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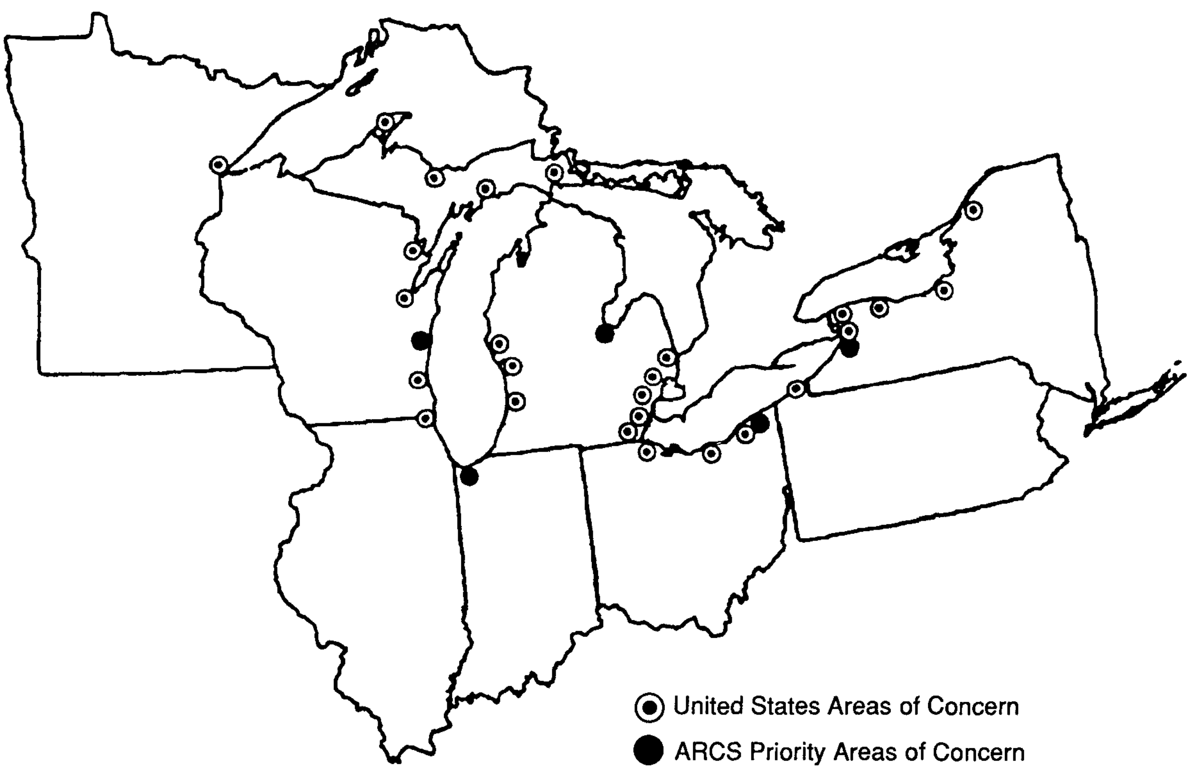
Assessment and Remediation Of Contaminated Sediments (ARCS) Program

→RISK ASSESSMENT AND MODELING OVERVIEW DOCUMENT



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Great Lakes National Program Office
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77 West Jackson Boulevard
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DISCLAIMER

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This report was prepared for the Risk Assessment/Modeling Work Group as part of the Assessment and Remediation of Contaminated Sediments (ARCS) Program administered by USEPA's Great Lakes National Program Office (GLNPO) in Chicago, Illinois. Dr. Marc Tuchman of GLNPO served as chairman of the Risk Assessment/Modeling Work Group. Mr. David Cowgill of GLNPO and Dr. Tuchman served as the ARCS Program managers.

ABSTRACT

This document provides an overview of risk assessment and modeling methods as applied to areas with contaminated sediments in the Great Lakes region. The document was prepared under the Assessment and Remediation of Contaminated Sediments (ARCS) Program, administered by the U.S. Environmental Protection Agency's (USEPA) Great Lakes National Program Office (GLNPO), in Chicago, Illinois.

The goal of the risk assessment and modeling studies was to develop and demonstrate a comprehensive human health and ecological risk assessment framework for use in the evaluation of alternative remedial actions for contaminated sediments. As part of that effort, risk assessment and modeling studies were performed at selected Areas of Concern in the Great Lakes region. The goal of those studies was to provide estimates of potential changes in exposure and risk that may occur either under a no-action alternative or following implementation of various remedial alternatives for contaminated sediments. The risk estimates may then be used to aid in the selection of an appropriate remedial action. This document does not provide detailed guidance on risk assessment and modeling methods, but refers the reader to pertinent source documents for further information.

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ACRONYMS AND ABBREVIATIONS

AOC	Area of Concern
ARCS	Assessment and Remediation of Contaminated Sediments
Corps	U.S. Army Corps of Engineers
CSF	cancer slope factor
CSO	combined sewer overflow
GLNPO	Great Lakes National Program Office
IRIS	Integrated Risk Information System
LOAEL	lowest-observed-adverse-effect level
NOAEL	no-observed-adverse-effect level
PCB	polychlorinated biphenyl
QA/QC	quality assurance and quality control
RAM	Risk Assessment/Modeling
RAP	Remedial Action Plan
RfC	reference concentration
RfD	reference dose
STORET	storage and retrieval
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

1. INTRODUCTION

This document provides an overview of risk assessment and modeling methods as applied to areas with contaminated sediments in the Great Lakes region. It was prepared under the Assessment and Remediation of Contaminated Sediments (ARCS) Program, administered by the U.S. Environmental Protection Agency's (USEPA) Great Lakes National Program Office (GLNPO) in Chicago, Illinois.

