wide variety of crops: alfalfa, barley, citrus, corn, cotton, fruit trees, nuts, oats, peanuts, potatoes, small fruits, soybeans, sugar beets, sugarcane, sunflowers, tobacco, vegetables, and wheat. Pyrethroids are also applied to anthills, lawns, livestock yards, ornamentals, pine, conifers, rangeland, turf, and for mosquito control. These uses for pyrethroids are conditionally registered pending the outcome of data requirements. Pesticides that have been conditionally registered (FIFRA section 3 [c][7]) are allowed to be applied throughout the duration of the registration process. Basically, the registration decision weighs both the economic consequences of denying registration and the environmental effects of granting registration. At this time, only permethrin has been unconditionally registered for agricultural crop uses (1982). Data requirements needed to support registration are based on a tiered testing system (appendix A).[1]

Since their first review by EEB in the 1970s, synthetic pyrethroids have caused a great concern for potential adverse effects on the aquatic environment, including a reduction in numbers of micro- and macroinvertebrates and fish. The available data on synthetic pyrethroids show that these compounds, as a class of insecticides, all appear to be highly toxic to aquatic organisms, both acutely and chronically. They also have a high potential to persist in water and sediment and to bioaccumulate in aquatic organisms. EEB believes that all synthetic pyrethroids are likely to exhibit these properties. Consequently, EEB has proposed a specific set of aquatic organism studies that will be required to support the registration of all synthetic pyrethroids. Based on EEB's experience in reviewing synthetic pyrethroids, many of the data that were specified as reserved (upper tier tests) are now required. These include: (1) a freshwater aquatic invertebrate life-cycle study, (2) an estuarine/marine invertebrate life-cycle study, (3) a fish full-life-cycle study, (4) an aquatic organism accumulation study, and (5) simulated (mesocosm) or actual (pond) aquatic field studies.

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[1]The manufacturer (registrant) of a specific pesticide is responsible for submitting the data necessary to support registration. Data requirements in the first three tiers of the testing protocol are bioassays. Initially, acute toxicity tests are conducted. Depending on the acute toxicity values, subchronic and chronic toxicity tests may be required (see Preston and Hitch, 1982, for a determination of when chronic testing is required). The last tier is the requirement for simulated or actual field testing. Field testing is required on a chemical-by-chemical basis, as determined by the available information on potential effects and exposure. The field study is used to verify if effects indicated in the bioassays manifest in the field applications.

EEB is also concerned with the adverse effects of synthetic pyrethroids on waterfowl (U.S. EPA, 1987b). It has concluded that a reduction in the diversity of species and number of organisms of an aquatic habitat may affect waterfowl recruitment and hatchling survival because of the reduced prey base.

Ecological Components. This case study focuses on aquatic invertebrates and fish at the individual, population, and community levels; waterfowl; and a few target species (e.g., bees) of terrestrial habitats.

Endpoint Selection. The assessment endpoint is the health of aquatic organisms and terrestrial wildlife. Measurement endpoints include laboratory acute and chronic toxicity tests with selected species (aquatic fish and invertebrates, nontarget insects, and waterfowl) and aquatic field tests (mesocosm and pond studies).

Comments on Problem Formulation

Strengths of the case study include:

- *The assessment outlines a framework of sequential testing within which field data are used to supplement initial toxicity data in order to verify potential effects. This is especially important for identifying ecological effects that could not have been predicted from laboratory studies.*

- *The assessment considers potential indirect effects as well as the direct toxic effects of the chemicals. It notes the importance of considering food chain effects as they relate to biomagnification and potential secondary effects such as the reduction in the food base for higher trophic levels.*

Limitations include:

- *The potential effects on waterfowl due to a reduced prey base were considered to be an important issue, but it was recognized that there are few data available for evaluating these effects. This observation indicates the kinds of information that may need to be developed in the future. The endpoint should be effects on waterfowl productivity rather than direct toxicity.*

- *Original laboratory toxicity test and field data were not provided for peer review.*

10.3.2. Analysis: Characterization of Ecological Effects

Overview of Test Methods. Several aquatic and avian species commonly used in toxicity testing have been chosen by EEB to represent all aquatic organisms and terrestrial wildlife in the United States. It is understood that these species are not necessarily the most sensitive to all pesticides tested, but they are generally recognized as good indicator species. Those species chosen by EEB have met selection criteria (Urban and Cook, 1986) and are consistent with the recommendations of various sources such as the American Society for Testing and Materials (ASTM) standards (1980, E 729-80, E 1022-84), the Committee on Methods for Toxicity Tests with Aquatic Organisms (1975), and the National Water Quality Laboratory Committee on Aquatic Bioassays (1971). Different indicator species are used depending on the bioassay performed. Typical species used as indicators for each test type (tiers 1 through 3) are listed in the tables presented in the Analysis: Characterization of Exposure section.

In mesocosm testing, the indicator fish species used is the bluegill. No predetermined aquatic invertebrate species are used as indicators. The species that are used are those found in similar aquatic habitats near the study site. Actual pond or terrestrial field studies investigate the effects to naturally occurring species of fish, aquatic invertebrates, birds, and small mammals.

Current policy is that toxicity testing not be performed on amphibians and reptiles. It is assumed that they are protected when fish, mammals, and birds are protected. Mammalian toxicity testing is performed for the human risk characterization process. The results are also used by EEB. The species used to represent mammalian wildlife are the laboratory rat, mouse, rabbit, and dog. Nontarget insect toxicity data are required for outdoor uses that may result in honey bee exposure. Cotton and sunflowers are such uses. Honey bees were selected to represent nontarget insects because of their economic importance.

Laboratory Testing. Numerous bioassays have been performed on synthetic pyrethroids by various sources under various testing protocols. However, for the registration of pesticides the bioassays must follow the protocols described in the Pesticide Assessment Guidelines: Subdivision E Hazard Evaluation—Wildlife and Aquatic Organisms (Preston and Hitch, 1982) and Subdivision L Hazard Evaluation—Nontarget Insects (Hitch, 1982). These protocols are consistent with standards recommended by ASTM (1980, E 729-80, E 1022-84), the Committee on Methods for Toxicity Tests with Aquatic Organisms (1975), and the National Water Quality Laboratory Committee on Aquatic Bioassays (1971). The aquatic and avian toxicity values presented in the following tables were obtained from bioassays that were conducted according to these protocols and thereby satisfied data requirements.

The toxicity values are presented as the LC_{50}, LD_{50}, and MATC. The LC_{50} is the median lethal concentration or the concentration at which 50 percent of the test organisms die. The LD_{50} is the median single lethal dose or the dose at which 50 percent of the test organisms die. The MATC is the maximum acceptable toxic concentration or the range of concentrations between the no observed effect concentration (NOEC) and the lowest observed effect concentration (LOEL). The MATC for chronic studies is a function of survival, reproduction, and growth. Toxicity values are expressed as milligrams per liter (mg/L), micrograms per liter (μ/L), or nanograms per liter (ng/L) for pyrethroid concentrations in water and as milligrams per kilogram (mg/kg) or micrograms per kilogram (μg/kg) for concentrations in nonaqueous media.

Synthetic pyrethroids are highly toxic to all freshwater organisms. Tables 10-1, 10-2, and 10-3 incorporate acute and chronic toxicological data from the various synthetic pyrethroids. The toxicity values presented show the approximate ranges found within each study type. These values come from EEB and HED (Health Effects Division; mammalian data) reviews of studies submitted to OPP. Both the studies themselves and the reviews are unpublished and generally unavailable to the general public except via the Freedom of Information Act. The typical indicator test species are listed in parentheses.

Synthetic pyrethroids also appear to be very highly toxic to marine and estuarine organisms. The ranges of acute and chronic toxicity values among the various generations of synthetic pyrethroids tested to date are presented in table 10-2 for each study type. The typical indicator test species are found in parentheses.

Synthetic pyrethroids are slightly toxic to practically nontoxic to birds and are moderately toxic to practically nontoxic to mammals. The ranges of LD_{50} and LC_{50} toxicity values among the various generations of synthetic pyrethroids tested to date are presented in table 10-3 for each study type. The typical indicator test species are listed.

There is wide variation in avian reproductive effects among synthetic pyrethroids. The avian NOELs are as high as 900 to 1,000 ppm for fluvalinate (study I.D. 073443, 1985) and cyfluthrin (study I.D. 254820, 1985), respectively. But fenvalerate (study I.D. 96385, 1978) increased bobwhite quail eggshell cracking at < 25 ppm, and cyhalothrin (study I.D. 073989, 1988) affects the number of viable embryos, hatchlings, and 14-day survivors of eggs incubated (set) at 50 ppm (LOEL).

Synthetic pyrethroids are generally highly toxic to honey bees on an acute contact LD_{50} basis. Restrictive labeling is required.

Field Testing. EEB has determined that aquatic field studies are required to complete the characterization of ecological effects for synthetic pyrethroids. These tests are required to verify results of laboratory bioassays. If no significant effects are observed in acceptable field studies at the estimated environmental concentration (EEC), then it will be concluded that there are no aquatic risks at this exposure level.

At the time this case study was prepared, one simulated (mesocosm) and one actual (pond) field study were found to be scientifically acceptable. These were the studies for lambda cyhalothrin and bifenthrin, respectively. The remainder of the field studies for other synthetic pyrethroids are either pending or currently under review. The results discussed in this section come from EEB reviews of field studies submitted to OPP. Both the studies themselves and the reviews are unpublished and generally unavailable to the general public except via the Freedom of Information Act.

Table 10-1. Toxicity of Synthetic Pyrethroids to Freshwater Organisms

Study	Test	Toxicity	Chemical	EPA I.D. Number	Year of EEB Review
Acute					
Bluegill	LC_{50}	0.13 ppb	Tefluthrin	261402	1986
	LC_{50}	2.8 ppb	Tralomethrin	072123	1984
Rainbow trout	LC_{50}	0.06 ppb	Tefluthrin	261402	1986
	LC_{50}	9.8 ppb	Permethrin	096699	1978
Invertebrate (*Daphnia magna*)	LC_{50}	0.038 ppb	Tralomethrin	UCCES 11507-74-03	1981
	LC_{50}	8.3 ppb	Flucythrinate	098288	1979
Chronic					
Fish early life stage (fathead minnow)	NOEL	>64 pptr	Fluvalinate	250141	1983
	LOEL	<410 pptr	Permethrin	096699	1978
Fish life cycle (fathead minnow)	NOEL	>31 pptr	Lambda Cyhalothrin	41519001	1990
	LOEL	<95 pptr	Bifenthrin	40791301	1989
Invertebrate life cycle (*Daphnia magna*)	NOEL	>1.3 pptr	Bifenthrin	41156501	1989
	LOEL	<45 pptr	Fluvalinate	250141	1983

Table 10-2. Toxicity of Synthetic Pyrethroids to Marine and Estuarine Organisms

Study	Test	Toxicity	Chemical	EPA I.D. Number	Year of EEB Review
Acute					
Fish (sheepshead minnow)	LC_{50}	0.13 ppb	Tefluthrin	40161137	1987
	LC_{50}	17.5 ppb	Bifenthrin	264646	1987
Mollusc (eastern oyster)	LC_{50}	3.42 ppb	Cyfluthrin	262443	1987
	LC_{50}	>1 ppm	Tefluthrin	40161135	1987
Crustacean (mysid shrimp)	LC_{50}	3.97 pptr	Bifenthrin	264647	1987
	LC_{50}	840 pptr	Tralomethrin	070692	1982
Chronic					
Invertebrate life cycle	NOEL	>0.17 pptr	Cyfluthrin	262443	1988
(mysid shrimp)	LOEL	<0.93 pptr	Tralomethrin	264510	1987
Fish early life stage	NOEL	>24 pptr	Cyfluthrin	265895	1987
(sheepshead minnow)	LOEL	<620 pptr	Cyfluthrin	265895	1987

Table 10-3. Toxicity of Synthetic Pyrethroids to Birds and Mammals

Study	Test	Toxicity	Chemical	EPA I.D. Number	Year of EEB Review
Rat	LD_{50}	56 mg/kg	Lambda Cyhalothrin	259805	1986
		1,000 mg/kg	Cyfluthrin	072008	1982
Mallard duck	LD_{50}	1,089 mg/kg	Fenpropathrin	249939	1983
		9,932 mg/kg	Fenvalerate	96385	1978
Northern bobwhite quail	LD_{50}	1,800 mg/kg	Bifenthrin	AROTAL06	1984
		2,510 mg/kg	Fluvalinate	241388	1980
Mallard duck	LC_{50}	1,280 ppm	Bifenthrin	AROTAL05	1984
		12,488 ppm	Cyhalothrin	073221	1985
Northern bobwhite quail	LC_{50}	2,354 ppm	Cyhalothrin	073221	1985
		15,000 ppm	Tefluthrin	261402	1986

Ideally, a field study should be conducted for each pesticide use. However, this is often impractical due to cost or time limitations. Therefore, field studies should be designed to incorporate as many uses as possible without compromising the study or limiting the ability to make regulatory decisions at the environmentally relevant concentrations. The lambda cyhalothrin mesocosm that was conducted adjacent to cotton fields was designed in this manner. The tested concentrations provide a range that includes the estimated concentrations of several different uses.

The loading concentrations of lambda cyhalothrin into adjacent waterways from cotton fields were determined from modeling (see Analysis: Characterization of Exposure). The concentrations are approximately equal to the levels used in the middle- and high-dose ponds of the mesocosm. There were significant effects on invertebrates and fish populations at these dose levels and also at the lowest dose level, which was equivalent to 1.7 percent of the exposure expected in ponds. Residues of 58 μg/kg were found in sediments tested for lambda cyhalothrin 2 months after the last application.[2] The chemical has also been shown to bioaccumulate in fish. Tests were performed with carp because it is a bottom feeder, thereby having direct contact with residues in the sediment. Residue concentrations in whole fish were 4,600 to 5,000 times greater than those found in the sediment.

A mesocosm study attempts to incorporate the structure and function of an ecosystem (Touart, 1988). The results of the lambda cyhalothrin mesocosm (U.S. EPA, 1989) show that invertebrate populations in the ponds were reduced in numbers when compared with the control ponds. Some groups of invertebrates in the high-dose ponds were essentially decimated after one or two treatments. Reduced numbers at all dose levels were evident into the posttreatment period. The fish in each treatment group had statistically significant lower weights and body lengths than those in the control group. The weights were reduced 20 to 30 percent for all fish and 30 to 40 percent when only the first year (young-of-the-year) fish are considered. This slowing of growth may affect maturation, reproduction, and ultimately the population structure. These effects occurred at all concentrations tested; the lowest concentration was 0.6 to 1 ng/L, which is 0.01 of the label application rate for cotton.

This mesocosm study has confirmed the toxic effects indicated by laboratory testing. The use of lambda cyhalothrin can reasonably be anticipated to cause significant adverse effects to the aquatic ecosystem. There were no additional data requirements for the registration of lambda cyhalothrin.

A pond study with bifenthrin (U.S. EPA, 1990, 1991) investigated effects to aquatic organisms resulting from application to cotton according to label instructions. The results have qualitatively shown that bifenthrin has the potential to "change the natural balance and degrade the ecological integrity of aquatic ecosystems." However, due to the lack of replications in this study, a quantitative assessment could not be made.

Bifenthrin was found to be more persistent than any other synthetic pyrethroid. For more than a year following application, residues were measurable in water (4 ng/L), sediment (37 μ/kg),

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[2]Units for sediments and tissues are on a mass wet weight basis.

and all 28 fish species analyzed (5 to 9 μg/kg except gizzard shad, which exhibited 78 μg/kg). There was also a severe reduction in survival and reproductive potential of *Daphnia* and snails. Zooplankton such as calanoid copepods were eliminated. Macroinvertebrate populations were also affected. There was a severe reduction in the chironomid insect population, and the entire population of mayflies *Caenis* and damselflies *Enallagma* were eliminated after the first application. Mayflies remained extremely rare throughout posttreatment, thereby indicating that recovery would take longer than a year.

Although bifenthrin did not cause high mortality or overt reproductive failure in bluegill sunfish, it did cause adverse effects on the population. A significant reduction occurred in the condition factor (physical condition of the fish) and in the growth rate of juvenile bluegill.

Gizzard shad were also adversely affected. Almost the entire population of shad (1,600) died the winter following application. At the time of the kill, residue measurements in shad reached 440 μg/kg. This was 55,000 to 147,000 times greater than the pond water (0.003 to 0.008 μg/L) and 9 to 11 times greater than the sediment (40 to 50 μg/kg). The field-measured bioconcentration factors (BCFs) from water approximate the estimated BCF of 50,000 (5 percent of the octanol-water partition coefficient) and the BCFs calculated from laboratory bioaccumulation studies. In these studies, bluegill sunfish were exposed to 1.2 ng/L concentration bifenthrin 6,090 times (whole body measurement). It could then be estimated that fish would bioaccumulate bifenthrin 15,000 to 41,000 times the water concentrations that were present at the time of the kill (0.003 to 0.008 μg/L). The various BCFs are similar despite the differences between laboratory and field conditions and the species measured.

It was concluded that the shad were not able to metabolize and excrete bifenthrin, but rather accumulated it in fatty tissues or reproductive organs. It was hypothesized that at times of stress, i.e., cold temperatures and/or diminished food supply, the shad relied on stored energy and, in the process, the residual bifenthrin was released at toxic concentrations. The data also indicated that stored residue may adversely affect reproduction (no young-of-year class). Given the persistence and bioaccumulative potential of bifenthrin, many other aquatic species may experience the same physiologic response as the gizzard shad.

EEB is very concerned about bifenthrin's apparent potential to cause devastating and lasting effects to aquatic ecosystems and population structure. Bifenthrin is extremely toxic; i.e., a NOEL of less than 1 ng/L, so low that the pesticide cannot be detected by traditional analytic means at concentrations that elicit a biological effect. It is apparent from the pond study that pesticide residues as well as biological effects persist longer than 1 year after application ceases. It is EEB's opinion that bifenthrin has the potential, in the long term, to cause extreme population shifts in aquatic ecosystems and possibly eradication of certain aquatic species.

The results of a residue monitoring study with cypermethrin applied to cotton (U.S. EPA, 1982) showed that repeated aerial applications (16) at 0.125 lb a.i./A at 5-day intervals will result in cypermethrin residues in a stream ecosystem 8 miles downstream from the treated fields. During the study there were three runoff events. Maximum cypermethrin residues found in the stream were 24 ng/L at 165 meters from the point of runoff, 13 ng/L at 2 miles from the point of runoff, and 3 ng/L 8 miles from the point of runoff. Because of the results of this residue monitoring study, an actual pond study was required as the condition for a cotton registration. The pond study has since been found to be invalid, and a mesocosm study is currently in progress.