BACKGROUND

Although toxic discharges in the Great Lakes and elsewhere have been reduced in the last 20 years, persistent contaminants in sediments continue to pose a potential risk to human health and the environment. High concentrations of contaminants in bottom sediments and associated adverse effects have been well documented throughout the Great Lakes and associated connecting channels. The extent of sediment contamination and its associated adverse effects have been the subject of considerable concern and study in the Great Lakes community and elsewhere. Contaminated sediments can have direct toxic effects on aquatic life, such as the development of cancerous tumors in fish exposed to polycyclic aromatic hydrocarbons in sediments. The bioaccumulation of toxic contaminants in the food chain can also pose a risk to humans, wildlife, and aquatic organisms. As a result, advisories against consumption of fish are in place in many areas of the Great Lakes. These advisories have also had a negative economic impact on the affected areas.

To address concerns about the deleterious effects of contaminated sediments in the Great Lakes, Annex 14 of the Great Lakes Water Quality Agreement between the United States and Canada stipulates that the cooperating parties will identify the nature and extent of sediment contamination in the Great Lakes, develop methods to assess impacts, and evaluate the technological capability of programs to remedy such contamination. The 1987 amendments to the Clear Water Act, in § 118(c)(3), authorized GLNPO to coordinate and conduct a 5-year study and demonstration projects relating to the appropriate treatment of toxic contaminants in bottom sediments. Five areas were specified in the Act as requiring priority consideration in conducting demonstration projects: Saginaw Bay, Michigan; Sheboygan Harbor, Wisconsin; Grand Calumet River, Indiana; Ashtabula River, Ohio; and Buffalo River, New York. To fulfill the requirements of the Act, GLNPO initiated the ARCS Program. In addition, the Great Lakes Critical Programs Act of 1990 amended the section, now § 118(c)(7), by extending the program by 1 year and specifying completion dates for certain interim activities. ARCS is an integrated program for the development and testing of assessment techniques and remedial action alternatives for contaminated sediments. Information from ARCS Program activities will help address contaminated sediment concerns in the development of Remedial Action

Plans (RAPs) for all 43 Great Lakes Areas of Concern (AOCs), as identified by the United States and Canadian governments, as well as lakewide management plans.

To accomplish the ARCS Program objectives, the following work groups were established:

- The **Toxicity/Chemistry Work Group** was responsible for assessing the current nature and extent of contaminated sediment problems in the five priority AOCs by studying the chemical, physical, and biological characteristics of contaminated sediments and their biotic communities, and for demonstrating cost-effective assessment techniques that can be used at other Great Lakes AOCs.
- The **Risk Assessment/Modeling (RAM) Work Group** was responsible for assessing the current and future risks presented by contaminated sediments to human and ecological receptors under various remedial alternatives (including the no-action alternative) at the five priority AOCs.
- The **Engineering/Technology Work Group** was responsible for evaluating and testing available removal and remedial technologies for contaminated sediments, for selecting promising technologies for further testing, and for performing field demonstrations at each of the five priority AOCs.
- The **Communication/Liaison Work Group** was responsible for facilitating the flow of information from the technical work groups and the overall ARCS Program to the interested public and for providing feedback from the public to the ARCS Program on needs, expectations, and perceived problems.

This document is intended to provide an overview of the risk assessment and modeling methods developed by the ARCS RAM Work Group and to provide general guidance on their application to other Great Lakes AOCs.

RISK MANAGEMENT FRAMEWORK

Sediment contamination is of concern primarily because of the potential risks it poses to humans, wildlife, and aquatic organisms. Therefore, the management of contaminated sediments includes the overall process of risk management. For this project, risk management is defined as the process of integrating findings from a risk assessment with engineering, policy, and nontechnical concerns to make decisions about sediment remediation at a specific site or to set remediation priorities among sites. Risk management should be distinguished from risk assessment, which is the process of producing qualitative or quantitative estimates of the potential risks associated with exposure to specific concentrations of contaminants under specific current or future exposure conditions at a site.

The general objective of the ARCS RAM Work Group was to develop and demonstrate a comprehensive risk management framework for: 1) identifying existing risks to human health and ecological receptors at sites with contaminated sediments, 2) estimating the potential impact of various sediment remedial alternatives on contaminant concentrations in various media and their associated risks, and 3) comparing existing and potential future risks to aid in the selection of sediment remedial alternatives.

Steps in the overall risk management process are illustrated in Figure 1-1. A general discussion of each of the steps is provided below, followed by a more detailed description of the use of risk assessment and modeling in the ARCS RAM studies.

Step 1. **Initial Screening of Potential AOCs:** The first step in the risk management process involves the use of screening-level assessments to identify sites that may pose a potential threat to human health or ecological receptors based, in part, on sediment contamination. The Great Lakes states, the U.S. and Canadian governments, and the International Joint Commission have designated 43 AOCs around the Great Lakes on the basis of impairment of beneficial uses. All but one of these AOCs have been identified as having sufficient sediment contamination to pose potential threats to human health or ecological receptors.