Field studies are typically designed to investigate the ecological effects from a single application season. These effects can be qualitatively and quantitatively measured; however, multiple-year testing would be required to determine long-term effects on affected populations. At this time the extent of the population effect is not required by the testing to characterize risk. A limiting factor in the investigation of long-term effects is the lack of scientific knowledge about how to interpret the significance of effects manifested at one level of organization (e.g., on individuals or local populations) on higher levels of organization (e.g., populations, communities, and ecosystems). Because of the lack of understanding of ecological systems, the reviewer of a field study must rely on good scientific judgment in interpreting the results.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *Acute and chronic laboratory toxicological data for fish and aquatic invertebrates are scientifically sound, and the amount of data is sufficient to establish general toxicity levels for synthetic pyrethroids.*

Limitations include:

- *EEB does not have field study data quantifying population effects of synthetic pyrethroids to waterfowl. A study of this magnitude would require multiple-year testing and would need to incorporate natural seasonal predator and prey population fluctuations. The lack of scientific information in this area would make a study of this magnitude difficult to evaluate.*

- *Systemwide effects on aquatic components could be evaluated only on the basis of two field studies. The other field studies still are being conducted or are in review.*

- *Because synthetic pyrethroids adsorb to sediment, laboratory bioassays need to be developed to address the effects of sediment residues on benthic dwelling micro- and macroinvertebrates. The probability of chemicals becoming released by the sediment and becoming resuspended into the water column also needs to be investigated.*

- *The field study does not consider long-term recovery of affected populations, the effects of multiple-year exposures, or long-term impact on fish reproduction during the second year. Under the current field test protocols, fish populations are exposed after the peak reproduction of fishes; no information is available on the effects of bioaccumulation on egg and sperm production or spawning success.*

Comments on Analysis: Characterization of Ecological Effects (continued)

- *Ecological function is not well understood, and the ability to examine interactions among organisms is not well developed. It is not known what level of an effect is needed to sustain a long-term impact on fish and waterfowl populations. The high natural variability of biota, both temporally and spatially, within and among ponds in the field studies also adds to the uncertainty because it may be difficult to separate out the effects of a chemical.*

General comments:

- *Several questions were raised by reviewers on this assessment. What uncertainties exist with the current selection of test species and methods? Is the data base adequate or should other tests be required on a case-by-case basis? Will addition of other tests reduce uncertainty in the risk assessment process? For example, should sediment bioassays be required because synthetic pyrethroids bind to soil sediments and are persistent? Because field studies have identified the loss of sensitive cosmopolitan species, such as mayflies, should these animals be suggested as additional test species? Because synthetic pyrethroids are known to affect chlorophyll concentrations, should toxicity tests be conducted with phytoplankton? Reviewers indicated that it was not reasonable to assume that the toxicity data for fish will protect reptiles or amphibians. Amphibian toxicity test methods exist and could be used. Biomagnification of pesticides may occur in snakes if they are the top carnivore in the test area.*

- *Aquatic invertebrate populations have been adversely affected. The population of some species has been severely reduced and in others it has been completely eliminated. Based on available information, population effects to fish are anticipated because there was a reduction in biomass and a slowing of fish growth. These effects will delay reproduction and thereby cause population effects. The extent to which the population will be affected is unknown. Multiple-year testing would be necessary in order to assess the long-term impact on fish and aquatic invertebrate populations.*

- *Space limitations prevented a more thorough discussion of the laboratory and field studies methodologies. For the field studies, a brief overview of a typical mesocosm protocol would have been useful to emphasize the statistical considerations used in the experimental design, such as the use of replicated treatment and control ponds and the temporal and spatial considerations that drive the sampling of biota, water, and residues.*

10.3.3. Analysis: Characterization of Exposure

Approximately 10 million acres of cotton were harvested in 1987 (U.S. Department of Commerce), and the largest cotton-producing states were Texas, California, and Mississippi. The bulk of the United States cotton crop is planted during April but may not be completed until mid-June, depending on seasonal conditions. Synthetic pyrethroids may be sprayed on cotton as early as 7 days after planting.

Approximately 2 million acres of sunflowers were harvested nationwide in 1987 (U.S. Department of Commerce). Sunflowers are grown in much the same way as corn. Planting dates in the Northern Great Plains begin May 1 and extend through late June. Synthetic pyrethroid treatments are applied as necessary throughout the season up to 28 days preharvest (early September).

Pesticide residues expected to be found in terrestrial and aquatic ecosystems are calculated by various methods that will be explained in subsequent sections. These calculations are expressed as the EEC.

EECs are derived from worst-case scenarios for single or multiple applications over a single season. A worst-case scenario incorporates practical extremes in determining the EEC, such as using the maximum application rate, lowest application interval, maximum residue concentration, etc. This is done to provide a margin of safety for most application situations. Several synthetic pyrethroids will be used as examples in these calculations.

Terrestrial Exposure. EEB has developed a nomograph (Urban and Cook, 1986) from Hoerger and Kenaga (1972) and Kenaga (1973). These calculations estimate pesticide residues found on terrestrial wildlife food items. Residues are determined according to the rate of a single spray application and the surface area to which it is applied. The application of lambda cyhalothrin is used as an example. It can be applied at 0.03 lb active ingredient per acre (a.i./A) (maximum label rate for cotton). The residues shown in table 10-4 are maximum values expected to be found on these various substrates immediately after application (U.S. EPA, 1988).

A computer program designed by R. Lee (unpublished) of EEB is used to calculate the daily accumulated terrestrial or aquatic concentrations from multiple applications and degradation rates. It is based on the first order kinetic rate equation (Williams and Williams, 1967). Multiple applications of lambda cyhalothrin, up to a maximum of 0.2 lb a.i./A per season, can be applied to cotton by ground or air equipment. To calculate lambda cyhalothrin residues on cotton leaves from multiple applications, the following information was used: an initial estimated residue of 4 mg/kg, which is found on leaves and leafy crops (as determined from the previous nomograph example), an application interval of 3 days (as stipulated on the label), 7 applications per season (maximum application rate per season 0.2 lb a.i./A ÷ maximum rate per application of 0.03 lb a.i.), and a 23-day foliar half-life (determined by the equation presented below). The average and maximum residues estimated from this model are 15 and 22 mg/kg, respectively. The maximum residues on small insects was 11 mg/kg, and the average residue was 7 mg/kg (U.S. EPA, 1988).

Table 10-4. Residues Found on Various Substrates After Application (U.S. EPA, 1988)

Substrate	Maximum Residues (mg/kg)
Short range grass	7
Long grass	3
Leaves and leafy crops (cotton)	4
Forage (alfalfa, clover) and small insects	2
Pods containing seeds (sunflower seeds) and large insects	4
Fruit	0.21
Soil (top 0.1 inch)	0.66
Top 6 inches of water (direct application)	0.022

Because a foliar half-life bioassay had not been conducted, the 23-day foliar half-life value was estimated from information obtained from residue chemistry testing. The following formulas, which are based on the first-order kinetic rate equation, were used.

$$K \text{ (decay rate)} = \frac{\log \text{ (original concentration)} - \log \text{ (remaining concentration)}}{t \text{ (time elapsed in days)}}$$

The relationship between the decay rate (K) and the decay half-life (t1/2) is

$$t1/2 \text{ (foliar half-life)} = \frac{0.0693 \text{ (constant)}}{K}$$

Aquatic Exposure. To obtain an approximation of an aquatic EEC, the (unpublished) algorithms presented below are used. The EEC is determined for a 1-acre pond that receives pesticide runoff and drift from the surrounding 10 acres that have been treated once by aerial or mist blower application. Cyfluthrin's maximum label application rate of 0.1 a.i./A on cotton will be used as an example (U.S. EPA, 1987a). Synthetic pyrethroids adsorb strongly to soil particles; therefore, their potential to runoff is lessened. However, as in the case of cyfluthrin, the soil half-life is approximately 60 days. Therefore, the runoff event does not have to occur immediately after application in order for residue concentrations to reach toxic levels. Nevertheless, aquatic exposure is most likely to result from spray drift.

Runoff Contribution for EEC

Runoff = A * B * C * D

where:

A = the amount of active ingredient applied (0.1 lb)

B = the application efficiency (0.6 or 60 percent was chosen)

C = the runoff coefficient (0.001 or 0.1 percent was assumed)[3]

D = the area of drainage basin (10 acres was selected)

Using the above equation, 0.0006 lb of active ingredient was estimated in runoff.

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[3]The percentage runoff is usually 1, 2, or 5 percent. These percentages correspond to the solubility of the pesticide, respectively <10 mg/L, 10 to 100 mg/L, and >100 mg/L. Because the solubility of synthetic pyrethroids is extremely low (cyfluthrin = 0.002 mg/L), 0.1 percent was used.

The EEC in a 1-acre pond was estimated for two water depths (6 feet and 6 inches) as follows:

6 feet deep = 61 μg/L (conversion factor)[4] $\times$ 0.0006 lb a.i. = 0.037 μg/L

6 inches deep = 734 μg/L (conversion factor) $\times$ 0.0006 lb a.i. = 0.44 μg/L

Drift Contribution for EEC

Drift = A * E

where:

A = the amount applied (0.1 lb)

E = the amount of drift (0.05 or 5 percent was used)[5]

Using the above equation, a drift of 0.005 lb of active ingredient was estimated.

The EEC in a 1-acre pond due to drift was estimated as follows:

6 feet deep = 61 μg/L $\times$ 0.005 lb a.i. = 0.305 μg/L

6 inches deep = 734 μg/L $\times$ 0.005 lb a.i. = 3.67 μg/L

If pertinent environmental fate data are available, the EEC can be calculated using computer simulation models. These are the Simulator for Water Resources in Rural Basins (SWRRB) model (Arnold et al., 1990), the Pesticide Root Zone Model (PRZM; Carsel et al., 1984), and the Exposure Analysis Modeling System (EXAMS II) (Burns, 1990). SWRRB and PRZM are surface-water runoff (hydrologic) models that estimate loading or the amount of residue that will enter a body of water either adsorbed to soil particles or having been desorbed from them. The environmental fate half-lives for photolysis, soil metabolism, and soil adsorption are necessary to run this model. EXAMS II is a chemical fate (kinetic) model that estimates the chemical fate and concentration of a pesticide once it has reached a body of water. Calculations are adjusted according to whether the water is lotic or lentic as in the case of streams or ponds. The environmental fate half-lives for photolysis, hydrolysis, microbial breakdown, and sediment adsorption are necessary to run this model. The models can be run to determine the highest

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[4]61 μg/L and 734 μg/L are standard concentrations calculated from 1 lb of pesticide per acre applied to water with a depth of 6 feet and 6 inches, respectively.

[5]On average, 5 percent (range is 2 to 10 percent) of the application rate per acre can be expected to drift from a field without a buffer zone to the center of an adjacent 1-acre pond (i.e., 105 feet from the edge of the field) (Akesson and Yates, 1964; Garner and Harvey, 1984). Nigg et al. (1984) also reported 5 percent drift.

maximum EEC found immediately after application or the average EEC over a given time period. The average EEC could be determined for single or multiple applications throughout the time period. The average EEC calculated over a given time period of 21 or 32 days, for example, could then be compared to the effect levels in the *Daphnia magna* Life Cycle (21 days) Chronic Toxicity Test or the Fish Early Life Stage Test (32 days).

Cotton. Cyfluthrin is used as an example to calculate the aquatic EEC on cotton (U.S. EPA, 1987a). Based on the SWRRB/EXAMS II results, the initial concentration may reach 500 ng/L in the water column of a 1-acre pond 6 feet in depth, which receives runoff and some spray drift from nearby cotton fields. The dissolved concentration reaching streams exiting the ponds would be much less (20 ng/L). These estimates are based on an application rate of 0.089 lb a.i./A, 9 applications per year with a 5-day spray interval, and a half-life of 193 days at pH 7.

Sunflowers. SWRRB/EXAMS II model estimates for another synthetic pyrethroid, tralomethrin, show water column concentrations of up to 8,800 ng/L in peaks that dissipate rapidly, and concentrations of up to 50 ng/L that persist more than 7 days. A drift of only 1 percent would result in concentrations in temporary potholes of more than 150 ng/L (based on 0.01 × application rate applied directly to 1 acre with 6-inch average depth). Water residues measured in an actual pond study were as high as 37 ng/L (U.S. EPA, 1987b).

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *The EECs were determined from various methodologies. The predicted adverse effects based on toxicity data and EECs were confirmed by field testing.*

Limitations include:

- *The characterization of exposure would have benefited by the inclusion of information on numbers of acres per crop that could be sprayed with synthetic pyrethroids, the range of concentrations used per application, the maximum seasonal concentration that can be applied, the number of applications per growing season, the interval between applications, the method of application, and the relationship between application and timing of various ecological parameters of interest (e.g., fish reproduction).*

- *It was difficult to judge the reasonableness of the exposure estimates.*

10.3.4. Risk Characterization

Risk to Ecological Components in Aquatic Environment. The risk characterization methods used in this case study are a combination of the quotient method and the overall weight of evidence. OPP has presented data showing that aquatic organisms could be exposed during applications to synthetic pyrethroids at levels that equal or exceed acute and chronic toxicity values. In the case of cotton use, the highest instantaneous EEC is 500 ng/L. Actual measured residue concentrations are included for bifenthrin. The lowest LC_{50} (acute) values ranged from 2.4 to 38 ng/L, and the lowest NOELs (chronic) ranged from 0.17 to 46 ng/L. (Note that the same species were not necessarily used in both acute and chronic testing). The number of times the EEC or actual measured residues exceed toxicity concentrations is presented in table 10-5.

The EECs for these synthetic pyrethroids exceed the acute toxicity levels by factors ranging from 1 to 208 times and the chronic toxicity levels from 1 to 3,000 times. Synthetic pyrethroid residues found in the water column can be expected to be adsorbed to the sediment. Aquatic sediment testing has revealed residue concentrations of lambda cyhalothrin at 58 µg/kg 2 months after application, and 37 µg/kg of bifenthrin a year after application. Even if synthetic pyrethroid residue concentrations in water or sediment are below acute toxicity levels, it has been shown that fish can bioaccumulate lambda cyhalothrin residues 4,600 to 5,000 times and bifenthrin residues 56,000 times (bioconcentration factor). Because of bioaccumulative capabilities of these chemicals, persistence becomes more of a concern than the ambient water or sediment concentrations.

Based on the available toxicity and exposure data, EEB has determined that the use of synthetic pyrethroids on cotton and sunflowers poses a significant risk to aquatic organisms. The Special Review criteria in 40 CFR Part 154.7(a)(3) of the regulations have been met because the EEC >1/2 LC_{50} and >1/20 LC_{50} (Urban and Cook, 1986). Consequently, there is a high risk of adverse effects to nonendangered and endangered aquatic species, respectively. One-half and 1/20th of the LC_{50} values are used to incorporate margins of safety.

The lambda cyhalothrin mesocosm (U.S. EPA, 1989) demonstrated that fish were more sensitive under field conditions than in the laboratory. The NOEL in the fish full-life-cycle study was 31 ng/L. This concentration is 52 times greater than the mesocosm concentrations where adverse effects to fish occurred (0.6–1 ng/L). It should be noted that bluegill were used in the mesocosm and fathead minnows were tested in this bioassay. Both species are used as indicator species; therefore, the toxicity levels can be compared.

Risk to Ecological Components of the Terrestrial Habitats. The terrestrial EECs for both cotton and sunflower uses are not expected to approach acute toxicity values for avian or mammalian species; therefore, the risk concern for direct acute effects to these species is low. Most of the synthetic pyrethroids have adverse effects on eggshell integrity and other reproductive impairments, but toxicity to avian species is not expected because the terrestrial EECs are lower than chronic effect levels. To date, field studies have not addressed adverse effects of synthetic pyrethroids on waterfowl. Nevertheless, EEB has concluded that the use of synthetic pyrethroids on sunflowers may pose a significant risk to breeding waterfowl and hatchlings because of their reliance on aquatic prey organisms.

Table 10-5. EECs or Actual Measured Residues Exceeding Toxicity Concentrations (Cotton Use Pattern)

Chemical	Lowest Acute LC_{50} (ng/L)	Lowest Chronic NOEL (ng/L)	Estimated EEC from SWRRB/ Exams (ng/L)	Measured Residues (ng/L)	Estimated EEC Exceeds LC_{50}	Estimated EEC Exceeds NOEL
Cyfluthrin	2.4 (40069501, 1987)[a]	0.17 (262443, 1988)	500 (water) (189625, 1987)	-	208x	3,000x
Tralomethrin	38 (11507-74-03, 1981)	46 (41860701, 1991)	50 (water) (194619, 1987)	-	1x	1x
Bifenthrin	3.9 (264647, 1987)	1.3 (41156501, 1989)	4 (water) 17,000 ng/kg (sediment) (279-EUP-RNR, 1984)	2 (water) 10,000 - 60,000 ng/kg (sediment) (40981801, 1991)	1x (water)	3x (water)

[a] EPA identification number, year of EEB review.

Risk to the Waterfowl Prey Base. Ninety percent of the sunflower growing region is in North Dakota, South Dakota, and Minnesota. These states are in the prairie pothole region, which is responsible for more than 50 percent of the annual duck production in North America (Smith et al., 1964). During the reproduction period, adults and ducklings are highly dependent on aquatic macroinvertebrates (Krapu and Swanson, 1975; Sheehan et al., 1987). Sheehan et al. (1987) also state that "inadequate nutrition adversely affects reproduction" and that calcium and protein requirements come from aquatic invertebrates that supply 2 to 4 times the protein levels of plants. It has been shown that laying hens consumed more than 70 percent animal food and that some species are totally dependent on invertebrates (Sheehan et al., 1987).

Essential amino acids must be in the proper ratios for protein formation. Krapu and Swanson (1975) state that when pintail hens feed on several invertebrate prey items, they are balancing their amino acid needs. Joyner (1980) investigated seven morphological and biological variables that might affect pond selection by ducks. It was concluded that only invertebrate numbers and taxa present were correlated with duck usage. This information shows the importance of prairie potholes containing large invertebrate populations with adequate species diversity.

Laying hens are opportunistic feeders and will switch prey when there is a seasonal change in species abundance. Laying efforts generally terminate, however, when the abundance of selected prey items starts to decline. Females also terminate renesting attempts when prey availability is reduced (Sheehan et al., 1987). Renesting attempts are very important, especially in the pothole regions because of the high destruction rate of nests from agricultural operations (Krapu and Swanson, 1975).

Evidence indicates that ducklings have a high dependence on animal food, primarily invertebrates, for periods of 1 to 7 weeks post-hatch. A diet of greater than 50 percent animal protein is required to sustain a growth rate in mallard ducklings. Growth rate is correlated with fat deposit, plumage development (both important in temperature control), strength, and activity. Therefore, duckling growth is roughly equivalent to survival, and fast growth increases their likelihood of survival (Sheehan et al., 1987).

The National Research Council of Canada (1986) has stated that the survival of young birds that feed in ponds adjacent to pyrethroid-treated agricultural lands may be affected by the timing of application. Fenvalerate is conditionally registered for use on sunflowers and can be expected to be applied in the prairie pothole region during the stages of waterfowl reproduction that rely most heavily on aquatic invertebrates and emerging insects. These stages of reproduction are the last half of the duck egg-laying phase and the entire hatching and early duckling phase.

Risk to Bees and Plant Pollination. Sunflower production used to be almost entirely dependent on bees and other insects for pollination. Most new sunflower hybrids have been selected to self-pollinate without pollinator activity. But some hybrid plants such as the F1 type still require over 20 bees per 100 heads in bloom (McMullen, 1985). Synthetic pyrethroids are generally highly toxic to honey bees under laboratory conditions. But some of them have repellent properties that make exposure less likely in the field.

Sources of Uncertainty. There are uncertainties in characterizing risks because variations and assumptions have been made in the exposure and toxicity assessments. Environmental fate data were not available to be used in determining the terrestrial EEC from multiple applications, but a method was presented in which foliar half-life was estimated. The aquatic EEC equation contains assumed values for application efficiency and percentage drift. Both the SWRRB and EXAMS II models contain variations that are inherent in chemical fate data because such data are usually expressed as a range and not a fixed number. These models also contain hydrologic variations due to regional climatic, topographical, soil type, and soil texture differences. It is assumed that the limited number of test species used in toxicity testing represent all wildlife species. Adverse effects to waterfowl were based not on direct field testing but on indirect evidence obtained from the literature and a measured decrease in their prey base (supported by field studies). Even though there are uncertainties in the risk characterization process, EEB has been able to accurately predict when certain chemicals are expected to cause adverse ecological effects. As in the case with synthetic pyrethroids, both the mesocosm study and pond study have confirmed EEB's presumption of risk.