Step 2. **Risk Assessment Planning:** In this step, existing information is first compiled to describe the physical features of the AOC, the general distribution of sediment contaminants and their potential sources, and the human and ecological receptor populations likely to be present. Contaminants of concern, biological species, endpoints (measured biological or ecological qualities), and primary exposure pathways for human and ecological receptors are then identified for use in the risk assessment. This information is used to develop preliminary remedial action objectives, which are general descriptions of what remedial actions should accomplish, including the reduction of risks associated with exposure to contaminated sediments. Potential remedial actions may then be identified. As part of risk assessment planning, deficiencies in the available data that might preclude an adequate baseline risk assessment should be identified. Supplementary field sampling (Step 3) may then be conducted if necessary. The risk assessment planning step provides the organizational framework for the subsequent steps in the risk management process.

Step 3. **Supplementary Field Sampling:** If data gaps were identified as part of the previous step, supplementary field sampling efforts may be required to collect the information necessary for a detailed site assessment. Additional information may need to be gathered on the physical, biological, and chemical conditions of the system to further

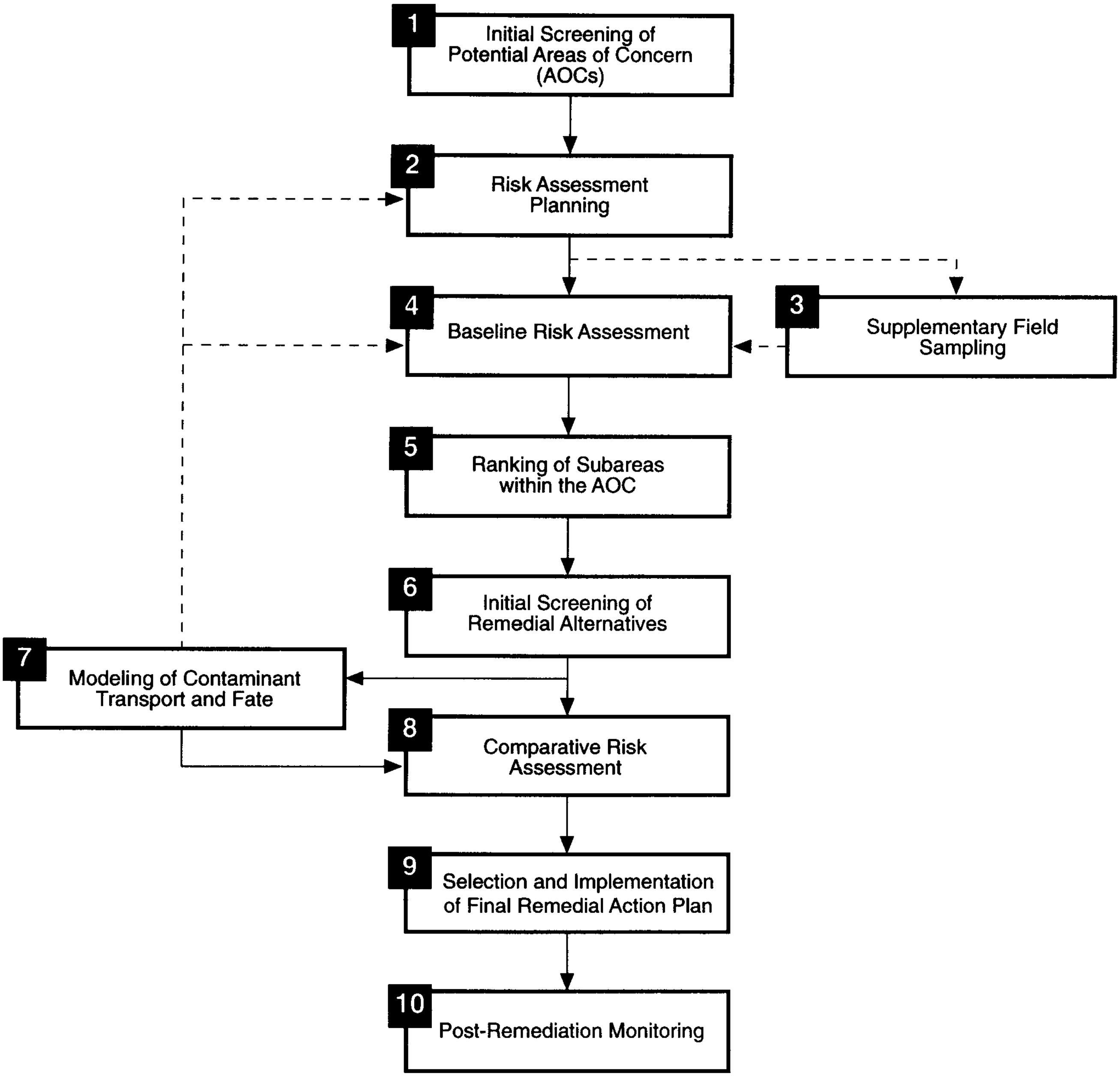

Figure 1-1. Overview of the comprehensive risk management process.

characterize the nature and extent of the sediment contamination problem. The data are also used to develop appropriate sediment remedial alternatives, to support mass balance modeling, and to conduct the comparative risk assessment of the remedial alternatives. A separate *ARCS Assessment Guidance Document* (USEPA 1993) describing field sampling methods is being prepared as part of the ARCS Program.

Step 4. **Baseline Risk Assessment:** A baseline risk assessment estimates current risks to humans, wildlife, and aquatic organisms resulting from direct and indirect exposure to contaminated sediments in the absence of any sediment remediation. The baseline risk estimates, developed using conservative, or health protective, assumptions are used to determine which contaminants and exposure pathways pose the greatest risk, to determine whether remediation is likely to be required, and to provide a baseline against which any future remedial action can be evaluated.

Step 5. **Ranking of Subareas Within the AOC:** Within a particular AOC, there will be spatial variations in the concentrations and types of sediment contaminants; variations in the risks the sediment contaminants pose to humans and ecological receptors resulting from varying exposure potential, bioavailability, or toxicity; and variations in the costs associated with sediment remediation. Available information on sediment chemistry, toxicity tests, and benthic community structure may be combined in a numerically based ranking system to prioritize specific subareas within an AOC for remedial action. Additional detail on sediment ranking procedures developed under the ARCS Program is provided in the *ARCS Assessment Guidance Document* (USEPA 1993). The results of the human health and ecological risk assessments may be qualitatively considered along with the numerical sediment ranking in this prioritization process.

Step 6. **Initial Screening of Remedial Alternatives:** There is a wide variety of possible sediment remedial alternatives, only a few of which may be practical at a particular site. This step in the risk management process involves the selection of a limited number of possible remedial alternatives (e.g., no action, *in situ* treatment, or removal alternatives) for further evaluation. Additional field sampling may be required following the selection of the sediment remedial alternatives to be evaluated. The selection of candidate subareas for sediment remediation and possible remedial alternatives is based on a detailed site assessment, which delineates the nature and extent of sediment contamination within subareas of the AOC.

Step 7. **Modeling of Contaminant Transport and Fate:** To assess the human health and ecological risks posed by various sediment remediation scenarios, contaminant releases must be estimated for each of the remedial alternatives. Previous steps of the risk management process provide information concerning the nature and extent of existing sediment contamination and estimates of baseline human health and ecological risks. However, those steps provide little information that can be used directly to estimate changes that may occur as a result of remediation. In this step, transport and fate models are used with physical, chemical, and biological data for the AOC to evaluate the effectiveness of the various remedial alternatives in reducing contaminant concentrations in environmental media of concern. Outputs from these models may include predictions of contaminant concentrations in air, water, soil, sediments, and biota based on present or projected contaminant loadings or expected changes in contaminant concentrations over time following remediation.

Step 8. **Comparative Risk Assessment:** The objective of this step is to estimate changes in risks, relative to the baseline risk, that would result from implementation of the various remedial alternatives evaluated. For example, the comparative risk assessment can be used to estimate the impacts of various remedial alternatives on human health risks from consumption of contaminated fish over time. This assessment integrates data from all previous steps into a risk assessment framework. Ideally, this comparative risk assessment should include an estimation of both the changes in risks at the AOC following sediment remediation and the changes in risks at the site of disposal of the contaminated sediments. The remedial action objectives that had been developed under Step 2 are then refined during this step.

Step 9. **Selection and Implementation of Final Remedial Action Plan:** In this step, information from the comparative risk assessment is used in conjunction with other factors (e.g., economic, political) to select the most appropriate remedial alternative(s) to implement.

Step 10. **Post-Remediation Monitoring:** The last step in the risk management process is to monitor the AOC following sediment remediation to demonstrate successful reductions in sediment contamination and associated risks to human health and ecological receptors. Monitoring should focus on parameters that have the greatest influence on risk estimates and remedy selection. For example, if the human health risk estimates are predominantly based on concentrations of polychlorinated biphenyls (PCBs) in fish, this parameter should be used as the indicator of remedial effectiveness.

The studies of the ARCS RAM Work Group have provided support for Steps 4–8 of the risk management process. Other ARCS studies deal specifically with other aspects of the decision-making process. The final results of the ARCS RAM studies are estimates of contaminant concentrations and potential risks associated with various sediment remedial alternatives that may then be used, along with other information collected at a site, to select the appropriate remedial action from among the various alternatives. The following sections of this document provide an overview of the use of risk assessment and modeling in the ARCS RAM studies.

2. HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENTS FOR CONTAMINATED SEDIMENTS

As part of the ARCS RAM studies, baseline risk assessments were performed to estimate the current health risks to humans and wildlife exposed to sediment-derived contaminants in the absence of any remediation. The results of these assessments can be used by ARCS risk managers in prioritizing sites and making decisions concerning the need for sediment remediation. The risk assessment approach developed and used in the ARCS Program is intended to produce conservative estimates of risk in order to ensure adequate protection of human health and the environment. This approach to risk assessment is specifically designed not to underestimate risks and, therefore, is likely to overestimate risks at many sites. The following sections provide an overview of the risk assessment approach used under the ARCS Program. This approach may be used to assess potential human and ecological risks at other sites with contaminated sediments.

HUMAN HEALTH RISK ASSESSMENT

Individuals in the Great Lakes region may be exposed to sediment contaminants through various activities that result in intake of contaminants through dermal, ingestion, and/or inhalation pathways. For the ARCS risk assessments, human health risk estimates were determined for both carcinogenic effects (i.e., increased probability of an individual developing cancer over a lifetime) and noncarcinogenic effects (i.e., chronic or subchronic effects other than cancer) over a range of exposure scenarios. The risk estimates were calculated by using conservative exposure assumptions and USEPA-verified toxicity values called cancer slope factors (CSFs) and reference doses (RfDs) (for noncarcinogenic effects). The primary guidance used to conduct these risk assessments was obtained from USEPA's *Risk Assessment Guidance for Superfund—Volume I: Human Health Evaluation Manual (Part A)* (USEPA 1989c), although the use of additional USEPA guidance for risk assessment (USEPA 1988b, 1989a,b, 1991) is also described herein. The following sections describe the main components used in performing the ARCS human health risk assessments (Figure 2-1), including specific examples and recommendations. Baseline human health risk assessments were conducted under the ARCS Program for the five priority AOCs: Saginaw River, Michigan (Crane 1992b); Sheboygan River, Wisconsin (Crane 1993a); Grand Calumet River, Indiana (Crane 1993b); Ashtabula River, Ohio (Crane 1992a); and Buffalo River, New York (Crane 1993c). The same human health risk assessment framework can be applied to any site with contaminated sediments.