Comments on Risk Characterization

Strengths of the case study include:

- *The case study presented for synthetic pyrethroids is a good example of the process used by EPA for the risk assessment of pesticides. Its several strengths are given below.*
  - *—The case study illustrates how various types of information are used and integrated; these include laboratory acute and chronic toxicity data, estimated environmental concentration of pesticides from models, and field studies on actual or simulated aquatic ecosystems. The field studies were used to confirm the presumption of risk indicated by the toxicity data and exposure assessment.*
  - *—The selection of synthetic pyrethroids was excellent because these chemicals are extremely toxic to aquatic invertebrates and fish, persistent in the environment, and bioaccumulated.*
  - *—The case study illustrates the sensitivity of biological systems to chemical concentrations that are below the level of detection.*
  - *—Because the case study concentrated on conditionally registered pesticides, the study illustrated how the Agency deals with data gaps in the risk assessment process.*
  - *—This case study illustrated a range of potential ecological effects for different use patterns, cotton and sunflowers, that would likely affect two different types of aquatic habitats, ponds and prairie potholes.*
  - *—Given the state of the art in ecological risk assessment, this study illustrated that both quantitative and qualitative data are useful in making professional judgments.*

Comments on Risk Characterization (continued)

- *It has been shown that aquatic invertebrates are an essential food source for breeding waterfowl hens and hatchlings. Therefore, an indirect qualitative relationship between synthetic pyrethroids and waterfowl can be drawn. If synthetic pyrethroid use severely reduces or eliminates aquatic invertebrate populations, then adverse effects to waterfowl may occur.*

Limitations include:

- *The assessment could have benefited from a better integration of field and laboratory data.*

- *No rationale was provided for using the LC_{50} in quotient method. The technical basis for using 1/2 and 1/20 the LC_{50} to trigger other laboratory and field studies was not described, although a literature citation is given.*

- *The relationship between pesticide-induced aquatic invertebrate population reductions and adverse effects to waterfowl requires additional documentation.*

General comment:

- *The process for risk assessment creates myriad problems for the registrant because of the length of time it takes to provide the data for the risk assessment and the rapid changes in methods for measuring ecological effects and predicting exposure. The registrant must use state-of-the-art methods to generate data to be used in the risk assessment several years later. During this time, the process must be open to receive new information, especially information not required currently to characterize potential ecological effects.*

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APPENDIX A

TIER TESTING SYSTEM FOR ECOLOGICAL EFFECTS

TIER TESTING SYSTEM FOR ECOLOGICAL EFFECTS

TIER I

Terrestrial:
- Mammalian toxicity data
- Acute oral LD_{50} test—bird
- Dietary LC_{50} test—bird
- Seed germination/seedling emergence and vegetative vigor
- Acute contact LD_{50} test—honey bee

Aquatic:
- 96-hour LC_{50} test—coldwater fish
- 96-hour LC_{50} test—warmwater fish
- 48-hour (or 96-hour) LC_{50}/EC_{50} test—freshwater aquatic invertebrate
- 96-hour EC_{50} test—algae

TIER II

Terrestrial:
- Wild mammal toxicity data
- Avian reproductive studies
- Special studies with avian or mammalian species (e.g., avian cholinesterase test, secondary toxicity)
- Honey bee residue on foliage
- Seed germination/seedling emergence and vegetative vigor

Aquatic:
- 96-hour LC_{50} test—estuarine/marine fish
- 96-hour LC_{50} test—estuarine/marine crustacean
- 48-hour EC_{50} test—bivalve embryo-larvae or
  96-hour EC_{50} test—bivalve shell deposition
- Fish early-life-stage MATC or Effect/No Effect Level
- Aquatic invertebrate life cycle MATC or Effect/No Effect Level
- Fish bioaccumulation factor, e.g, 1000X
- Special aquatic organism test data (e.g., fish acetylcholinesterase levels)
- Aquatic plant growth testing

TIERS III and IV

Terrestrial:
- Simulated and actual field testing with avian and/or mammalian species
- Field test for pollinators

Aquatic:
- Fish full-life-cycle MATC or Effect/No Effect Level (Tier III)
- Field testing for aquatic organisms (Tier IV) (e.g., mesocosm or pond study)

SECTION ELEVEN

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

TOXIC DISCHARGES TO SURFACE WATERS: ASSESSING THE RISK TO AQUATIC LIFE USING NATIONAL AND SITE-SPECIFIC WATER QUALITY CRITERIA

AUTHORS AND REVIEWERS

AUTHORS

Robert L. Spehar
Environmental Research Laboratory-Duluth
U.S. Environmental Protection Agency
Duluth, MN

Anthony R. Carlson
Environmental Research Laboratory-Duluth
U.S. Environmental Protection Agency
Duluth, MN

REVIEWERS

Anthony F. Maciorowski
(Lead Reviewer)
Fish Culture and Ecology Laboratory
U.S. Fish and Wildlife Service
Kearneysville, WV

Robert Brooks
School of Forest Resources
Pennsylvania State University
University Park, PA

G. Allen Burton
Biological Sciences Department
Wright State University
Dayton, OH

Rick D. Cardwell
Parametrix, Inc.
Bellevue, WA

Michael J. Dover
The Cadmus Group, Inc.
Peterborough, NH

Joseph Makarewicz
Department of Biological Sciences
State University of New York
at Brockport
Brockport, NY

Thomas P. O'Connor
National Status and Trends Program
National Oceanic and Atmospheric
Administration
Rockville, MD

Jerry R. Schubel
Marine Sciences Research Center
State University of New York
at Stony Brook
Stony Brook, NY

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LIST OF ACRONYMS

CCC	criterion continuous concentration
CMC	criterion maximum concentration
EC_{50}	effective concentration for 50 percent of organisms tested
EPA	U.S. Environmental Protection Agency
FAV	final acute value
FCV	final chronic value
FPV	final plant value
FRV	final residue value
LA	load allocations
LC_{50}	lethal concentration for 50 percent of organisms tested
NOEL	no observed effects level
NPDES	National Pollutant Discharge Elimination System
WLA	wasteload allocations
WQC	water quality criteria

ABSTRACT

Guidelines recommended by the U.S. Environmental Protection Agency (EPA) for deriving national and site-specific water quality criteria for the protection of aquatic life were evaluated in a case study of the St. Louis River basin in Duluth, Minnesota. The recalculation, indicator species, and resident species procedures of the site-specific approach were used to modify the national cadmium criteria to site-specific criteria. The procedures accounted for differences in species sensitivity and the biological availability and/or toxicity of cadmium due to physical and/or chemical characteristics of site water. The national and site-specific criteria approaches were used as a tool to assess the risk of ambient metal exposure to resident aquatic life.

These guidelines provide an approach to applying national water quality criteria on a site-specific basis to reflect local environmental conditions. Because the national criteria are designed to protect the biological integrity of all water bodies, these criteria serve as benchmarks and may require adjustments for site-specific applications (e.g., risk assessments). Consideration of local conditions assures that criteria for a particular body of water are tailored specifically to its aquatic life and uses.

A major strength of the site-specific guidelines is that they are based on national guidelines that have undergone extensive scientific review to assure their general applicability. However, the water quality criteria guidelines constitute only one approach to assessing the risk of pollutants on aquatic systems. For comprehensive, ecologically based risk assessments, this approach should be used in conjunction with other EPA procedures, such as the whole-effluent approach used for dealing with mixtures of chemicals, as well as procedures for developing sediment, wildlife, and biological criteria.

11.1. RISK ASSESSMENT APPROACH

This case study does not represent a complete risk assessment as defined in the *Framework for Ecological Risk Assessment* (figure 11-1). Although it provides useful information on the stressor and its anticipated ecological effects, this case study does not characterize the exposure where all stressors to the St. Louis River Estuary are considered. Instead, cadmium was measured from a limited monitoring program at this site so that this metal could be used as an example of a stressor. To conduct a complete risk assessment, a significant amount of information, as defined by the objectives of any case study, would need to be gathered to identify and quantify all stressors for a site (Stephan et al., 1985).

The following case study is based on an earlier study by Spehar and Carlson (1984a, b) that was designed to demonstrate how to use the U.S. Environmental Protection Agency's (EPA's) site-specific approach for modifying national water quality criteria. The current case study was developed as an example by reorganizing information from the earlier study into the present risk assessment format. The results show how the principles of deriving site-specific criteria can be used as a tool for making ecological risk assessments.

The procedures of the national and site-specific guidelines (Carlson et al., 1984; Stephan et al., 1985), along with their stated assumptions and limitations, need to be understood before the current case study will have meaning in such assessments. These procedures constitute only one approach—the chemical-specific approach—that EPA uses in its water quality-based program for limiting toxins in surface waters. The chemical-specific approach focuses on the protection of aquatic life and does not completely address other issues involving wildlife or terrestrial communities. To conduct a comprehensive risk assessment of a particular site, the chemical-specific approach should be used in combination with the whole-effluent approach (for characterizing mixtures) and EPA's procedures for deriving sediment, wildlife, and biological criteria.

11.2. STATUTORY AND REGULATORY BACKGROUND

The Clean Water Act of 1977 (Section 304[a][1]) requires EPA to review and publish water quality criteria necessary to protect public water supplies and to safeguard the propagation of shellfish, fish, and wildlife. The criteria provide scientific data and guidance on the environmental effects of pollutants and can be used to derive water quality-based regulatory requirements such as effluent limitations, water quality standards, or toxic pollutant effluent standards (U.S. EPA, 1980a).

National water quality criteria have been developed by applying a set of guidelines (Stephan et al., 1985) to data for certain pollutants designated as toxic under Section 307(a)(1) of the Clean Water Act of 1977 pursuant to an agreement in the case of *Natural Resources Defense Council et al.* v. *Train*, 1976. According to these guidelines, water quality criteria should be based on an array of data for the types of plant and animal species that occupy various trophic levels. Based on these data, criteria can be derived that should adequately protect the types of species necessary to support an aquatic community.

Figure 11-1. Structure of Analysis for Assessing Risk Using National and Site-Specific Criteria

PROBLEM FORMULATION

Stressors: cadmium; other stressors interact with this metal but were not considered.

Ecological Components: invertebrates and fish species.

Endpoints: assessment endpoint is protection of aquatic community structure functions and integrity. Measurement endpoint includes single species toxicity tests.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
A limited number of measurements were made of cadmium concentrations in river water.	Data supporting the national criterion for cadmium and new toxicity tests using St. Louis River water were used to establish site-specific criteria for cadmium.

RISK CHARACTERIZATION

The limited data on cadmium concentrations in the St. Louis River were compared to the national and site-specific criteria. The limitations of the approach are described.

Although the criteria represent a reasonable estimate of pollutant concentrations consistent with the maintenance of designated uses, each state may modify these values to reflect local conditions. Because national criteria may be either underprotective or overprotective, EPA has developed guidelines (U.S. EPA, 1983; Carlson et al., 1984) to adapt national water quality criteria to local conditions (i.e., site-specific criteria). Most national criteria (see appendix A) are based on information obtained from toxicity and bioconcentration tests conducted in the laboratory. Some criteria, however, as stated in the national guidelines, can be based on assessments of adequate field data from actual sites. These assessments include information on aquatic life (residue content) that may be useful for protecting wildlife populations.

In other cases, toxicological information obtained on laboratory-tested aquatic species may not be applicable to species in specific water bodies. The species at a particular site may be more or less sensitive than those included in the national criteria data base, or the physical and/or chemical characteristics of the water at the site may alter the biological availability and/or toxicity of the material.

This case study uses laboratory data and EPA's site-specific approach as a tool for assessing the risk posed by cadmium to the St. Louis River Estuary.

11.3. CASE STUDY DESCRIPTION

11.3.1. Problem Formulation

Site Description. The St. Louis River system, located primarily in southern St. Louis County, Minnesota, encompasses approximately 1,400 miles of streams. The St. Louis River and its tributaries account for 815 of these 1,400 miles (Spehar and Carlson, 1984a). The mouth of the St. Louis River is a freshwater estuary containing approximately 11,500 acres of water. It has been developed into a major industrial port that serves as the economic base for the cities of Duluth, Minnesota, and Superior, Wisconsin. The site chosen for a source of dilution water in this study was located approximately 34 miles upstream from the mouth of the Duluth-Superior estuary, at the State Highway 33 crossing in the city of Cloquet, Minnesota.

The St. Louis River Estuary was chosen as the specific site for the case study because enough information on the biological and chemical characteristics of this site was known. In addition, sufficient information was available to meet the requirements for conducting a risk assessment by using the components of the risk assessment framework. For example, a limited water quality monitoring data set (Hammermeister et al., 1983) was available and resident aquatic species for the site were previously characterized and documented (Spehar and Carlson, 1984a) for use in defining ecological components.

Stressors. Cadmium was used as an example to demonstrate EPA's site-specific approaches to risk assessment (Spehar and Carlson, 1984a, b). Because this was a sample exercise, other stressors that were known to exist at the site, such as different metals, organic chemicals, and conventional pollutants (as single chemicals or as mixtures in the water column and/or sediment), were not considered for use in this case study. Cadmium was chosen in the earlier publication because it is known to be highly toxic to aquatic organisms.

Cadmium is commonly found in the environment in treated municipal wastes (U.S. EPA, 1980b). In addition, its chemistry in water may be influenced by changes in water quality (Geisy et al., 1977; Calamari et al., 1980; Reid and McDuffie, 1981), which would be a major consideration in modifying the present national criteria for use in this case study. Detailed toxicological characterizations were conducted according to EPA's guidelines for deriving national and site-specific criteria (Carlson et al., 1984; Stephan et al., 1985).

Ecological Components. Although the components in this case study are the resident species of the St. Louis River Estuary (Spehar and Carlson, 1984a), they are represented in this case study by the array of species defined by the minimum data sets in the national and site-specific guidelines (Stephan et al., 1985; Carlson et al., 1984, respectively). The specific components used in this representation are delineated in the procedures described in the characterization of ecological effects section of this case study.

An assessment of the biota in the St. Louis River was conducted according to the procedures delineated below for modifying the national criteria for cadmium at this site. The approach assumes that protection of 95 percent of the tested species according to EPA's guidelines (Carlson et al., 1984; Stephan et al., 1985) will adequately protect community structure, function, and integrity.

Endpoints. The assessment endpoint is protection of aquatic community structure, function, and integrity. Measurement endpoints included the acute and chronic toxicity test results used in this exercise to represent the array of resident species at this site, as well as those delineated by the guidelines for deriving both the national and the site-specific criteria.

Comments on Problem Formulation

Strengths of the case study include:

- *The causal relationship for cadmium exposure and aquatic life toxicity is well documented.*

- *Site-specific water quality criteria may be better suited than the national criteria for cadmium.*

- *The procedures are designed to derive site-specific water quality criteria by allowing substantial flexibility in the method used. This flexibility should permit regulatory agencies to choose the most appropriate and efficient means of obtaining the information needed to modify national criteria for each particular site for use in risk assessments. Site-specific water quality criteria for cadmium and the St. Louis River Estuary obtained from the site-specific guidelines appear to be logical because they take into account the national cadmium criteria and the physical, chemical, and biological characteristics of water at this site.*

Comments on Problem Formulation (continued)

Limitations include:

- *The endpoints, although derived in a rigorous and standardized manner, do not relate directly to meaningful ecological endpoints that can be readily measured in the field. Water quality criteria guidelines encourage the use of field studies for deriving criteria; however, field studies are usually not available for most chemicals. Consequently, in most cases, laboratory data are used. Although criteria derived from single species responses from laboratory tests have been validated with ecologically based studies (U.S. EPA, 1989), it is difficult to demonstrate that criteria derived in this manner do in fact produce the desired results at the ecosystem level.*

- *The water quality-based approach assumes that the aquatic ecosystem is at risk when water quality criteria are exceeded, but does not delineate the actual species, populations, communities, or other system components at risk.*

- *The case history is limited to the toxicological risk of cadmium without considering other contaminants or ecosystem stressors.*

11.3.2. Analysis: Characterization of Ecological Effects

Background. Two types of stress-response assessments were used for this case study. The first derived ambient freshwater aquatic life water quality criteria for cadmium (U.S. EPA, 1985) and was completed before the present case study according to the national guidelines (Stephan et al., 1985). The second assessment derived freshwater site-specific water quality criteria for cadmium in the St. Louis River (Spehar and Carlson, 1984a, b) as outlined in the following procedures.

EPA criteria consist of three parameters (magnitude, frequency, and duration) that are developed for two levels of effect (acute and chronic). Uncertainties associated with the derivation of EPA national and site-specific criteria are not detailed here; however, validation studies are available (U.S. EPA, 1983) that support the use of these approaches until more meaningful ecological endpoints can be measured readily on an ecosystem basis.

The following is an abbreviated list of endpoints and definitions that will be helpful in understanding the present procedures for deriving site-specific criteria for cadmium. A more complete list can be found in the national and site-specific guidelines for deriving water quality criteria (U.S. EPA, 1983; Stephan et al., 1985).

- Acute value: A 48- to 96-h LC_{50} or EC_{50}, depending on the species.

- Chronic value: The geometric mean of the lower chronic limit (highest tested concentration in an acceptable chronic test that did not cause significant decreases from the control) and the upper chronic limit (lowest tested concentration in an acceptable chronic test that caused significant decreases from the control).

- Acute/chronic ratio: The ratio of an acute value for a species to a comparable chronic value for that species tested in the same water.

- Final acute value (FAV): An estimate of the concentration of a material corresponding to a cumulative probability of 0.05 in the acute toxicity values for the genera with which acute tests have been conducted for that material. (For an exception and the effects of water quality characteristics on this value, see Stephan et al., 1985.)

- Final chronic value (FCV): An estimate of the concentration of a material corresponding to a cumulative probability of 0.05 in the chronic toxicity values for genera with which chronic tests have been conducted for that material, usually obtained by dividing the FAV by the final acute/chronic ratio. (For an exception and the effects of water quality characteristics on this value, see Stephan et al., 1985.)

- Final acute/chronic ratio: The geometric mean of all the species' mean acute/chronic ratios available for both freshwater and saltwater species. (For variations in the calculation of this value, see Stephan et al., 1985.)

- Final residue value (FRV): The lowest of the residue values obtained by dividing the maximum permissible tissue concentrations by the appropriate bioconcentration or bioaccumulation factors.

- Criterion maximum concentration (CMC): A criterion value that is equal to one-half of the FAV.

- Final plant value (FPV): The lowest result from a test with an important aquatic plant species in which the concentrations of the test material were measured and the endpoint was biologically important.

- Criterion continuous concentration (CCC): The criterion value that is the lowest of the FCV, FRV, and FPV, unless other data from field studies or laboratory tests show that a lower value should be used.

National Criteria for Cadmium. With the possible exception of a locally important but highly sensitive species, freshwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration (CCC in μg/L) of cadmium does not exceed the numerical value given by $e^{(0.7852[\ln(\text{hardness})]-3.490)}$ more than once every 3 years on average and if the 1-hour average concentration (CMC in μg/L) does not exceed the numerical value given by $e^{(1.128[\ln(\text{hardness})]-3.828)}$ more than once every 3 years on average. For example, at hardness values of 50, 100, and 200 mg/L as $CaCO_3$, the 4-day average concentrations of cadmium are 0.66, 1.1, and 2.0 μg/L, respectively, and the 1-hour average concentrations are 1.8, 3.9, and 8.6 μg/L, respectively (U.S. EPA, 1985).