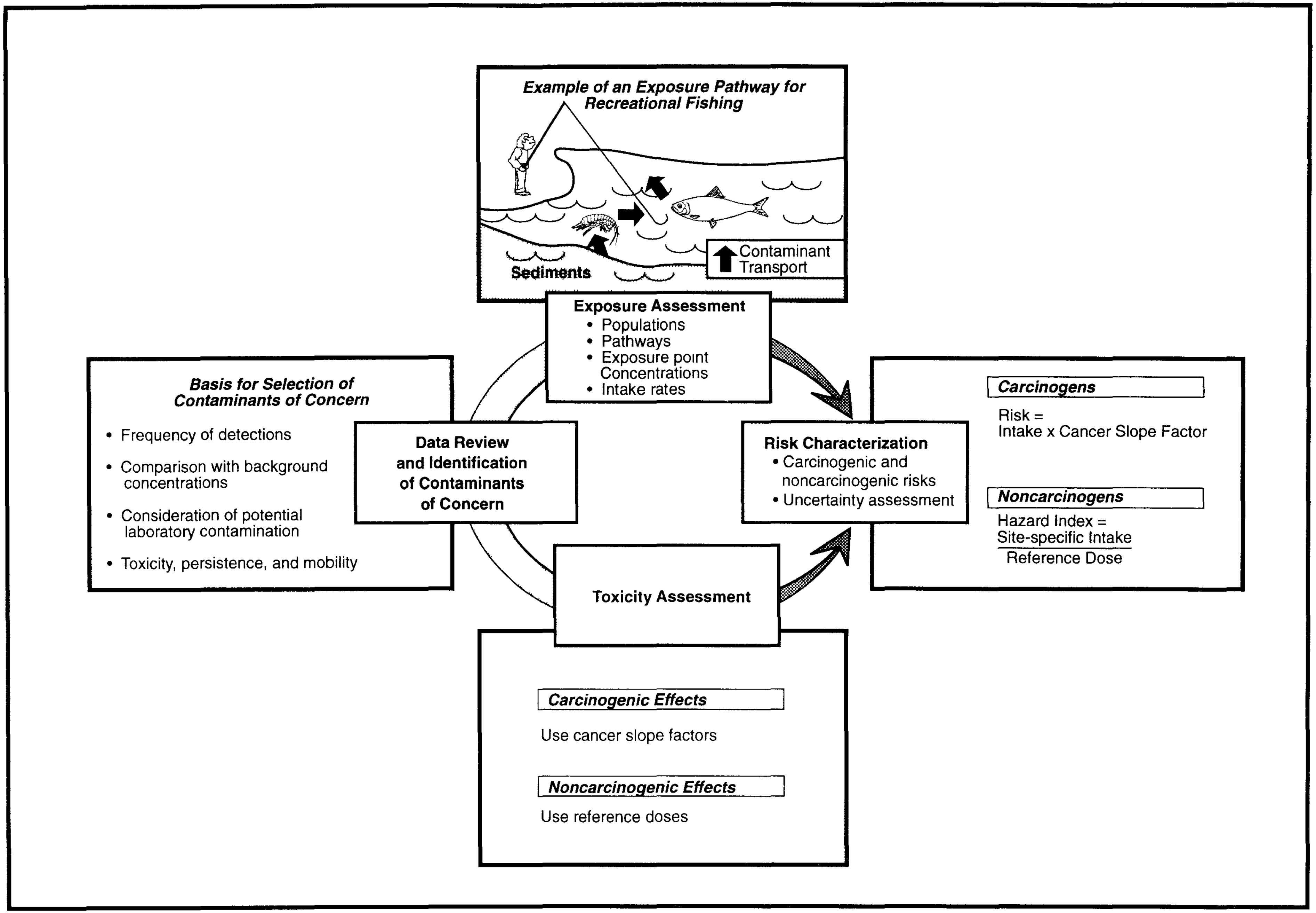

Figure 2-1. Components of a human health risk assessment.

Data Review

Prior to beginning the human health risk assessment, site data must be collected and analyzed to determine contaminant concentrations in the media of interest (e.g., air, water, sediments, biota) and potential routes of exposure to contaminated media. Available historical data, including information on site use and possible contaminant sources, should be reviewed to focus sampling efforts on contaminants known or likely to be present. Such data may include analytical data from previous sampling efforts, descriptions of the past uses of the site or other site records, and interviews with site personnel that may suggest what contaminants may be present. For example, RAPs contain information about monitoring and scientific studies that have been conducted at Great Lakes sites. USEPA's storage and retrieval (STORET) database is also a good source of water quality data that are routinely collected at U.S. Geological Survey (USGS) and statewide gaging stations. In addition, to determine current or future uses of the site, personnel at various local, State, and Federal agencies that deal with public health, natural resource, and fish and wildlife issues should also be contacted for information about the site. In particular, applicable zoning regulations, land use plans, and restrictions on site uses (e.g., fishing or hunting bans) should be described in the risk assessment. The risk assessor should visit the site, preferably during the period of greatest activity or during several seasons, to observe recreational and business uses of the AOC. It may also be helpful to interview game wardens, lifeguards, and local officials regarding site use.

A thorough sampling program for sediments, water, fish, and other important media should be conducted using appropriate quality assurance and quality control (QA/QC) procedures, such as those identified in *Guidance for Data Usability in Risk Assessment* (USEPA 1990). Water, sediment, and fish samples should be collected preferentially in areas where people are known to be using the site, such as public beaches, or are likely to use the site in the future, such as areas near shorelines or along access roads. Samples should also be collected at one or more reference sites (i.e., areas that are unlikely to be influenced by sediment contaminants within the AOC or by other anthropogenic sources). In addition, several species of fish should be sampled, including species that feed on the bottom, such as carp and catfish, and species that feed in the water column, such as walleye. Both whole body and skin-on or skin-off fillets should be analyzed for various organic and inorganic chemicals, especially those chemicals detected in the sediments and known to bioaccumulate in fish tissue.

The need for air sampling should be evaluated on a case-by-case basis. Exposures to contaminants in air are likely to contribute much less to overall site risks than exposures via other pathways such as consumption of fish. Thus, air sampling is not generally required for screening-level evaluations and was not conducted as part of the ARCS risk assessments. However, the air pathway may be important at sites with volatile contaminants of concern, where upland soils are exposed, or where areas of sediments are dry for a substantial proportion of the time and thus may be a source of airborne particulates. At some such sites, simple models may be used to estimate exposures to contaminants via the air pathway.

Once data from current and historical sources have been reviewed, the most appropriate data available for the media of interest should be selected for use in the risk assessment. The adequacy of QA/QC procedures followed in generating the analytical data should be a key criterion in selecting data. However, a complete QA/QC review may not be possible, particularly when analyses are conducted using historical data. For example, the ARCS risk assessments relied primarily on historical data, and in many cases, little QA/QC information was supplied with the data. Thus, risk assessment staff should work with the regulatory agency's project manager in determining whether a data set is adequate for a specific risk assessment application. In addition, the implications of any limitations in available data should be discussed in the risk assessment document. For example, unsuitable detection and quantification limits are often a major limitation in the use of historical data sets.

Identifying Contaminants of Concern

A list of all the contaminants detected in the media of interest at the site should be made. Inorganic chemicals present at concentrations near background levels and chemicals that are infrequently detected or that may be present as laboratory contaminants may be excluded. Where the list of contaminants of concern is extensive, a screening step can be conducted to exclude contaminants that only contribute a minimal amount to the overall site risk. For example, risks associated with the maximum detected concentration can be calculated using toxicity data available in USEPA's Integrated Risk Information System (IRIS) database and exposure assumptions that assume a higher degree of exposure than is likely to occur at the site. Such an approach is considered to be conservative because it incorporates assumptions that may overestimate risks in order to ensure that risks are not underestimated. Using this approach, contaminants can then be excluded when they contribute an individual risk of less than 1×10^{-7} (for carcinogens) or a hazard quotient of less than 0.1 (for noncarcinogens). A carcinogenic risk of 1×10^{-7} corresponds to a one-in-ten-million chance of an individual developing cancer during their lifetime. Use of these conservative target risk levels and worst-case exposure assumptions (e.g., use of the maximum detected concentration) generally ensures that chemicals with significant risks due to the cumulative effects of multiple contaminants and multiple exposure pathways are not prematurely excluded from the risk assessment.

Exposure Assessment

In the exposure assessment, the magnitude, frequency, duration, and route of direct and indirect exposures of individuals to sediment-derived contaminants from an AOC are determined. Populations that may be exposed (i.e., receptor populations) should first be identified by considering the site's proximity to population centers, the accessibility of the site, and any features such as beaches or fishing piers that would attract visitors. The predominant types of receptor populations to consider are residents, workers, and recreational visitors. Recreational uses of AOCs may include fishing, swimming, boating, or beach activities. Current and potential future exposures should be evaluated. In particular, future exposures should be evaluated when future uses of a site may increase the potential for exposure to site contaminants. For example, if a site may be used for residential purposes in the future, exposure to site soils and sediment could be greatly increased in comparison with a current recreational scenario.

Exposures to contaminants can potentially occur via three exposure routes: ingestion, dermal contact, and inhalation, each of which is in turn part of numerous exposure pathways. Ingestion of contaminants can result from inadvertent consumption of contaminated soils or sediment, or through consumption of drinking water, surface water, or wildlife. Dermal contact involves direct contact of the skin with either contaminated sediments, riverplain soils, or overlying water. Inhalation of airborne vapors or dust may introduce contaminants of concern into the respiratory system. The ingestion exposure pathways often result in higher exposure estimates than the dermal or inhalation pathways because of the greater absorption of contaminants through the gastrointestinal tract as compared with absorption through the skin, and the relatively high levels of intake of contaminants in soil, water, and food as compared with inhalation of contaminants.