Site-Specific Criteria. Numerous assumptions are associated with the site-specific guidelines, most of which also apply to and are included in the national guidelines. A few assumptions need to be emphasized. The principal assumption is that the species sensitivity ranking and toxicological effect endpoints (e.g., survival, growth, and reproduction) derived from laboratory tests will be similar to those derived from site situations. Another assumption is that the protection of all of the site species all of the time is not necessary because aquatic life can tolerate some stress and occasional adverse effects.

It is also assumed that the site-specific guidelines are an attempt to improve the protection of the various uses of aquatic life by accounting for toxicological differences in species sensitivity or the biological availability and/or toxicity of a material at specific sites. Modification of the data set must always be scientifically justifiable and consistent with the assumptions, rationale, and spirit of the national guidelines.

In these types of assessments, EPA criteria are developed (using toxicological data) as national recommendations to assist states in developing water quality standards. Standards are then adopted by the states to designate uses and to define ambient characteristics of receiving waters. These standards must be maintained to allow those uses and must be met before wastewaters can be discharged legally.

Procedures for calculating site-specific criteria. Three procedures from the site-specific guidelines (U.S. EPA, 1983; Carlson et al., 1984) were used in this study to modify the national CMC and CCC for cadmium (see the rationale for the site-specific guidelines listed in these publications). All procedures were conducted by using the pertinent species, exposures, and calculations that would be needed to modify the national criteria. This characterization was made even though all three procedures would not necessarily be used in an actual site modification.

The three procedures used to calculate the site-specific criteria for cadmium in the St. Louis River Estuary were as follows: (1) the *recalculation procedure*, to account for differences in cadmium sensitivity between species resident in the St. Louis River Estuary and those species contained in the national cadmium criteria document (U.S. EPA, 1985); (2) the *indicator species procedure*, to account for differences in the biological availability and/or toxicity of cadmium due to physical and/or chemical characteristics of the St. Louis River water and laboratory water by deriving a water-effect ratio (toxicity in site water divided by the toxicity in laboratory water); and (3) the *resident species procedure*, to account simultaneously for differences in both resident species sensitivity and differences that may be attributed to water quality. Acute lethality was the only endpoint considered in the recalculation procedure; the indicator species and resident species procedures utilized test data on lethality, growth, and reproductive potential.

Procedures to determine a final residue value or a final plant value (which are required in the national guidelines for deriving water quality criteria) are not used in this case study because they are not sensitive endpoints for cadmium. Cadmium is not lipid-soluble and will not biomagnify in aquatic systems and affect higher food chain organisms such as wildlife. Plants are not as sensitive as aquatic animals to cadmium and would not be affected at criteria levels based on animal tests. Thus, the criteria concentrations calculated in this case study are based on the most sensitive FAVs and FCVs, which were determined from toxicity tests with aquatic animals.

A detailed description of how to define a site; the rationale, assumptions, and limitations of the site-specific procedures; and the relationship of site-specific procedures to those used for deriving national water quality criteria are included in the site-specific guidelines (U.S. EPA, 1983; Carlson et al., 1984).

Recalculation procedure. The recalculation procedure modifies the national CMC for cadmium (U.S. EPA, 1985) by eliminating data for nonresident species from the national data base.

The data set for resident species in the St. Louis River was sufficient to meet the minimum data set requirements of the national guidelines (Stephan et al., 1985) (see table 11-1). Thus, no additional acute tests in laboratory water were needed. The site-specific FAV for this procedure and cadmium was 1.4 μg/L for the data set when calculated by using the procedure described in the national guidelines (Stephan et al., 1985). The CMC was derived by the following equation: site-specific CMC = site-specific FAV/2.

The value obtained from this equation is 0.7 μg/L. However, because the toxicity of cadmium has been related to the hardness of the water, this relationship was taken into account by using the method described in the national guidelines to adjust for hardness before the site-specific CMC was calculated (table 11-1). The site-specific CMC for cadmium from the preceding equation, adjusted for the hardness of the St. Louis River, was 0.8 μg/L when derived by using the recalculation procedure. (A hardness value of 55 mg/L as $CaCO_3$ was used as an example for this exercise and was determined from the lowest hardness measured monthly in this water over a year; a more comprehensive monitoring data program would probably be needed to determine the most appropriate value for this site on a seasonal basis.)

The recalculation procedure does not require testing to determine a site-specific CCC. A site-specific FCV can be derived by dividing the site-specific FAV by the national final acute/chronic ratio; if the national final acute/chronic ratio was not used to calculate the national FCV, the national FCV then becomes the site-specific FCV. A national final acute/chronic ratio for cadmium was not derived because enough chronic values were available to calculate an FCV directly (U.S. EPA, 1985). Therefore, the site-specific FCV determined by using the recalculation procedure is the same as the national FCV adjusted for the hardness of the St. Louis River, or 0.7 μg/L.

Indicator species procedure. The indicator species procedure is based on the determination of a water effect ratio to account for the differences in the toxicity of cadmium in the St. Louis River and in laboratory water due to physical and/or chemical characteristics of these waters. Tests with two sensitive species, one fish and one invertebrate species, were required for this procedure. A cladoceran (*Simocephalus serrulatus*) and rainbow trout (*Oncorhynchus mykiss*) were selected as indicator species. Tests for each species were conducted in both types of water under

Table 11-1. Recalculation Procedure: Acute Toxicity Data for Cadmium From National Criteria Document for Resident Species of the St. Louis River (modified from Spehar and Carlson, 1984a)[a]

Rank	Genus Mean Acute Value (μg/L)	Species	Species Mean Acute Value (μg/L)
19	8,325	Goldfish (*Carassius auratus*)	8,325
18	5,708	Channel catfish (*Ictalurus punctatus*)	5,708
17	3,800	Snail (*Amnicola sp.*)	3,800
16	3,641	Green sunfish (*Lepomis cyanellus*)	5,147
		Pumpkinseed (*Lepomis gibbosus*)	1,347
		Bluegill (*Lepomis macrochirus*)	6,961
15	3,514	White sucker (*Catostomus commersoni*)	3,514
14	2,310	Mayfly (*Ephemerella grandis*)	2,310
13	1,200	Midge (*Chironomus sp.*)	1,200
12	322.8	Mayfly (*Paraleptophlebia praepedita*)	322.8
11	215.5	Common carp (*Cyprinus carpio*)	215.5
10	204.9	Amphipod (*Hyalella azteca*)	204.9
9	156.9	Snail (*Physa gyrina*)	156.9
8	104.0	Snail (*Aplexa hypnorum*)	104.0
7	83.02	Cladoceran (*Ceriodaphnia reticulata*)	83.02
6	62.55	Amphipod (*Gammarus pseudolimnaeus*)	55.90
		Amphipod (*Gammarus sp.*)	70.00
5	55.72	Cladoceran (*Daphnia pulex*)	55.72
4	43.74	Cladoceran (*Simocephalus serrulatus*)	45.93
		Cladoceran (*Simocephalus vetulus*)	41.65
3	30.5	Fathead minnow (*Pimephales promelas*)	30.50
2	4.481	Coho salmon (*Oncorhynchus kisutch*)	5.894
		Chinook salmon (*Oncorhynchus tshawytscha*)	4.254
		Rainbow trout (*Oncorhynchus mykiss*)	3.589
1	1.638	Brown trout (*Salmo trutta*)	1.638

[a]Site-specific final acute value (calculated for a hardness of 50 mg/L from genus mean acute values) = 1.416 μg/L.
Site-specific criterion maximum concentration = (1.416 μg/L)/2 = 0.708 μg/L (for a hardness of 50 mg/L).
ln (site-specific criterion maximum intercept) = ln (0.708) - [slope x ln (50)] = - 4.758.
Site-specific criterion maximum concentration = e [1.128 (ln hardness) - 4.758] = 0.788 μg/L adjusted for a hardness of 55 mg/L.

similar test conditions. A water-effect ratio was calculated by using the following equation: water-effect ratio = site water LC_{50}/lab water LC_{50}.

Measured LC_{50} values for a toxicant must be significantly different (U.S. EPA, 1983; Carlson et al., 1984) in the two waters for this procedure to be valid. If the values are not different, then the national CMC becomes the site-specific CMC.

The 96-h LC_{50} values for cladocerans and rainbow trout were statistically different in site water and laboratory water (table 11-2), and their water-effect ratios were similar. Consequently, they could be used to calculate a site-specific CMC in the following equation: site-specific CMC (7.4 μg/L) = geometric mean water-effect ratio (4.6) × national CMC (1.6 μg/L). The national CMC was adjusted for the hardness of the laboratory water (45 mg/L as $CaCO_3$) before the site-specific CMC was calculated.

The site-specific CCC for the indicator species procedure can be derived from three optional methods (U.S. EPA, 1983; Carlson et al., 1984):

- by calculating (no testing required) the national acute/chronic ratio (if one is present) and applying it to the site-specific FAV; if the national acute/chronic ratio was not used to establish a national FCV, the national FCV may be used as the site-specific FCV;

- by performing two acute and two chronic tests with both a fish and an invertebrate species in site water and applying the resulting acute/chronic ratio to the site-specific FAV;

- by conducting chronic tests with both a fish and an invertebrate species in both site water and laboratory water and by applying the chronic water-effect ratio to the national FCV.

When derived by using the first method, the site-specific CCC from the indicator species procedure was 0.7 μg/L, or the same as the national CCC. A final acute/chronic ratio was not used to establish the national FCV, so the site-specific FCV is the same as the national value. (Note that the FCVs and CCCs for both the national and site-specific values are the same in this example because the FRVs and FPVs were not calculated in this exercise [see explanation above]; therefore, the FCVs become the CCCs for both procedures.)

The site-specific CCC calculated by using the second method was 0.3 μg/L, based on a geometric mean acute/chronic ratio of 50 (from tests performed in the St. Louis River water [table 11-3]) and the following equation: site-specific FCV = site-specific FAV/(site-specific final acute/chronic ratio, or 14.8/50 = 0.3 μg/L). The site-specific chronic value was obtained by using a site-specific FAV of 14.8 (twice the site-specific CMC obtained from the indicator species procedure).

Table 11-2. Indicator Species Procedure: Acute Values (LC_{50}) for Indicator Species Exposed to Cadmium in St. Louis River and Reconstituted Water (modified from Spehar and Carlson, 1984a)

Organism	St. Louis River Water (μg/L)	Reconstituted Water (μg/L)	Water-Effect Ratio[a]
Cladoceran (*Simocephalus serrulatus*)	123	24.5	5.0
Rainbow trout (*Oncorhynchus mykiss*)	10.2	2.3	4.4

[a]Geometric mean water-effect ratio = 4.6.

Table 11-3. Acute (LC_{50}) and Chronic Values for Aquatic Organisms Exposed to Cadmium in St. Louis River Water (Spehar and Carlson, 1984a)

Organism	Acute Value (μg/L)	Chronic Value (μg/L)	Acute/ Chronic Ratio[a]
Fathead minnow (*Pimephales promelas*)	1,830	18.9	97
Cladoceran (*Ceriodaphnia reticulata*)	129	5.0	26

[a]Geometric mean acute/chronic ratio = 50.

The site-specific CCC calculated by using the third method (the chronic water-effect ratio), based on studies for two species (fathead minnow and cladoceran), was determined to be 1.0 because the chronic values obtained from tests in site and laboratory water were not significantly different (the chronic limits overlapped) (U.S. EPA, 1983) (table 11-4). Since the mean chronic water-effect ratio was not different from 1.0, the site-specific FCV is the same as the national FCV adjusted for the hardness of the St. Louis River, or 0.7 $\mu g/L$, using this method. Although tests were not conducted specifically to obtain a site-specific chronic value by using this third method, comparisons were made between present chronic tests in St. Louis River water and tests conducted at different times with the same species in Lake Superior water for use as an example of a chronic water-effect ratio. According to the site-specific guidelines (U.S. EPA, 1983; Carlson et al., 1984), tests in both waters should be run at the same time with organisms from the same population and under the same test conditions.

Resident species procedure. The resident species procedure allows for modification of the national criteria for cadmium on the basis of tests conducted in site water with a set of resident species of the St. Louis River (table 11-5). Because the minimum data set requirements for resident species were met at the St. Louis River site, substitute families were not needed. (Note: A family in a phylum other than Arthropoda or Chordata [e.g., Rotifera, Annelida, Molluska, etc.] was not included in this data set because it was not a requirement of the national guidelines at the time these tests were conducted.) The site-specific FAV calculated by using the prescribed method for resident species (Stephan et al., 1985) was 3.8 $\mu g/L$. The resident species site-specific CMC was calculated as follows: site-specific CMC = site-specific FAV/2, or 3.8/2 = 1.9 $\mu g/L$.

The site-specific CCC of the resident species procedure was obtained by using the first two methods described under the indicator species procedure, based on a site-specific FAV of 3.8 $\mu g/L$. The site-specific FCVs for these methods were 0.7 and 0.1 $\mu g/L$, respectively. The third method should not be used to calculate a site-specific FCV using the resident species procedure (U.S. EPA, 1983).

Summary of criteria calculation. Comparison of cadmium water quality criteria derived from the national and site-specific procedures showed that criteria values varied according to the procedure used (table 11-6). Site-specific criteria derived from the recalculation procedure were similar or slightly lower than those of the national criteria. The lower acute site-specific criterion was derived using a smaller number of genera to calculate the site-specific FAVs than was used to calculate the national criterion (19 versus 44).

Site-specific criteria for cadmium derived from the indicator species procedure were higher on an acute basis (CMC) than those derived from the national and recalculation procedures. This result was expected because criteria derived from the indicator species procedure were based on a water-effect ratio attributed to site water characteristics that decreased the toxicity of cadmium. Cadmium was found to be less toxic to several species in St. Louis River water than in laboratory water. On a chronic basis (CCC), site-specific criteria using this method were the same as or slightly lower than the national criteria. The lower value was due to a large acute/chronic ratio (50, see table 11-3) measured for two species in tests conducted in site water.

Table 11-4. Chronic Toxicity Values of Two Species Exposed to Cadmium in St. Louis River and Lake Superior Water (Spehar and Carlson, 1984a)

	Chronic Value (μg/L)		
Organism	St. Louis River Water	Lake Superior Water	Chronic Water-Effect Ratio[a]
Fathead minnow (*Pimephales promelas*)	18.9 (13–26)[b]	13[c] (9–18)	1.0
Cladoceran (*Ceriodaphnia reticulata*)	5.0 (3.4–7.2)	5.2[d] (3.6–7.5)	1.0

[a]Chronic water-effect ratios are 1.0 because values in site and laboratory are not different (chronic limits overlap).
[b]Chronic limits.
[c]Data from Carlson et al. (unpublished manuscript).
[d]Data from D.I. Mount (unpublished manuscript).

Table 11-5. Resident Species Procedure: Minimum Data Set of Resident Aquatic Species Exposed to Cadmium in St. Louis River Water (Spehar and Carlson, 1984a)[a]

Rank	Organism	LC_{50} (μg/L)
8	Bluegill (*Lepomis macrochirus*)	8,800
7	Channel catfish (*Ictalurus punctatus*)	7,900
6	Fathead minnow (*Pimephales promelas*)	3,390
5	Mayfly (*Paraleptophlebia praepedita*)	449
4	Amphipod (*Hyalella azteca*)	285
3	Cladoceran (*Simocephalus serrulatus*)	123
2	Amphipod (*Gammarus pseudolimnaeus*)	54
1	Rainbow trout (*Oncorhynchus mykiss*)	10

[a]Site-specific final acute value for this resident species data set = 3.8.

Table 11-6. **Cadmium Water Quality Criteria Derived From National and Site-Specific Procedures (modified from Spehar and Carlson, 1984a)**

Criterion Derivation Procedure	Criterion Maximum Concentration (µg/L)	Criterion Continuous Concentration (µg/L)
National	2.0[a]	0.7[a]
Site-specific		
recalculation	0.8[a]	0.7
indicator	7.4[b,c]	0.7(a)[d] 0.3(b) 0.7(c)
Resident	1.9	0.7(a) 0.1(b)

[a]Adjusted for a water hardness of the St. Louis River of 55 mg/L as $CaCO_3$.
[b]The national data base containing 44 genera was used in this calculation.
[c]A national criterion of 1.6 µg Cd/L (adjusted for a hardness of 45 mg/L hardness [as $CaCO_3$]) was used in this procedure.
[d]Letters in parentheses indicate optional methods used for calculating the site-specific criterion continuous concentration.

The large acute/chronic ratio indicates that chronic toxicity was not greatly affected by water quality at this site. Chronic values were similar to those from tests with similar species that were conducted in laboratory water (U.S. EPA, 1985). However, acute values were higher than previously reported, indicating that the water quality characteristics of this site decreased toxicity on a short-term basis, probably through mechanisms affecting the bioavailability of this chemical.

The site-specific CMC derived from the resident species procedure was approximately two times higher than the criterion derived from the recalculation procedure, but was lower than that obtained from the indicator species procedure. This result also was expected and was attributed to a combination of the use of a limited data base of eight species (which resulted in a lower criterion) and the use of site water tests (which raised the criterion due to the mitigating effects of the site water). In addition, the site-specific CCC derived from the chronic water-effect ratio method (the third method under the indicator species procedure) was the same as the national FCV (adjusted for the hardness of the laboratory water) because the mean chronic water-effect ratio was not significantly different from 1.0. The low CCC obtained from the second method for both the indicator and resident species procedures was attributed to the large site-specific acute/chronic ratio (50) used in the calculation of the site-specific FCV.

Although all of the procedures from the site-specific guidelines (U.S. EPA, 1983) were tested, only one would probably be used in an actual site criteria modification. If species sensitivity was the important factor (for example, where species at a particular site are more or less sensitive than those used to derive the national criteria), then the recalculation procedure would be the recommended approach because it would require no testing. The indicator species procedure would be appropriate when water quality at a site may mitigate the toxicity of a chemical and the resident species are similar to those used to calculate the national criteria. This is especially true for metals like cadmium, for which biological availability and/or toxicity are significantly affected by variations in water quality characteristics of the site water. When both species sensitivity and water quality are important considerations for a particular site, the resident species procedure would be the best approach because it is designed to account for differences due to both factors.

Recommended Site-Specific Procedure. For the St. Louis River and Estuary, the indicator species procedure would be the recommended approach for deriving site-specific criteria for cadmium. The present study showed that cadmium was less toxic in site water than in laboratory water, which resulted in a water-effect ratio. This ratio was designed to account for water quality effects, and its use with the national CMC should provide a site-specific CMC that would adequately protect the resident aquatic species. The recalculation procedure would not be appropriate in this case because there was no real difference between the sensitivity range of species represented in the national data set and species found at this site. This rationale would also apply to the resident species procedure, which in part accounts for species sensitivity. The resident species procedure could be used for this site, but additional testing in site water would be required.

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *This case study illustrates how a site-specific tool (water quality criteria) can be developed for evaluating potential effects.*

Limitations include:

- *Acute toxicity of cadmium to 25 aquatic species indigenous to the St. Louis River is ranked for the recalculation procedure. Rankings are based on sensitivity in laboratory studies rather than on relative importance in the river system. For the recalculation procedure, lethality is the only test endpoint.*

- *The hardness values used for the test site require a larger data base.*

- *The stress-response assessment is limited to toxicological effects of site water on selected surrogate species, with no consideration of higher dimensional ecological phenomena.*

11.3.3. Analysis: Characterization of Exposure

A study by Hammermeister et al. (1983) provided monitoring data for cadmium from a relatively clean water upstream sampling station and a downstream sampling station below major municipal and industrial wastewater discharges into the St. Louis River Estuary. This study was used as an example to provide an exposure concentration for use in this exercise. A total of six grab samples per station was taken over a 6-month period beginning on January 29, 1983, and ending on June 23, 1983. Cadmium was found to be present at 1.0 μg/L, a concentration that slightly exceeded both the national and site-specific CCCs (table 11-6). Cadmium concentrations above these criteria were measured in only one of the downstream samples.