The potential pathways by which people may be exposed to contaminants from an AOC are then examined to determine whether they are complete or incomplete. An exposure pathway is complete if there is: 1) a source and mechanism of chemical release, 2) a retention or transport medium (or media) whereby chemicals are transferred between media, 3) an exposure point where contact occurs, and 4) an exposure route by which contact occurs (USEPA 1989c). An exposure pathway is incomplete if any of these conditions is not met. The exposure pathways that were complete for most of the five priority ARCS sites included: 1) consumption of contaminated fish, 2) dermal contact with contaminated water, 3) limited dermal contact with contaminated sediments, and 4) limited ingestion of surface water while swimming. Incidental ingestion of sediment may also be of concern at some sites.

All complete exposure pathways should be evaluated in the exposure assessment unless certain criteria apply. These criteria include: 1) the potential magnitude of exposure from a pathway is low, or 2) the probability of the exposure occurring is very low and the risks associated with the occurrence are not high (USEPA 1989c). For example, at the Saginaw River AOC, there are no beaches along the river and swimming may occur only infrequently when people jump off recreational boats into the water. In addition to contacting the water, these people could ingest some water while swimming. In this case, the risk from ingesting surface water was considered insignificant, and an assumption was made that the health risk from dermal contact would be even lower than the health risk associated with ingestion (Crane 1992b). At some sites, it may be reasonable to assume that no fishing takes place because of the absence of edible fish or shellfish or because sites are physically inaccessible or remote.

Once the exposure pathways to be quantitatively evaluated are selected for a site, contaminant concentrations and exposure parameters are used to calculate the chronic or subchronic intake level of each contaminant (in mg of chemical per kg body weight per day) (Table 2-1). For each current and potential future exposure scenario, exposure parameters may be selected to represent typical, reasonable maximum, and, in some instances, worst-case exposure conditions. Typical, or average, exposures and reasonable maximum exposures (i.e., the maximum exposure that is reasonably expected to occur at a site) are usually evaluated for each complete pathway. In general, an average exposure case is calculated using site concentrations and exposure parameters that best represent the central tendency of the data. Under the reasonable maximum exposure case, 95th or 90th percentile values are used for contact rates, intake rates, and exposure frequency and duration variables, and the upper 95 percent confidence limit on the average concentration is used for the exposure point concentration in the contaminated media. (See also USEPA [1992c] for further clarification of calculation of exposure point concentrations.)

Site-specific information is often not available for many exposure parameters; thus, assumptions about the types and frequencies of exposure may be made based on recommended USEPA values or on professional judgment. The following documents provide useful information on estimating exposure parameters and conducting the exposure assessment: *Superfund Exposure Assessment Manual* (USEPA 1988b); *Exposure Factors Handbook* (USEPA 1989b); *Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual* (USEPA 1989a); and *Standard Default Exposure Factors: Interim Final* (USEPA 1991), which is a supplement to the Superfund risk assessment manual (USEPA 1989c); and *Dermal Exposure Assessment: Principles and Applications* (USEPA 1992a).

In some cases, it may be appropriate to determine the fractional intake of exposure that occurs at a site. The fractional intake, which is the proportion of all exposure of a given type (e.g., the fraction of all fish consumed) that comes from the site, is generally estimated based on best professional judgment of factors such as the site size and accessibility and any restrictions on site use (e.g., warning signs, fishing bans, or barriers to the site). In calculating fractional intake for fish consumption, the abundance of edible fish and shellfish at a site should be considered. Some sites may not have any fish or may not have edible aquatic species. However, although it is important to consider these limits on site use, they may not be sufficient to prevent access at a site, and thus the risk assessment should not assume that exposure will not occur. For example, although all of the five priority AOCs examined for the ARCS Program had fish advisories in effect, some people continued to fish from the river.

Because recreational fishing is very popular in the Great Lakes region and consumption of contaminated fish is an important exposure pathway, several researchers have gathered data on consumption rates of fish by Great Lakes populations. A survey of the angler population in the AOC should be made to obtain a better estimate of local fish consumption rates and patterns. If these data cannot be obtained, the results of a survey of Michigan anglers and their families by researchers at the University of Michigan (West et al. 1989) may be used to estimate more "localized" consumption patterns. The survey results can be obtained from Patrick West at the University of Michigan. An important

TABLE 2-1. GENERIC EQUATION FOR CALCULATING CHEMICAL INTAKE LEVELS

$$I = \frac{C \times CR \times EFD}{BW \times AT}$$

where:

I	Intake =	the amount of chemical at the exchange boundary (mg chemical/kg body weight-day)
Chemical-Related Variables		
C	Chemical concentration =	the average concentration contacted over the exposure period (e.g., mg/L)
Variables that Describe the Exposed Population		
CR	Contact rate =	the amount of contaminated medium contacted per unit time or event (e.g., L/day)
EFD	Exposure frequency and duration =	how long and how often exposure occurs; often calculated using two terms, EF and ED, where: EF = exposure frequency (days/year) ED = exposure duration (years)
BW	Body weight =	the average body weight (kg) over the exposure period
Assessment-Determined Variables		
AT	Averaging time =	period over which exposure is averaged (days)

Source: U.S. EPA (1989c).

result of this survey was that Michigan anglers and their families had an average fish consumption rate of 19.2 g/person-day (West et al. 1989), nearly 3 times the average fish consumption rate of people in the United States as a whole (USEPA 1989c). Anglers in other Great Lakes states may be consuming fish at a comparable rate to Michigan anglers. Additional data on fish consumption rates for sport anglers on Lake Michigan are provided as part of a nationwide survey reported in Rupp et al. (1980).