The averages of monthly acute cadmium toxicity values for larval fathead minnows exposed in St. Louis River water (from the present procedures) were approximately five and two times higher than values obtained from concurrent cadmium exposures conducted in reconstituted and Lake Superior water, respectively, according to studies by Spehar and Carlson (1984a, b). These studies reported similar toxicity differences in four of five juvenile species exposed to St. Louis River water. The findings indicate that physical and/or chemical characteristics of the St. Louis River water reduced the toxicity of cadmium on an acute basis from what was observed in laboratory water.

The acute cadmium toxicity values calculated from the above larval fish exposures in river water varied by a factor of three and increased with increases in the concentration of suspended solids, total organic carbon, turbidity, and dissolved solids. Linear regression correlation coefficients for acute toxicity were calculated, and these parameters were 0.58, 0.60, 0.68, and 0.77, respectively. The LC_{50} values for tests conducted in reconstituted and Lake Superior water varied by a factor of less than 2. The larger variation in values obtained from tests conducted in site water was attributed to high and low stream flows, which influenced water quality factors throughout the year. The large degree of binding or complexing of cadmium that occurred during times when concentrations of particulates in this water were highest was the apparent cause of reduced cadmium toxicity. Although this effect on acute toxicity was not large in the present tests, larger variations in toxicity may occur in streams where particulate loads change significantly during different times of the year. The frequency of testing required to perform risk assessments by using the site-specific approach will depend on this seasonal variability.

The effect of seasonality on the physical and chemical characteristics of water and subsequent effects on biological availability and/or toxicity of cadmium justify the use of seasonally dependent site-specific criteria for the St. Louis River Estuary. A major implication of seasonally dependent criteria is whether the most sensitive time of the year coincides with the time when the flow is the basis for waste treatment facilities design or National Pollutant Discharge Elimination System (NPDES) permits. That is, if the physical and chemical characteristics of the water during low-flow seasons increase the biological availability and/or toxicity of cadmium, the permit limitations may be more restrictive than if the converse relationship were to apply.

The national and site-specific water quality criteria concentrations stated above contain duration (averaging period) and frequency (or average recurrence interval) periods (to account for excesses) that are based on biological, ecological, and toxicological data and are designed to protect aquatic organisms and their uses from unacceptable effects (Stephan et. al., 1985). Numeric criteria are used as the basis for determining wasteload allocations (WLAs) for point sources of contaminants or load allocations (LAs) for nonpoint sources (U.S. EPA, 1991). Limits on wastewater loads are set and nonpoint source allocations are established so that receiving water concentrations (i.e., for cadmium in the St. Louis River Estuary) do not exceed water quality criteria. Information on conducting exposure and wasteload allocation procedures are detailed in EPA's *Technical Support Document for Water Quality-Based Toxics Control* (U.S. EPA, 1991).

Comments on Analysis: Characterization of Exposure

Limitations of the case study include:

- *The case study does not focus on exposure.*

- *The exposure assessment includes the magnitude, duration, and frequency of exposure limits, but includes no implicit discussion of probable exposure to populations in the St. Louis River.*

11.3.4. Risk Characterization

The indicator species procedure was selected as the appropriate approach to use in conducting a risk assessment for this site. This procedure takes into account factors that would affect the bioavailability and/or the toxicity of the identified stressor (cadmium). However, because the indicator species procedure is used as an example, the actual data have limited practical use. To fully characterize risks to ecological components at this particular site, more information would be needed on exposure to all stressors and the specific components (i.e., aquatic, terrestrial, and other species such as wildlife) that are of concern within the scope of the study. The water quality criteria (including site-specific) approach is only intended to protect aquatic life and their uses (but simultaneously may protect wildlife to some degree) and is only one approach used by EPA to assess aquatic effects. This approach needs to be used in combination with the whole-effluent approach used for assessing the impact of mixtures of chemicals and with other approaches used to develop sediment, wildlife, and biological criteria.

The ambient concentration of 1.0 μg/L observed for cadmium in the St. Louis River Estuary was slightly above the site-specific values based on chronic tests (CCC), but was generally less than the site-specific criteria values based on acute tests (CMC). The site-specific CMC, based on the indicator species procedure, was higher than the national criteria because the site water decreased cadmium toxicity by decreasing its bioavailability to the resident species in the estuary. Because the ambient concentration of cadmium at the site was lower than the site-specific CMC, it would appear that cadmium would not pose a risk to the resident aquatic species on a short-term basis. However, because ambient concentrations are above site-specific criteria based on chronic tests, some sensitive resident species may be affected on a long-term basis. Additional risk characterization techniques would be needed to determine wasteloads from point sources or loadings from nonpoint sources to see if cadmium, mixtures of chemicals that include cadmium, or other stressors to the system do indeed cause in-stream toxicity when criteria are exceeded. WLA and LA models are available, as noted above, to conduct such characterizations. After these types of analyses are conducted, specific toxin limits can be defined by using models to predict design flows that should prevent ambient exposures from exceeding criteria levels for longer durations or at more frequent intervals than allowed.

Although some laboratory microcosm/mesocosm tests and field studies support the approaches of this case study, additional research is recommended to revise and improve the guidelines that form the underpinnings of the criteria derivation process. Recommendations for making these revisions and for proposed research to improve EPA's guidelines are discussed in U.S. EPA (1989). A refinement of these suggested revisions is currently being made by EPA as recommended by a workshop held in December 1990 to revise the National Water Quality Criteria Guidelines.

Comments on Risk Characterization

Strengths of the case study include:

- *Water quality criteria (WQC) are valuable impact assessment tools that can be considered for risk assessments when used with appropriate hypotheses and exposure scenarios. Although the case study is not an example of a risk assessment, the development of a benchmark (criterion) can be used in a quotient approach to risk assessment.*

- *This case study provides a good example of an ecological effects stressor-response assessment.*

Limitations include:

- *Sole reliance on water quality criteria compliance might be misleading. One cannot assume that ecosystem protection is "acceptable" if WQC are met, due to interactions or unknown exposures. Ecological field data may be necessary to verify or validate toxicological phenomena. Biosurveys are important to bridge the gap between the laboratory and field environments.*

- *A site-specific criterion maximum concentration of 7.4 μg/L is justified by the study data, but the relative risk and uncertainty are characterized rhetorically rather than quantitatively.*

- *A toxicity-based approach to water quality assumes ecological risk to populations and communities, but relies solely on organismic endpoints that do not address all ecological concerns. Without some type of community assessment, ecological risk and damage may be improperly assessed.*

General comments:

- *The development of ecological (i.e., biological) criteria in parallel with chemical-specific criteria would help validate predictions and generate biological endpoints that can be measured in the field.*

- *Exposure-response curves, rather than a single endpoint such as the no observed effects level (NOEL), could be used and an acceptable response level on the curve selected to set criteria.*

- *The effects of water quality characteristics (e.g., pH, temperature, organic content) on the bioavailability of pollutants, particularly metals, are only partially understood. Similarly, predictions of toxicity and persistence can be inaccurate. Therefore, laboratory techniques are needed to better simulate ambient conditions and field methods are needed to evaluate the predictions.*

Comments on Risk Characterization (continued)

- *Duration and frequency criteria should be based on effects that occur as a consequence of exposures similar to actual excess levels rather than worst-case scenarios. Better information is needed on actual "time-till-effect" relationships for specific pollutants and recovery times for ecosystems responding to moderate as well as catastrophic effects.*

- *Better techniques are needed for integrating pollutant-specific assessments. Mixtures of pollutants may interact in complex ways, or they may simply exhibit the effects of the predominant constituent(s). The tools available (i.e., whole-effluent approach) can be used to evaluate these interactions better.*

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APPENDIX A

WATER QUALITY CRITERIA FOR AQUATIC LIFE

APPENDIX A

Water Quality Criteria for Aquatic Life

available from the
National Technical Information Service (NTIS)
5285 Port Royal Road
Springfield, VA 22161
(703-487-4650)

Note: Multiple entries are given for those pollutants for which corrections and/or revised criteria have been published.

Pollutant	Federal Register Notice[a]	EPA Number	NTIS Number
Acenaphthene	1	EPA 440/5-80-015	PB81-117269
Acrolein	1	EPA 440/5-80-016	PB81-117277
Acrylonitrile	1	EPA 440/5-80-017	PB81-117285
Aldrin/Dieldrin	1	EPA 440/5-80-019	PB81-117301
Ammonia	4	EPA 440/5-85-001	PB85-227114
Antimony	1	EPA 440/5-80-020	PB81-117319
Arsenic	1	EPA 440/5-80-021	PB81-117327
	4	EPA 440/5-84-033	PB85-227445
Asbestos	1	EPA 440/5-80-022	PB81-117335
Benzene	1	EPA 440/5-80-018	PB81-117293
Benzidine	1	EPA 440/5-80-023	PB81-117343
Beryllium	1	EPA 440/5-80-024	PB81-117350
Cadmium	1	EPA 440/5-80-025	PB81-117368
	4	EPA 440/5-84-032	PB85-227031

Pollutant	Federal Register Notice[a]	EPA Number	NTIS Number
Acenaphthene	1	EPA 440/5-80-015	PB81-117269
Acrolein	1	EPA 440/5-80-016	PB81-117277
Carbon tetrachloride	1	EPA 440/5-80-026	PB81-117376
Chlordane	1	EPA 440/5-80-027	PB81-117384
Chlorinated benzenes	1	EPA 440/5-80-028	PB81-117392
Chlorinated ethanes	1	EPA 440/5-80-029	PB81-117400
Chloroalkyl ethers	1	EPA 440/5-80-030	PB81-117418
Chlorinated naphthalene	1	EPA 440/5-80-031	PB81-117426
Chlorinated phenols	1	EPA 440/5-80-032	PB81-117434
Chlorine	4	EPA 440/5-84-030	PB85-227429
Chloroform	1	EPA 440/5-80-033	PB81-117442
2-Chlorophenol	1	EPA 440/5-80-034	PB81-117459
Chlorpyrifos	6	EPA 440/5-86-005	PB87-105267
Chromium	1	EPA 440/5-80-035	PB81-117467
	4	EPA 440/5-84-029	PB85-227478
Copper	1	EPA 440/5-80-036	PB81-117475
	4	EPA 440/5-84-031	PB85-227023
Cyanide	1	EPA 440/5-80-037	PB81-117483
		EPA 440/5-84-028	PB85-227460
DDT	1	EPA 440/5-80-038	PB81-117491
Dichlorobenzenes	1	EPA 440/5-80-039	PB81-117509
Dichlorobenzidine	1	EPA 440/5-80-040	PB81-117517

Pollutant	Federal Register Notice[a]	EPA Number	NTIS Number
Acenaphthene	1	EPA 440/5-80-015	PB81-117269
Acrolein	1	EPA 440/5-80-016	PB81-117277
Dichloroethylenes	1	EPA 440/5-80-041	PB81-117525
2,4-Dichlorophenol	1	EPA 440/5-80-042	PB81-117533
Dichloropropanes/Dichloropropenes	1	EPA 440/5-80-043	PB81-117541
2,4-Dimethylphenol	1	EPA 440/5-80-044	PB81-117588
Dinitrotoluene	1	EPA 440/5-80-045	PB81-117566
Diphenylhydrazine	1	EPA 440/5-80-062	PB81-117731
Dissolved oxygen	5	EPA 440/5-86-003	PB86-208253
Endosulfan	1	EPA 440/5-80-046	PB81-117574
Endrin	1	EPA 440/5-80-047	PB81-117582
Ethylbenzene	1	EPA 440/5-80-048	PB81-117590
Fluoranthene	1	EPA 440/5-80-049	PB81-117608
Haloethers	1	EPA 440/5-80-050	PB81-117616
Halomethanes	1	EPA 440/5-80-051	PB81-117624
Heptachlor	1	EPA 440/5-80-052	PB81-117632
Hexachloroobutadiene	1	EPA 440/5-80-053	PB81-117640
Hexachlorocyclohexane	1	EPA 440/5-80-054	PB81-117657
Hexachlorocyclopentadiene	1	EPA 400/5-80-055	PB81-117655
Isophorone	1	EPA 440/5-80-056	PB81-117673

Pollutant	Federal Register Notice[a]	EPA Number	NTIS Number
Acenaphthene	1	EPA 440/5-80-015	PB81-117269
Acrolein	1	EPA 440/5-80-016	PB81-117277
Lead	1	EPA 440/5-80-057	PB81-117681
	4	EPA 440/5-84-027	PB85-227437
Mercury	1	EPA 440/5-80-058	PB81-117699
	2	Correction	--
	4	EPA 440/5-84-026	PB85-227452
Naphthalene	1	EPA 440/5-80-059	PB81-117707
Nickel	1	EPA 440/5-80-060	PB81-117715
	6	EPA 440/5-86-004	PB87-105359
Nitrobenzene	1	EPA 440/5-80-061	PB81-117723
Nitrophenols	1	EPA 440/5-80-063	PB81-117749
Nitrosamines	1	EPA 440/5-80-064	PB81-117756
Parathion	6	EPA 440/5-86-007	PB87-105383
Pentachlorophenol	1	EPA 440/5-80-065	PB81-117764
	6	EPA 440/5-86-009	PB87-105391
Phenol	1	EPA 440/5-80-066	PB81-117772
Phthalate esters	1	EPA 440/5-80-067	PB81-117780
Polychlorinated biphenyls (PCBs)	1	EPA 440/5-80-068	PB81-117798
Polynuclear aromatic hydrocarbons	1	EPA 440/5-80-069	PB81-117806
Selenium	1	EPA 440/5-80-070	PB81-117814
Silver	1	EPA 440/5-80-071	PB81-117822

Pollutant	Federal Register Notice[a]	EPA Number	NTIS Number
Acenaphthene	1	EPA 440/5-80-015	PB81-117269
Acrolein	1	EPA 440/5-80-016	PB81-117277
Tetrachloroethylene	1	EPA 440/5-80-073	PB81-117830
2,3,7,8-Tetrachlorodibenzo-p-dioxin	3	EPA 440/5-84-007	--
Thallium	1	EPA 440/5-80-074	PB81-117848
Toluene	1	EPA 440/5-80-075	PB81-17855
Toxaphene	1	EPA 440/5-80-076	PB81-117863
	6	EPA 440/5-86-006	PB87-105375
Trichloroethylene	1	EPA 440/5-80-077	PB81-117871
Vinyl chloride	1	EPA 440/5-80-078	PB81-117889
Zinc	1	EPA 440/5-80-079	PB81-117897
	7	EPA 440/5-87-003	PB87-153581
Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses.	4	--	PB85-227049

[a]Federal Register Notices:

1. Federal Register, Vol. 45, pp. 79318-79379, November 28, 1980
2. Federal Register, Vol. 46, pp. 40919, August 13, 1981
3. Federal Register, Vol. 49, pp. 5831-5832, February 15, 1984
4. Federal Register, Vol. 50, pp. 30784-30796, July 29, 1985
5. Federal Register, Vol. 51, pp. 22978, June 24, 1986
6. Federal Register, Vol. 51, pp. 43665-43667, December 3, 1986
7. Federal Register, Vol. 52, pp. 6213, March 2, 1987

SECTION TWELVE

ECOLOGICAL RISK ASSESSMENT CASE STUDY:

MODELING FUTURE LOSSES OF BOTTOMLAND FOREST WETLANDS AND CHANGES IN WILDLIFE HABITAT WITHIN A LOUISIANA BASIN

AUTHORS AND REVIEWERS

AUTHORS

Michael S. Brody
Office of Policy, Planning, and Evaluation
U.S. Environmental Protection Agency
Washington, DC

Michael E. Troyer
Office of Technology Transfer and Regulatory Support
U.S. Environmental Protection Agency
Washington, DC

Yvonne Valette
Region 6
U.S. Environmental Protection Agency
Dallas, TX

REVIEWERS

Clifford Hupp (Lead Reviewer)
Water Resources Division
U.S. Geological Survey
Reston, VA

Joel Brown
Department of Biological Sciences
University of Illinois
Chicago, IL

Arthur L. Buikema
Biology Department
Virginia Polytechnic Institute and State University
Blacksburg, VA

Carolyn L. Fordham
Terra Technologies
Golden, CO

Christian E. Grue
Washington Cooperative Fish and Wildlife Research Unit
U.S. Fish and Wildlife Service
Seattle, WA

Ronald J. Kendall
Institute of Wildlife and Environmental Toxicology
Clemson University
Clemson, SC

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LIST OF ACRONYMS

ATP	adenosine triphosphate
COE	U.S. Army Corps of Engineers
DBH	diameter at breast height
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
HEP	Habitat Evaluation Procedure
HSI	Habitat Suitability Index
HU	Habitat Units
MSL	mean sea level
NEPA	National Environmental Policy Act of 1969
NOAA	National Oceanic and Atmospheric Administration
RCRA	Resource Conservation and Recovery Act
SAB	Science Advisory Board
SI	Suitability Indices
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey

ABSTRACT

The Lake Verret Basin, a part of the Atchafalaya River floodplain in southern Louisiana, is composed largely of bottomland hardwoods and cypress-tupelo swamps. These forested wetlands tolerate variable flood durations and provide high-quality habitat (e.g., food and shelter) for many wildlife species. However, if hydrologic changes such as long-term excess flooding become too great, these habitats can become nonforested marshes or open-water areas less suitable for wildlife species. Given the Administration's policy of "no net loss of wetlands" and the difficulty of replacing forested wetlands, any future destruction or degradation of these habitats should be avoided. Artificial levees first erected in the Atchafalaya River floodplain in the 1930s have deprived the Lake Verret Basin of sediment deposition, contributing to a net subsidence rate of 25 cm every 50 years. Subsequently, increased backwater flood heights and durations have caused the bottomland forest to succeed toward cypress-tupelo communities and have reduced opportunities for lower-elevation cypress-tupelo to regenerate. In this case study, an analysis is conducted on the ongoing processes in the Lake Verret Basin. Although construction of more levees has been proposed, this analysis predicts future conditions given current subsidence problems from past levee construction. Thus, this analysis serves as a set of baseline conditions with which to compare the changes in drainage and water flows that future levee projects would cause. This case study is based on work by Brody et al. (1989) and Conner and Brody (1989) to add the effects of subsidence into FORFLO, a bottomland forest succession model, and to forecast temporal and spatial forest community impacts in swamp areas and wet and dry bottomlands within the Lake Verret Basin. Model outputs of FORFLO are coupled with Habitat Suitability Index models to determine present and future wildlife habitat values for two species of birds and three species of mammals.

12.1. RISK ASSESSMENT APPROACH

This case study includes all major components of an ecological risk assessment (figure 12-1). Information originally provided by Brody et al. (1989) and Conner and Brody (1989) on the incorporation of FORFLO and Habitat Suitability Index (HSI) models allowed synthesis of previously collected impact assessment data into a format useful for assessing the probability of ecological risk as it pertains to modifications in hydrology and subsequent changes in habitat. Effects of other anthropogenic stressors (e.g., chemical pollutants) are not considered.

Because of the interaction of environmental factors in influencing forested wetland biology and succession, the FORFLO bottomland forest succession model (Pearlstine et al., 1985; Brody and Pendleton, 1987; Brody et al., 1989) was used for predicting the future plant communities in the Lake Verret Basin. The succession model outputs of FORFLO were used as input values to HSI models for assessing future value of the habitat to wildlife. FORFLO predicts tree species presence and abundance, individual tree size, canopy closure, flood duration, and other habitat measures that affect wildlife. It provides these outputs on an annual basis for as many years as required for the impact analysis. As such, a mechanism is available to quantify a field assessment of the current habitat values (HSI models), to quantify future habitat conditions (FORFLO), and to assess future habitat values (HSI models).

12.2. STATUTORY AND REGULATORY BACKGROUND

The National Environmental Policy Act of 1969 (NEPA) establishes a national environmental policy and goals for the protection, maintenance, and enhancement of the environment and provides a process for implementing these goals within federal agencies, including the U.S. Environmental Protection Agency (EPA). The NEPA process consists of an assessment of the environmental effects of federal projects including all alternatives and impacts. Such assessments (and management decisions) appear to be increasingly risk based since the risk assessment approach can help identify the relative efficiency and effectiveness of different risk reduction options (Stakhiv, 1986; Russell and Gruber, 1987).

Under the Clean Water Act, EPA is responsible for restoring and maintaining the physical, chemical, and biological integrity of the nation's waters, including wetlands. Although no comprehensive wetlands management program exists, Section 404 of the Act provides the primary legislative authority behind federal efforts to protect wetland use; however, the level of protection is limited to the regulation of "discharges" of dredged or fill materials into waters of the United States. Wetlands continue to be altered in many other ways, including excavation, draining, clearing, flooding, and other water diversions. It is estimated that Section 404 does not cover about 80 percent of the nation's wetland losses (Office of Technology Assessment, 1984). Currently, it appears that the scope of federal wetlands protection will change as a result of technical and policy revisions to the 1989 Federal Manual for Identifying and Delineating Jurisdictional Wetlands and Congress' reauthorization of the Clean Water Act. Whether these changes will enhance or degrade America's wetland resources is a subject of current debate.

The U.S. Fish and Wildlife Service (USFWS) developed HSI models (USFWS, 1981) to define a habitat-based approach for assessing the environmental impacts of federal water projects

Figure 12-1. Structure of Analysis for Modeling Losses of Bottomland Forest Wetlands

PROBLEM FORMULATION

Stressors: changes in water level elevations due to subsidence and other factors.

Ecological Components: tree species in bottomland habitat and five wildlife species (three mammals; two birds)

Endpoints: assessment endpoint was physical alteration or change in the forest community and associated habitat value. Measurement endpoints included the vegetation, hydrologic, and other input data required for the FORFLO and HSI models.

ANALYSIS

Characterization of Exposure	Characterization of Ecological Effects
A baseline change in water elevation was estimated from other studies for regional subsidence rates.	The FORFLO model was used to estimate changes in forest vegetation; the HSI models were used to relate these changes to effects on habitat for five species.

RISK CHARACTERIZATION

Changes (risks) to forest vegetation were estimated under a baseline exposure regime represented by regional subsidence.

Changes (risks) to wildlife were estimated for the future conditions forecast for forest vegetation.

The uncertainties associated with the use of the FORFLO and HSI models were described.

on fish and wildlife resources (i.e., for Environmental Impact Statements [EISs] required by NEPA and other reports required by the Fish and Wildlife Coordination Act). To evaluate the effects of hydrologic changes on forested wetland habitats, the USFWS developed a bottomland forest succession model, FORFLO (Pearlstine et al., 1985), that simulates the growth, reproduction, and competition of a mixed-tree-species forest stand.

In 1987, at the request of the EPA Administrator, a National Wetlands Policy Forum convened to suggest ways to improve wetland regulation and management. In its final report, *Protecting America's Wetlands*, the Forum recommended "...no overall net loss of the nation's remaining wetlands base, as defined by acreage and function." This policy has been endorsed by EPA. Furthermore, several of the Forum's recommendations have reappeared in EPA testimony to Congress during Clean Water Act reauthorization hearings in 1991. At present, EPA lacks risk assessment and management approaches for considering physical habitat alteration and biological diversity. The Science Advisory Board's (SAB's) report *Reducing Risk: Setting Priorities and Strategies for Environmental Protection* (U.S. EPA, 1990a) states that wetlands have "extraordinary value" and that the alteration and destruction of natural habitats (including wetlands) pose a high risk to the natural ecology and human welfare of very large areas. The SAB also considers the related issue of species extinction and overall loss of biological diversity (including genetic diversity) as a high-risk problem. As a result, the SAB has called on EPA managers to direct their efforts toward reducing risks posed by these and other "critical environmental problems" (U.S. EPA, 1990b).

12.3. CASE STUDY DESCRIPTION

12.3.1. Problem Formulation

Site Description. The Lake Verret Basin, a part of the Atchafalaya floodplain in south-central Louisiana, is bounded by the East Atchafalaya Basin Protection Levee on the west, the natural levee ridges of the Mississippi River and Bayou LaFourche on the east, Bayou Plaquemine to the north, and Louisiana Highway 20 and U.S. Highway 90 to the south (see figure 12-2). The watershed occupies approximately 99,000 ha, of which 48 percent are seasonally flooded bottomland hardwood areas and cypress-tupelo swamps (Soil Conservation Service, 1978). The basin lies entirely within the Mississippi River Deltaic Plain, and the land is low and flat with elevations ranging from 1 m mean sea level (MSL) in the northern portion of the basin to less than 1 m MSL in the southern portion.

Stressors. Nature and human activities are recognized as major contributors to declining wetland resources. In Louisiana, naturally occurring wetlands loss results from subsidence, rising sea level, normal wave action, pounding storm surges, and saltwater intrusion into freshwater areas. Human causes of wetlands loss include levee construction along the Mississippi and Atchafalaya Rivers, dredging and soil disposal, drainage, mineral extraction, wave action from vessel traffic, and agricultural, urban, and industrial expansion. An important factor contributing to the risk of deteriorating wetlands along coastal areas is the alteration of hydrologic conditions for flood control or navigation.

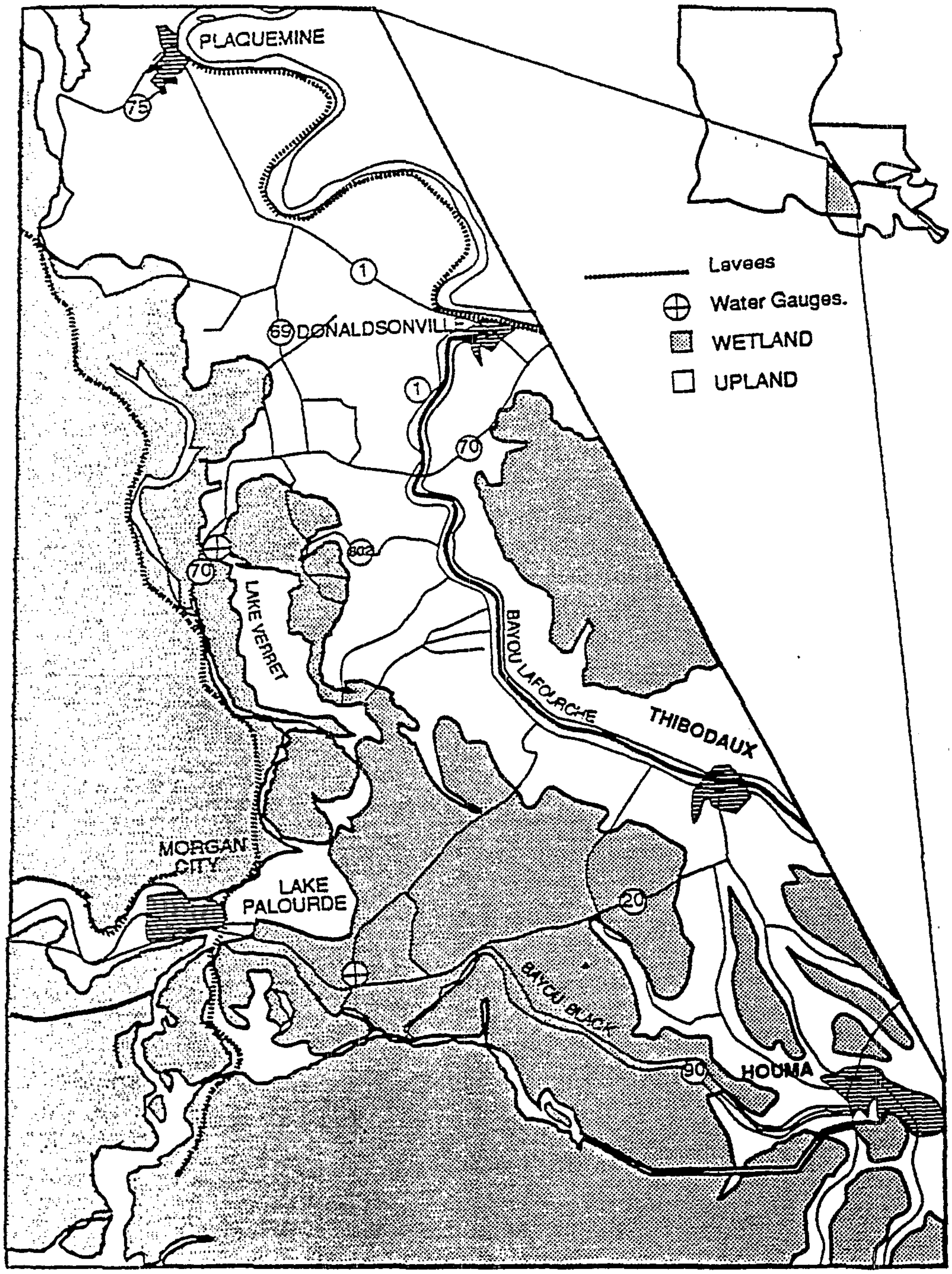

Figure 12-2. Location of Lake Verret Basin (Conner et al., 1986)

The subsidence that occurs in Louisiana can be divided into two general categories: tectonic subsidence and consolidation/compaction. Tectonic subsidence refers to the large-scale downward geologic displacement caused by sedimentary loading and associated settlement processes of deltaic formations. The consolidation/compaction aspect of subsidence is attributed to a variety of causes including overlying weight (levees, spoil mounds); subsurface withdrawal (oil and gas exploration); and dewatering (drainage and reclamation projects). Because of the difficulty in separating the effects of subsidence and sea level rise during any analysis of relative changes between land and water levels, the two factors are frequently identified by researchers as "relative sea level rise." Tidal gauges along the coast of Louisiana indicate that the rate of relative sea level rise is 9 to 13 mm per year (3 to 4 ft per century) (Slater, 1986).

Sedimentary processes are responsive to changes in hydrologic and biologic processes, and the rates of sediment accretion affect the ability of plants to adapt to the direct and indirect effects of relative and absolute variations in water level (Cahoon and Turner, 1986). Sediment input and organic accumulation counteract compaction and contribute to land accretion, but the supply and distribution of sediments are not static in recent times. According to Meade and Parker (1984), suspended sediments in the Mississippi River apparently have declined by more than 50 percent since the early 1950s.

Prior to the construction of artificial flood-control levees, the Verret Basin was part of the Mississippi-Atchafalaya River floodplain, and under natural conditions, floodwaters would bring sediment-laden waters into these forested areas and the swamp forests would be replaced by bottomland hardwood forests. However, artificial levees, erected for flood protection along the east side of the Atchafalaya floodplain, have deprived the basin of seasonal overbank flooding and sediment deposition, contributing to an approximate net subsidence rate of 25 cm over 50 years (Slater, 1986).[1] Water levels in the basin have since been influenced mainly by backwater flooding and subsidence and secondarily by rainfall and upland runoff. Additionally, the Atchafalaya River is lengthening its course due to the active deposition of sediments at its mouth and the subsequent development of its delta. The river gradient (slope) is steadily decreasing and has resulted in higher water levels at the mouth. Rising water levels in the lower part of the Atchafalaya Basin reduce the hydraulic gradient of drainage from the upper part of the Verret watershed, further increasing water levels (Boesch et al., 1983).

In this case study, an analysis is conducted on the ongoing processes in the Lake Verret Basin. This analysis predicts future conditions, given current subsidence problems from past levee construction. The primary hazard of concern in this case study is the rate and magnitude of water level changes over time (i.e., hydroperiod) and the resultant changes on forest community species and dynamics (i.e., habitat alteration). The analysis incorporates the three habitat types found

[1]In reality, the net subsidence rate probably varies within some range across the Atchafalaya Basin. Slater (1986) provided the best known estimate of net subsidence in the basin at the time of this study. The reader should note that predictions made by FORFLO and other models depend on the most accurate environmental data available as inputs. As such, emphasis should be placed on obtaining actual field data instead of using "default values" that may or may not be representative of the study site.

within the basin (drier bottomland hardwoods, wetter hardwoods, and cypress-tupelo swamps) with their current value to five wildlife species, their probable fates due to basin-wide hydrologic changes, and their most probable value to these same wildlife species in 50 years.

Ecological Components. As with most (if not all) assessments of the natural environment, everything cannot be measured. Instead, one must decide on a subset of parameters that, within a given timeframe and budget, will likely provide useful data for answering a scientific or regulatory question. If the objectives of the study are to assess the impacts of one or more stressors on an entire ecological community, then an ecologically based approach is desirable. Selection of "components" (or species to be evaluated) may be based on the approach of choosing a suite of representative species to provide an ecological perspective of the study area. Ideally, representative species should be sensitive to specific land-use actions, serve as indicators for a large segment of the wildlife community, and represent groups of species that use a common environmental resource (e.g., representative species for various trophic guilds).

Fauna present at the study site were not surveyed directly. Instead, the wildlife species or "components" used in this study were chosen from a list of common biota known to occupy the Atchafalaya Basin by an interagency evaluation team composed of representatives of the USFWS, U.S. Army Corps of Engineers (COE), National Marine and Fisheries Service, Louisiana Department of Wildlife and Fisheries, Louisiana Department of Natural Resources, and EPA (Gerry Bodin, USFWS, Lafayette, LA, personal communication). Wildlife components were selected to represent (i.e., act as surrogates or status indicators of) the various guilds of wildlife using ground cover, forest canopy, and water resources in the basin for feeding or reproduction. Species also were chosen for their commercial, recreational, or social importance, and their sensitivity to hydrological impacts (i.e., candidate species were screened for those variables found in nature and the HSI models that would be sensitive to hydrologic changes in the basin). The following five wildlife indicator species representative of habitat value in forested wetlands were chosen for the HSI models: gray squirrel, *Sciurus carolinensis* (Allen, 1982); swamp rabbit, *Sylvilagus aquaticus* (Allen, 1985); mink, *Mustela vison* (Allen, 1983); wintering wood duck, *Aix sponsa* (Sousa and Farmer, 1983); and downy woodpecker, *Picoides pubescens* (Schroeder, 1983). The selection matrix used to evaluate and choose these species is shown in table 12-1.

Detailed biotic assessments of the study site focused on surveying dominant canopy tree and wildlife cover vegetation in the wet bottomland hardwood, dry bottomland hardwood, and swamp areas. Tree species encountered in all three areas are presented in table 12-2.

Endpoints. The primary assessment endpoint was spatial and temporal change in the forest community and the associated habitat value of the forested wetlands. Measurement endpoints included the data input requirements for the FORFLO and HSI models, as described below.

FORFLO Model Data Inputs. Data needs for FORFLO may be divided into the categories of vegetation, hydrologic data, and other site data. These categories describe "ideal" data for input for each FORFLO application. Shortcuts can be taken occasionally, and extrapolations can be made with useful results still possible (Brody and Pendleton, 1987).

Table 12-1. Sample Selection Matrix for Wildlife Species

Species	Socioeconomic Value	Cover Types: Ground	Canopy	Water	Indicator for
Gray squirrel	Recreational		X		Mast production
Swamp rabbit		X			Utilization of ground surface
Downy woodpecker	Social		X	X	Snag density and canopy cover
Mink	Commercial	X		X	Edge effect between ground and water surface
Wood duck	Recreational and social		X	X	Wintering habitat and mast production

Table 12-2. Tree Species Observed in Three Simulated Sites (Brody et al., 1989)

Species	Dry Bottomland	Wet Bottomland	Swamp
Acer rubrum (red maple)	X	X	
Carya aquatica (water hickory)	X	X	
Celtis laevigata (sugarberry)	X		
Diospyros virginiana (persimmon)		X	
Fraxinus pennsylvanica (green ash)		X	X
Gleditsia triacanthos (honey locust)		X	
Liquidambar styraciflua (sweetgum)	X	X	
Nyssa aquatica (water tupelo)			X
Populus deltoides (cottonwood)		X	
Quercus lyrata (overcup oak)	X		
Quercus nigra (water oak)	X		
Quercus nuttallii (nuttall oak)	X	X	
Quercus virginiana (live oak)	X		
Salix nigra (black willow)		X	
Taxodium distichum (bald cypress)		X	X
Ulmus americana (American elm)	X	X	

Vegetation. For each site to be modeled, species composition, relative abundance, and density of canopy trees must be obtained. As currently formulated, FORFLO primarily models the growth of canopy tree species. For each tree species, estimates of the average diameter at breast height (DBH) and its standard deviation must be made. These data may be collected from either line transects or plots and should be at relatively constant elevations. Although not as important as DBH, average age and its standard deviation are required for each tree species.

Dominant species may be cored to establish age, or logging records may be used to establish stand ages. Development of age/DBH regressions also can be a reliable way to provide age data.

Other biological vegetation data required as model inputs are the beginning and ending dates and the length of the growing season. Finally, maximum potential stand biomass must be estimated as the total above-ground tree biomass.

Hydrologic data. The model requires hydrologic inputs in the form of an annual water stage hydrograph. FORFLO breaks the year into 24 half-month (15-day) periods; for each of these periods, the average water stage (height) and standard deviation of the water stage are required. These data, typically available from long-term gauge readings of the COE or the U.S. Geological Survey (USGS), describe current hydrologic conditions. When predicting impacts of projects that will in some way alter this hydrologic regime, future hydrologic conditions assuming project completion should be provided by the agency developing the project. The average water table depth during the growing season must be measured or estimated. For example, wells 4 feet deep can be easily dug, then measured on a weekly, biweekly, or monthly basis.

Other site data. To relate hydrologic information from a gauge reading to the bottomland site, the elevation of each sample vegetation plot must be established, typically to the nearest foot. The basic soil type must be generically described, for example, whether the soil is primarily sand, clay, or loam. Annual degree-days to a 42-degree base and standard deviation can be determined from National Oceanic and Atmospheric Administration (NOAA) climatological records.

Field studies based on point-centered quarter transects (Mueller-Dombois and Ellenberg, 1974) across a range of wet to dry sites from the top to the bottom of the basin were used to establish the initial conditions for FORFLO. Tree species frequency, dominance, density, and relative importance were estimated. At a more detailed level, tree species numbers, DBH, and replacement of tree species were measured for 2 years in wet and dry plots at the bottom of the basin, and in wet, transitional, and dry plots at the top of the basin. Tree bands during this period documented seasonal growth patterns, and tree corings in the study plots documented long-term growth increments. Basin water levels were estimated from a series of COE gauges.

HSI Model Data Inputs. The variables required for the gray squirrel, mink, downy woodpecker, swamp rabbit, and wood duck HSI models are listed in table 12-3. Tree species and size variables also were estimated from the point-centered quarter transects. Water regime variables were estimated from COE water gauge readings, maps developed by the National Wetlands Inventory (USFWS), Soil Conservation Service, and USGS, and aerial photography. Data also were collected at the field sites to estimate values of the cover variables. Sampling was

Table 12-3. List of All Variables Required in Five HSI Models (adapted from Brody et al., 1989)

Species Model	Variables
Squirrel	Percentage of canopy closure of hard mast trees
Squirrel	Number of tree species that produce hard mast
Squirrel, rabbit	Percentage of tree canopy closure
Squirrel	Average DBH of overstory trees
Squirrel	Percentage of shrub crown cover[a]
Rabbit, mink	Annual flood duration
Mink	Percentage of tree, shrub, and persistent emergent vegetation canopy closure[a]
Woodpecker	Basal area
Woodpecker	Number of snags
Wood duck	Percentage of water surface covered by winter cover[a]

[a]Variable not predicted by FORFLO.

stratified by the three major habitat types: wet bottomland hardwoods, dry bottomland hardwoods, and cypress-tupelo swamp. Within habitat types, circular (0.04 ha) plots were established randomly at each site. Estimates were made by visual observation using the method of Hays et al. (1981).

Future values for HSI model variables were taken directly from outputs of FORFLO for seven variables (table 12-3). Because FORFLO did not predict midstory or ground cover strata (i.e., percentage of shrub crown cover, percentage of shrub and persistent emergent vegetation canopy closure, and percentage of water surface covered by winter cover for the squirrel, mink, and wood duck models, respectively), predictions of future values of these variables used the field data directly. The field samples provided an average value of midstory and ground cover strata for each of the three habitat types. If a site changed habitat type, according to FORFLO predictions, these variables were assigned new values based on the average value determined by the field sampling. Thus, for each habitat type, midstory and ground cover strata were assigned the same average values in both the present and the future.

Comments on Problem Formulation

Strengths of the case study include:

- *The stressors, ecological components, and endpoints are clearly identified. A rationale is presented for selecting specific indicator species.*

- *This case provides a good example of problem formulation when the stressor involves physical stressors and habitat alterations.*

Limitations include:

- *The case study considers effects on the habitat of five animal species. A rationale is presented for their selection. However, there will always be some limitation with regard to the species chosen with respect to how well they may represent wildlife in general.*

- *Presentation of a species list for each habitat type would have been helpful. This would have permitted the reader to relate the selected species to the ranges that were present as well as to evaluate the implications of changes in habitat.*

12.3.2. Analysis: Characterization of Ecological Effects

Plant Response to Flood Conditions. In general, plant adaptations to flood stress may be characterized as physical or metabolic (Wharton et al., 1982). Physical adaptations include the ability to restore or maintain root structures in flooded conditions or to produce anatomically different roots that enhance survival in saturated soil conditions (e.g., more porous roots, pneumatophores). The primary metabolic plant adaptation to flood stress is a shift from the normal three-step process of glucose metabolism and energy (adenosine triphosphate [ATP]) production (glycolysis/Kreb's citric acid cycle/oxidative phosphorylation) to only glycolysis (an anaerobic process). In most flood-intolerant tree species, only glycolysis occurs in the absence of free oxygen (i.e., during flooded conditions) and, as a result, ethanol (an end product of glycolysis) may accumulate to phytotoxic concentrations. Flood-tolerant tree species have the capability of producing organic acid end products that can be used in cellular synthesis in stems and leaves instead of ethanol. This capability allows flood-tolerant tree species to avoid ethanol toxicity.

Despite a number of ecological studies of forested wetlands (see Conner and Day, 1982, for a review), current knowledge of vegetation dynamics, especially in response to flooding in wetland forests, is incomplete. Mature cypress (*Taxodium distichum*) and tupelo (*Nyssa aquatica*) do well under flooded conditions (Dickson et al., 1972; Kennedy, 1982). Increased flooding, however, can sometimes have serious consequences even for the most flood-tolerant trees. In Florida, Harms et al. (1980) found that in water from 20 to 100 cm deep, 0 to 16 percent of the cypress trees died in 7 years. In water over 120 cm deep, 50 percent of the cypress died after 4 years. In Louisiana, a long-term study of cypress survival was conducted in Lake Chicot (Penfound, 1949; Eggler and Moore, 1961). After 4 years of flooding with water 60 to 300 cm deep, 97 percent of the cypress were still alive. Eighteen years after flooding, 50 percent of the cypress were still alive. Most of the living trees in the deep water had dead tops (Eggler and Moore, 1961), but the numerous cypress trees still alive in the area indicate that cypress can survive for long periods in a permanently flooded situation. At Catahoula Lake in northwest Louisiana, Brown (1943) found cypress growing in waters with a seasonal variation in water level greater than 7.5 m. From the available data on flooding and cypress growth and survival, it appears that cypress can adapt to shallow (< 120 cm), permanent flooding. Even in deep water (> 120 cm), death and decline are gradual (Hall et al., 1946; Eggler and Moore, 1961; Harms et al., 1980; Klimas, 1987).

Other bottomland hardwood tree species are less tolerant of flooding than cypress and tupelo. Many factors such as age and size of the tree, soil type, depth of flooding, time of flooding, duration of flooding, and state of the floodwaters exert an influence on tree growth and survival (Hook and Scholtens, 1978). The relative waterlogging tolerance ranking of the major tree species in the Lake Verret Basin is summarized in table 12-4.

The references reviewed above provide some idea of what happens when water levels are raised suddenly and large increases in depth occur, but the rate of change in vegetational communities caused by a gradual rise in water level as documented in the coastal forests of Louisiana (Conner and Day, 1988) is harder to determine.

<u>Selection of the FORFLO and HSI Models</u>. Because of the interaction of environmental factors in influencing forested wetland biology and succession, FORFLO was chosen for predicting the future plant communities in the Lake Verret Basin. FORFLO is a simulation model developed by modifying FORET (Shugart and West, 1977), a well-known upland deciduous forest succession model. A number of other upland forest models have been derived from FORET, such as FORAR (Meilke et al., 1977, 1978), FORMIS (Tharp, 1978), BRIND (Shugart and Noble, 1981), and FORICO (Doyle, 1981). Shugart (1984) and Dale et al. (1985) provide a more comprehensive

Table 12-4. Waterlogging Tolerance Ranking of Major Tree Species in Lake Verret Basin (adapted from Hook, 1984)

Ranking[a]	Species
Most tolerant	*Nyssa aquatica* (water tupelo) *Taxodium distichum* (bald cypress)
Highly tolerant	*Carya aquatica* (water hickory)
Moderately tolerant	*Acer rubrum* (red maple) *Fraxinus pennsylvanica* (green ash) *Liquidambar styraciflua* (sweetgum) *Quercus nuttallii* (nuttall oak) *Ulmus americana* (American elm)
Weakly tolerant	*Celtis laevigata* (sugarberry) *Quercus nigra* (water oak)

[a]Most tolerant—those species capable of living from seedling to maturity in soils waterlogged almost continually, except for short durations during droughts.

Highly tolerant—those species capable of living from seedling to maturity in soils waterlogged for 50 to 75 percent of the year.

Moderately tolerant—those species capable of living from seedling to maturity in soils waterlogged about 50 percent of the year.

Weakly tolerant—those species capable of living from seedling to maturity in soils temporarily waterlogged for 1 to 4 weeks of the year or about 10 percent of the growing season.

Least tolerant—those species capable of living from seedling to maturity in soils occasionally waterlogged for only a few days, usually <2 percent of the growing season.

discussion of these and other forest succession models. SWAMP (Phipps, 1979) was the only other forested wetland model available, but FORFLO, which includes some modified growth functions from SWAMP, was developed for use in regulatory applications.

The wildlife assessment processes of HSI models were linked to FORFLO's ability to forecast future habitat conditions. HSI models are based on published studies of the basic food, shelter, and reproductive requirements of selected wildlife species. The succession model outputs of FORFLO provided input values to HSI models for assessing future value of the habitat to wildlife. FORFLO predicts tree species presence and abundance, individual tree size, canopy closure, flood duration, and other habitat measures that affect wildlife, on an annual basis for as many years as required for the impact analysis. Thus, a mechanism is available to quantify a field assessment of the current habitat values (HSI models), to quantify future habitat conditions (FORFLO), and to assess future habitat values (HSI models).

The parameters influencing tree species presence and growth in the FORFLO model are shown in figure 12-3. The model contains a library of tree data for common bottomland hardwood species that defines how tree growth and reproduction respond to changes in site quality, flooding, and the presence of other tree species. For example, each tree species has an assumed range of annual duration in which it can survive, a maximum growth rate at an optimal water level, and particular needs for wet and dry periods throughout the year for seed germination, seedling survival, and growth (for reviews of water-tolerance characteristics of bottomland tree species, see Bedinger, 1971; Teskey and Hinckley, 1977; Hook and Scholtens, 1978; Bedinger, 1979; Clark and Benforado, 1981). Temperature is represented by the degree-days at a site. The range of a tree species is determined by its maximum and minimum degree-day requirements, and it is assumed that the tree species grows best at the center of its range. The effects of shading and crowding are incorporated and represent competition among trees on a site, which may reduce their growth. Trees are entered into the model either as seedlings or sprouts. Flood duration, browsing, and soil variables reduce or enhance successful recruitment. Subsidence or accretion of ground elevation on a site directly affects flood durations and depth of the water table. The probability of a tree dying increases as the tree approaches the maximum age for the species. Trees also have a high probability of death in the model when their growth slows to less than 10 percent of their optimum growth. The model tracks the species type, DBH, and age of each tree on the simulated plot from the time the tree enters the plot as a seedling or sprout until it "dies."

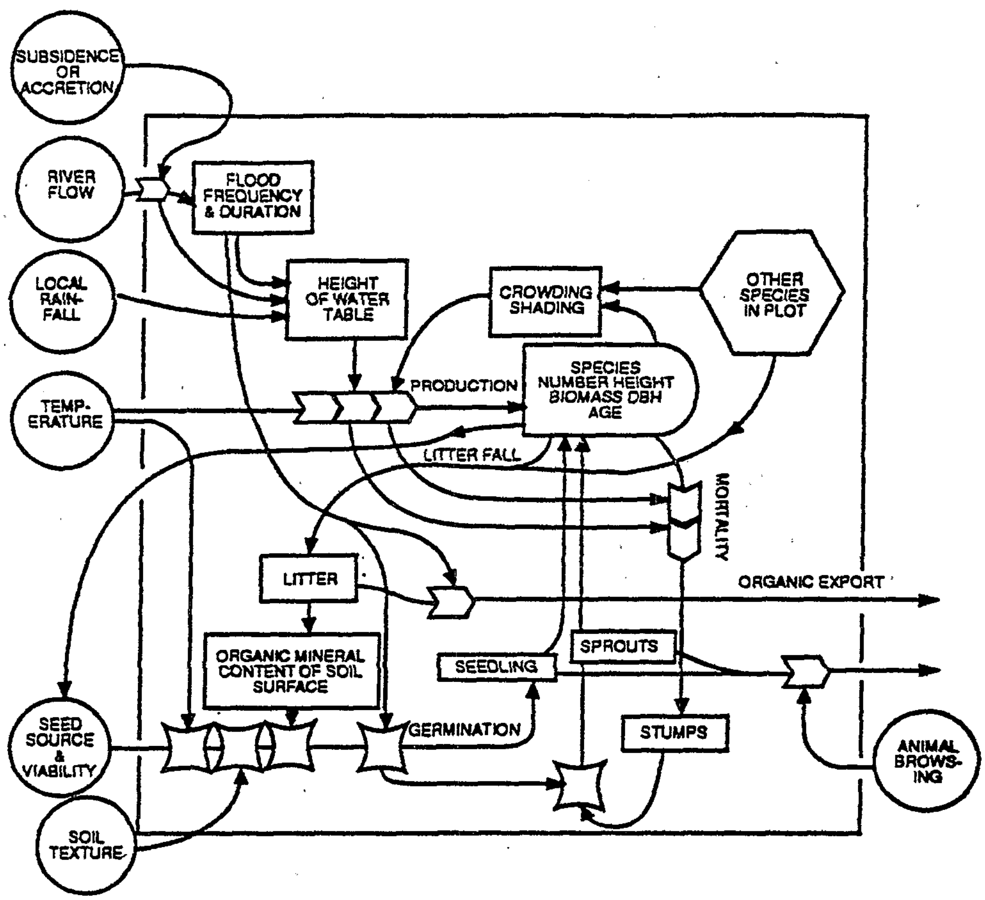

Figure 12-3. Diagram of dynamics contained in FORFLO (Pearlstine et al., 1985)

Comments on Analysis: Characterization of Ecological Effects

Strengths of the case study include:

- *The FORFLO model is based on a range of studies that relate tree growth and survival to changes in water-level elevation.*

- *The HSI models provide a framework for evaluating potential ecological effects associated with habitat modification.*

- *This case study is a good illustration of the application of ecological effects models.*

Limitations include:

- *The FORFLO and HSI models were developed independently. As such, the output of FORFLO does not satisfy all the input requirements for the HSI model.*

- *Reviewers felt it would have been helpful to have more discussion of the sensitivity of model output to the selected input variables.*

- *The presence and success of a wildlife species will depend on more than the availability of suitable habitat. Other factors not considered in the model (such as completion, predation, disease, etc.) may be important, and this introduces some uncertainty into the analysis.*

12.3.3. Analysis: Characterization of Exposure

The exposure regime was simulated by the FORFLO model, which simulates changes in water elevation and assesses the impacts of the timing, duration, and magnitude of hydrological effects. For this case study, this simulation model was used to represent the change in hydrological conditions associated with natural subsidence, providing a baseline against which other conditions can be compared. A value of 0.5 cm/yr was estimated to be the lowest estimate of subsidence in the basin. Selected simulations also were conducted using a higher subsidence rate of 1.0 cm/yr.

The results of applying the FORFLO simulation model for the two subsidence rates are presented as part of the risk characterization (section 12.3.4). Exposure and effects information, however, is incorporated within the simulation model.

Comments on Analysis: Characterization of Exposure

Strengths of the case study include:

- *The study examines a baseline case for subsidence. This can be used to gauge the effects of more severe exposure conditions.*

Limitations include:

- *The model does not address physical burial of seedlings by sedimentation and other similar factors. It should be noted that sedimentation processes independent of water level are not addressed by the model. This factor introduces uncertainty into the analysis.*

- *Limited information was available for selecting subsidence rates. It would be helpful to know the projected range of subsidence rates for the area.*

- *Levee construction is one of several processes that can influence hydroperiod (and depth of flooding). Since the flood gauge readings used as input to the FORFLO model integrate water-level changes from many sources, relative source contributions are not easily separated.*

- *The model may not apply in other situations where the detrimental effects of sedimentation, such as burial of seedlings, are critical.*

12.3.4. Risk Characterization

Risks to Vegetation Due to Subsidence. FORFLO was first used to test the effects of subsidence on the bottomland forest communities. For illustrative purposes, subsidence was set at a low enough rate to enable the development of a mature community before it was replaced by a more water-tolerant community. The results showed classic succession as the forest community responded to increased flooding durations. Upland tree species first responded favorably to the increased wetting of the soil, but as the flood durations continued to increase, the upland tree species were replaced by a bottomland hardwood community and finally cypress-tupelo. As the modeled flooding conditions became too great to support cypress-tupelo, the site became nonforested to marsh or open water. This trend also was observed in the subcanopy with the exception of a quicker response.

FORFLO was applied to a Lake Verret Basin wet bottomland hardwood site as characterized in table 12-5. With a low subsidence rate of 0.5 cm/yr, succession from wet bottomland hardwood to water tupelo (*Nyssa aquatica*) dominance occurred within 50 years (figure 12-4). As the simulation continued, bottomland hardwoods were completely replaced by water tupelo and some bald cypress (*Taxodium distichum*) within 120 years, and water tupelo was no longer sustained after 240 years.

Table 12-5. **Initial Tree Composition and Density for Three Simulated Sites (Brody et al., 1989)**

Species	Wet Bottomland Hardwood	Dry Bottomland Hardwood	Swamp
Acer rubrum (red maple)	0.02	0.03	--
Carya aquatica (water hickory)	0.09	0.03	--
Celtis laevigata (sugarberry)	0.39	--	
Diospyros virginiana (persimmon)	0.02	--	--
Fraxinus pennsylvanica (green ash)	0.28	0.05	--
Gleditsia triacanthos (honey locust)	--	0.05	--
Liquidambar styraciflua (sweetgum)	0.02	0.15	--
Nyssa aquatica (water tupelo)	--	--	0.78
Populus deltoides (cottonwood)	0.14	--	--
Quercus lyrata (overcup oak)	--	0.05	--
Quercus nigra (water oak)	--	0.39	--
Quercus nuttallii (nuttall oak)	0.02	0.05	--
Quercus virginiana (live oak)	--	0.26	--
Salix nigra (black willow)	0.28	--	--
Taxodium distichum (bald cypress)	0.02	--	2.96
Ulmus americana (American elm)	0.02	0.10	--

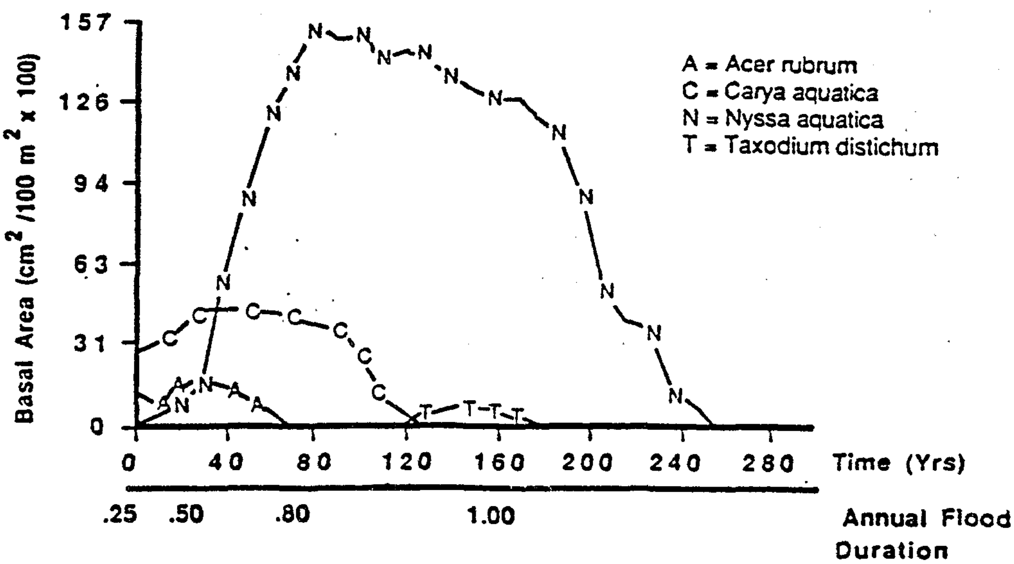

Figure 12-4. **Succession in wet bottomland hardwood site when a subsidence rate of 0.5 cm/yr is assumed. Lower horizontal scale expressed as proportion of the year that it is flooded (Brody et al., 1989)**

Because 0.5 cm/yr was the lowest estimate of subsidence in the basin, the same plot was simulated at a 1.0 cm/yr subsidence rate. With this rate, bottomland hardwoods were completely replaced by the swamp community in 70 years, followed by the complete removal of swamp species from the plot by year 140. The conservative estimate of 0.5 cm/yr was used for the remaining simulation of the Lake Verret Basin forest succession. If longevity and continued conservation of bottomland hardwoods in the basin are the desired results, then these simulations represent the best-case scenario.

For a drier bottomland hardwood site (table 12-5), water hickory (*Carya aquatica*) was dominant by year 50, but bottomland hardwoods were no longer regenerating and were replaced by the recruitment of water tupelo and cypress (figure 12-5). When a swamp site (table 12-5) was modeled, cypress and tupelo quickly stopped regenerating as flood durations increased, but almost 200 years elapsed before all the mature trees on the site were dead (figure 12-6). Net production in the swamp began declining almost immediately with the increased flooding duration, indicating that while the trees were not immediately killed, they were stressed many years before dying.

Potential Risks to Wildlife. Current and 50-year predictions of future HSI values show a general trend toward a loss of wildlife values in all habitat types (table 12-6). HSIs for squirrel, woodpecker, and wood duck decreased in all but a few cases where they remained virtually the same. The swamp habitat lost almost all values to wildlife species. Habitat for two modeled wildlife species, mink and swamp rabbit, increased in value as the flood duration in some areas increased and retained some cover vegetation. Loss of squirrel habitat value was caused primarily by the disappearance of hard mast-producing tree species (table 12-7). The only remaining mast-producing tree was the flood-tolerant water hickory. Although not a preferred mast source of the gray squirrel, this tree species is used if it is the only type available. Habitat values for the downy woodpecker declined as the potential for snags for nest sites decreased. During the onset of forest decline there may be a short-term increase in the number of dead or dying trees, but that also diminishes as the rate of replacing old trees with new trees declines. The suitability values (table 12-7) for wood duck seem counter-intuitive, with the dry forest having higher value than swamp. Within the swamp, however, water levels are permanently too deep for winter-persistent herbaceous cover. This variable for cover is necessary in the model to supply quality wintering habitat. Drier areas have much more cover and are often flooded in winter and spring months, with permanently flooded areas nearby. The increasing flood durations cause the eventual decline in winter cover for wood duck.

Summary of Risks to Vegetation and Wildlife. Regional subsidence is a major process altering the Lake Verret Basin. Flood duration and heights are increasing and apparently have been for many years. Quantitative descriptions of the ongoing changes must concentrate on the rates of these changes as well as their structure. Even with the conservative estimates of subsidence that were assumed in this study, FORFLO predicted a rapid decline in bottomland hardwoods and in the well-being of swamp tree species. Eventual nonforested conditions were predicted throughout most of the basin as a result of increases in water levels caused by land subsidence. These sites may succeed to fresh marsh or open water. Higher subsidence rates may be realistic (Conner et al., 1986; Slater, 1986) and would accelerate these trends.

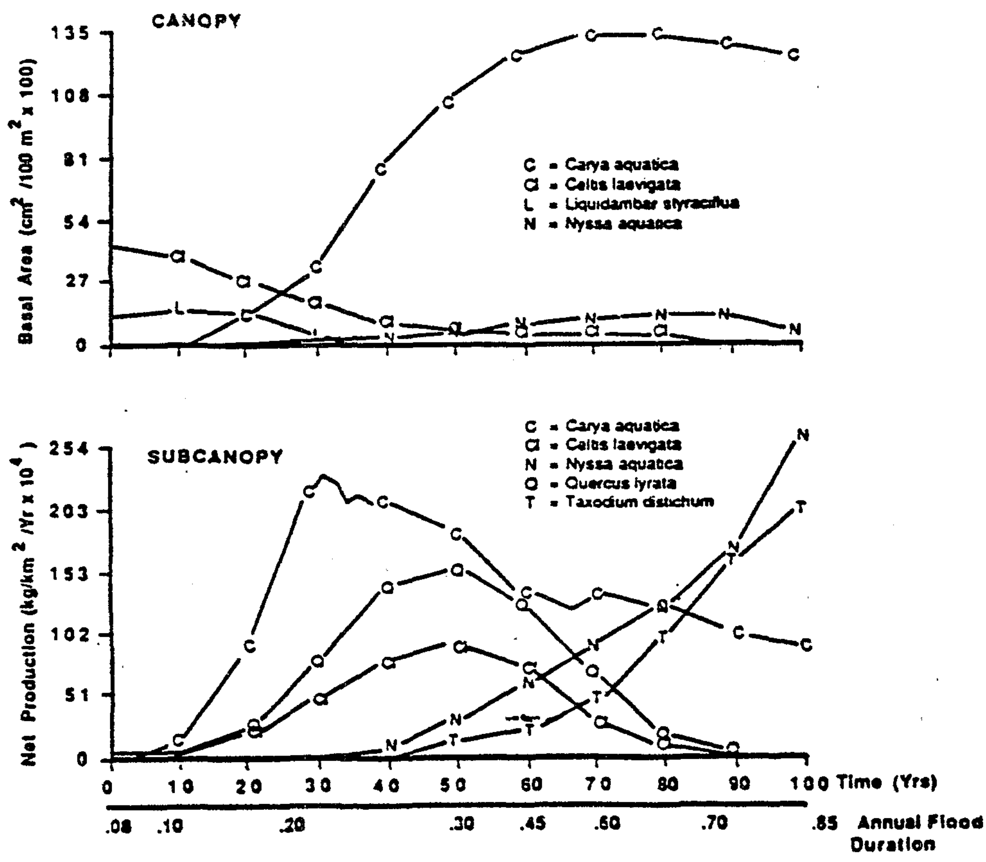

Figure 12-5. Succession in the canopy and subcanopy of the dry bottomland hardwood site when a subsidence rate of 0.5 cm/yr is assumed (Brody et al., 1989)

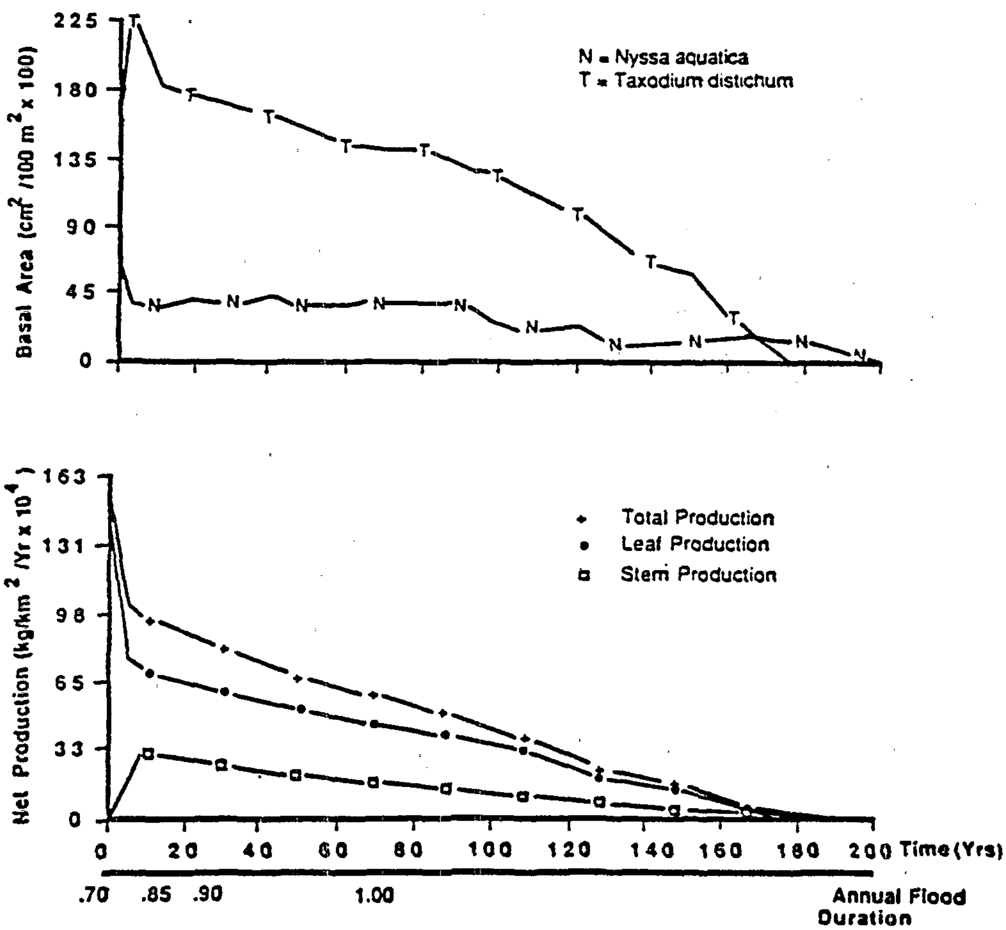

Figure 12-6. **Succession and production in the cypress-tupelo swamp when a subsidence rate of 0.5 cm/yr is assumed (Brody et al., 1989)**

Table 12-6. Current and Future Habitat Suitability Indices (HSI) for Five Species Evaluated (Brody et al., 1989)

	Dry Bottomland Hardwood		Wet Bottomland Hardwood		Swamp	
	Current HSI	Future HSI	Current HSI	Future HSI	Current HSI	Future HSI
Gray squirrel[a]	0.66	0.43	0.47	0.27	0	0
Mink[a]	0	0.37	0.60	0.83	1.00	0.38
Downy woodpecker	0.5	0.28	0.50	0.28	0.50	0.09
Swamp rabbit	0.57	0.77	0.42	0.10	0	0
Wood duck[a]	0.7	0.7	0.62	0.30	0.54	0

[a]FORFLO did not predict midstory or ground cover strata variables for the gray squirrel, mink, and wood duck HSI models (i.e., percentage of shrub crown cover, percentage of shrub and persistent emergent vegetation canopy closure, or percentage of water surface covered by winter cover). Predictions of future values of these variables were based on collected field data that provided an average value of midstory and ground cover strata for each of the three habitat types.

Table 12-7. **Current and Future Suitability Indices (SI) for Variables of Five HSI Models (Brody et al., 1989)**

Variable	Dry Bottomland Hardwood		Wet Bottomland Hardwood		Swamp	
	Current SI	Future SI	Current SI	Future SI	Current SI	Future SI
Mink						
Percentage of tree, shrub, and persistent emergent vegetation canopy closure	1	0.97	1	0.72	1	0.15
Annual flood duration	0	0.14	0.36	0.96	1	1
Swamp rabbit						
Percentage of tree canopy closure	0.57	0.96	0.53	1	0.76	0.44
Flood duration	1	0.8	0.8	0.1	0	0
Gray squirrel						
Percentage of canopy closure of hard-mast trees	0.55	0.93	0.45	0.38	0	0
Number of tree species that produce hard mast	0.8	0.2	0.5	0.2	0	0
Percentage of tree canopy closure	0.86	1	0.82	1	0.95	0.28
Average DBH[a] of overstory trees	1	1	1	1	1	1
Percentage of shrub crown cover	1	0.96	0.95	1	0.62	0.64
Downy woodpecker						
Number of snags	1	0.28	1	0.28	1	0.09
Basal area	0.5	1	0.5	1	0.5	0.88
Wood duck						
Percentage of water surface covered by winter cover	0.7	0.7	0.62	0.3	0.54	0

[a]DBH = diameter at breast height.

Several processes changing the basin's forested wetland are affecting wildlife and have probably been doing so for many years. The decline of hard mast-producing tree species means that less winter food is available. Some oak populations remain, but as the simulations show, they have no chance of any significant regeneration. There will be fewer tree species, as only the most flood-tolerant will regenerate. FORFLO provides no direct prediction of ground cover, but it may be inferred from the flood duration predictions of the model that ground cover will eventually disappear. The positive side of these changes is that wetter conditions will probably create short-term benefits to some aquatic wildlife species.

The changes in wildlife habitat are both an indirect consequence of the changing hydrology altering the forest communities and their habitat structure and a direct consequence of an even longer period of flooding. Once again, the rate of change is as significant an attribute as the changes themselves. This factor is particularly true for practical applications of this assessment methodology to federal projects. In these applications, the project's impact on wildlife habitat typically will be considered for time periods representing the "life of the project." Thus, the analysis using HSI models was performed at the 50-year point, but simulated forest changes were applied for far longer periods to provide insight on longer-term impacts on wildlife habitat in the basin.

Uncertainties and Limitations. This section presents the uncertainties and limitations associated with the case study results. The discussion begins first with the FORFLO model and its validation, then proceeds to the HSI models and their relationship with the Habitat Evaluation Procedure (HEP).

FORFLO. FORFLO primarily assesses the presence or absence of trees as a function of hydrological conditions over time. Not all interspecific and intraspecific interactions between wildlife and plant species are represented in the model (e.g., understory shrub species, exotics). Nevertheless, since canopy-level vegetational succession is predictable to a certain extent, the structural and physical features of habitats also are predictable. Thus, future habitat values were projected by incorporating the outputs of FORFLO as inputs to HSI models. However, the number of individuals fluctuates naturally over time and is often independent of the structural and physical features of the available habitat. These fluctuations can be difficult to measure or predict and are often caused by other stochastic events such as disease, predation, and competition. FORFLO does not factor in chemical stresses and thus cannot express these types of impacts for interpretation within the HSI models.

Confidence in the predictions of FORFLO models has been established by independent field validation studies. For example, results of a study by Pearlstine et al. (1985) showed good agreement between FORFLO simulations and field observations of forest composition along a 25-km reach of forested floodplains in South Carolina (see figure 12-7). Field observations in this particular study were conducted by both the USFWS and the University of South Carolina. Importance values (the sum of the relative density, relative dominance, and relative frequency of each tree species) are used to describe observed and predicted forest composition. The tree species listed along the bottom of the graph are all the species available to the FORFLO model for recruitment (Pearlstine et al., 1985).

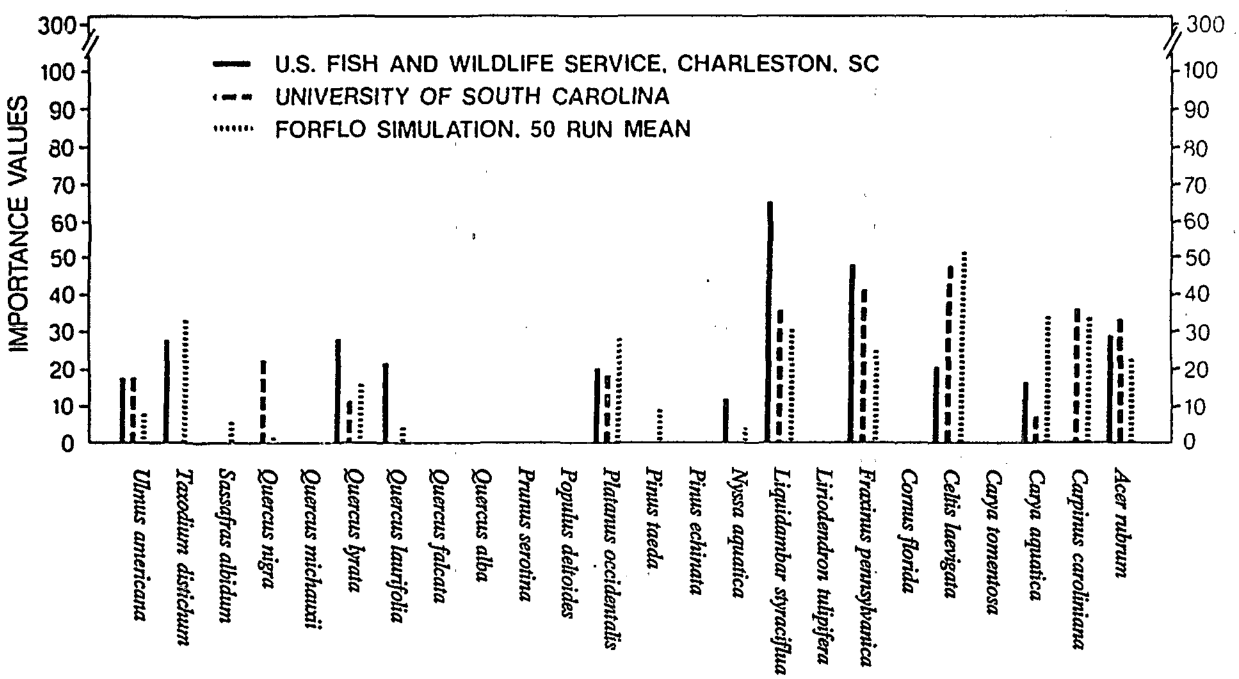

Figure 12-7. Comparison of FORFLO simulation results with field observations (Pearlstine et al., 1985)

HEP and HSI models. HSI models are an integral part of the HEP developed by the USFWS. In HEP analyses, the end result is to derive and compare Habitat Units (HUs), which represent both the quality and quantity of habitat for chosen wildlife indicator species (in this case study, gray squirrel, mink, downy woodpecker, swamp rabbit, and wood duck). Results of HSI models represent habitat quality; habitat quantity is expressed as some unit of surface area (typically acres). Hus for each wildlife species are derived by the following equation:

$$HU = (HSI \times Acreage)$$

Present and future Hus were beyond the scope of the original study by Brody et al. (1989). As a result, the conclusions in this case study do not take into account instances, for example, where low-quality, high-acreage habitats may be more beneficial to the wildlife indicator species than high-quality, low-acreage habitats.

Like any other approach used for impact or risk assessment, HSI models and HEP have limitations that define the limits of application and identify potential problem areas where good professional judgment is required. A habitat approach basically limits application of the methodology to those situations in which measurable and predictable habitat changes are important variables, but there are no assurances that wildlife populations will exist at the potential levels or optimal levels predicted by habitat analyses. Another limitation is that the wildlife species/habitat-based assessment methodology is applicable only for the wildlife species evaluated and does not necessarily relate to other wildlife species associated with other ecosystem components. Nevertheless, this should not prevent users of HEP and the FORFLO/HEP linkage from making scientifically sound, qualitative statements about potential beneficial or adverse impacts to other important flora and fauna in the study area. Keep in mind, however, that such statements must include caveats that they are qualitative inferences, not reliable facts.

HEP does not provide the user with any guidance for performing future predictions. Therefore, projected impacts are only as reliable as the user's ability to predict future conditions. *FORFLO is designed to provide a better methodological approach for predicting future habitat conditions.*

Comments on Risk Characterization

Strengths of the case study include:

- *The case study is a good example of an ecological risk assessment where physical alteration and habitat modification are the stressors.*

- *The case study illustrates a methodology that could be used to assess future alterations (e.g., accelerated rates of change in water elevation).*

Comments on Risk Characterization (continued)

- *The study was considered to be of good scientific quality. FORFLO provides a valuable tool for predicting changes in habitat quality as input to HSI.*

- *The case study illustrates how ecological effects-exposure models may be used to predict changes (risks) associated with forecasted changes in exposure. Further, the case study indicates an effort to use and combine available tools to assess risks.*

Limitations include:

- *Even if habitat is optimal, wildlife populations may not exist.*

- *HSIs are applicable for the species modeled, but not necessarily for other wildlife species.*

- *Only limited validation of FORFLO/HSI model predictions is available; thus, there is uncertainty in the analysis.*

12.4. RECOMMENDATIONS AND FUTURE RESEARCH NEEDS

This case study illustrates that FORFLO can improve decisions on environmental problems regarding terrestrial or wetland forests. Despite model limitations, there are obvious benefits to be derived from continued work on HSI models, FORFLO, and the interface between them. HSI models are already fairly well known, and FORFLO is an appropriate choice to forecast habitat changes in floodplain forests. FORFLO has potential for providing EPA managers with more insight on (1) the physical impacts of proposed Resource Conservation and Recovery Act (RCRA) sites, dredge and fill activities, and other Agency actions affecting forested wetlands; (2) the probable success of wetland remediation efforts; (3) the likelihood that socially beneficial functions associated with forested wetlands will be present through time; and (4) possible hydrological criteria. FORFLO has been linked with geographic information systems and also may play an important role in assessing cumulative impacts.

EPA's Science Advisory Board (U.S. EPA, 1990b) has ranked physical habitat alteration and biological depletion as posing high ecological risks and has called upon EPA to direct its efforts to "those most critical environmental problems where the greatest risk reduction can be obtained." As a result, ecological risk assessors within the Agency should expand beyond chemical-specific assessments and consider the potential regulatory applications of FORFLO, HSI models, and other ecological models and techniques that assess physical environmental stressors.

Future research to improve FORFLO could include better linkages with HSI models or other regulatory applications, better stress-response descriptions of the effects of hydrological changes on tree species, stress-response information on shrub species and tree species not already in the model, and the incorporation of properties that would enable the model to make predictions at the landscape or regional level.

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