In selecting appropriate consumption rates, subsistence fishing or hunting should be considered (in addition to the average and reasonable maximum exposure cases) at sites with special subgroups of people who rely on locally caught fish, waterfowl, or other aquatic-related wildlife as their main source of protein. Examples may include members of a particular ethnic community who traditionally rely on fish as an important part of their diet (e.g., the southeast Asian community of Hmong in Sheboygan, Wisconsin) or indigent people who spend time in the area and may rely on locally caught fish for their main source of protein.

Toxicity Assessment

In a toxicity assessment, available data are reviewed to determine and quantify the relationship between the level of exposure to a contaminant (dose or intake level) and the increased likelihood and/or severity of adverse effects. This relationship is termed the dose-response relationship and provides the basis for deriving quantitative toxicity values used in the risk assessment. For carcinogenic health effects, CSFs are used to estimate the risk of developing cancer that corresponds to estimated exposure concentrations. This risk is in addition to the risk of developing cancer due to other causes and thus is often termed excess cancer risk.

The potential for noncarcinogenic health effects from oral exposures is typically evaluated by comparing estimated daily intake levels with RfDs, which represent daily intake levels at which no adverse effects are expected to occur. For assessment of inhalation exposures, USEPA has recently begun issuing reference concentrations (RfCs) that represent exposure concentrations at which no adverse effects are expected to occur.

Carcinogens and systemic toxicants are treated differently, because according to current scientific theory it is plausible that for any dose of a carcinogen there could be some finite increase in cancer risk. Systemic toxicants are considered to act via a threshold mechanism, which allows for the identification of a safe dose. Hazard identification and dose-response evaluations for more than 600 chemicals have been conducted and verified by USEPA work groups; additional chemicals are awaiting review. USEPA-verified toxicity values can be obtained by accessing USEPA's IRIS database. The IRIS User Support group can provide technical assistance and information on how to access IRIS and can be reached at (513) 569-7254.

Brief toxicity profiles on contaminants of concern should be prepared as part of the toxicity assessment. At a minimum, such profiles should contain information on the derivation of toxicity values for the contaminants of concern and should describe any uncertainties associated with the toxicity values. The following data should be gathered, to the extent available, for all contaminants of concern that have been shown to be carcinogenic in experimental animals or in human populations:

- Current CSFs from IRIS
- Weight-of-evidence classifications, which characterize the degree to which the available evidence indicates that an agent is a human carcinogen
- Type of cancer for Class A carcinogens (i.e., contaminants that have been shown to cause cancer in humans).

Pertinent data to be identified and discussed in the baseline risk assessment for contaminants associated with noncarcinogenic effects include the following:

- Current RfDs (and RfCs, if applicable) from IRIS
- Confidence level in the overall database and the critical study on which the toxicity value is based, including identification of the critical effects
- Effects that occur at doses higher than those required to elicit the critical effect
- Uncertainty factors used by USEPA in deriving the toxicity value
- 1- and 10-day health advisories for shorter-term oral exposures.

Inclusion of these background data in the toxicity assessment assists risk managers in interpreting the findings of the risk assessment.

Risk Characterization

The purpose of the risk characterization step is to combine the exposure and toxicity estimates into an integrated expression of human health risk. Three means of expressing carcinogenic and noncarcinogenic risks are presented in the risk assessment. First, chemical-specific risks are estimated for each exposure pathway. Second, these chemical-specific risks are added to estimate a cumulative pathway-specific risk. Finally, risks are added across all chemicals and relevant pathways to estimate the total human health risks to individuals exposed to contaminants from the AOC. Table 2-2 illustrates how a summary table of risk estimates may be arranged. The approaches used to quantify carcinogenic and noncarcinogenic health risks are described below.

Carcinogenic risk is expressed as the upper-bound excess probability of an individual developing cancer over their lifetime following exposure to a given chemical concentration for a specified period of time. Carcinogenic risk estimates are computed by multiplying the chronic daily intake prorated over a lifetime of exposure by the CSF for each carcinogen of interest. Carcinogenic effects are summed for all chemicals in an exposure

TABLE 2-2. ESTIMATED CARCINOGENIC AND NONCARCINOGENIC RISKS TO INDIVIDUALS RESIDING IN THE LOWER SAGINAW RIVER AREA OF CONCERN

	Individual Risks			Additive Risks	
Type of Risk and Exposure[a]	Walleye	Carp	Waterfowl	Walleye + Waterfowl	Carp + Waterfowl
Carcinogenic					
Typical	1×10^{-5}	1×10^{-4}	6×10^{-6}	2×10^{-5}	1×10^{-4}
Reasonable Maximum	2×10^{-4}	3×10^{-3}	2×10^{-4}	4×10^{-4}	3×10^{-3}
Subsistence	2×10^{-3}	2×10^{-2}	1×10^{-3}		
Noncarcinogenic (hazard index)					
Typical	0.02	0.08	0.001	0.02	0.08
Reasonable Maximum	0.2	0.5	0.02	0.2	0.5
Subsistence	1	4	0.08		

Source: Crane (1992b)

[a] Noncarcinogenic risks were averaged over the same period as the exposure duration, while carcinogenic risks were averaged over a period of 70 years (i.e., average lifetime of an individual).

pathway (e.g., consumption of fish, incidental ingestion of sediments). This summation of carcinogenic risks assumes that there are no synergistic or antagonistic chemical interactions and that all chemicals produce the same effect. USEPA believes it is prudent public health policy to consider actions to mitigate or minimize exposures to contaminants when estimated excess lifetime cancer risks exceed the 10^{-5} to 10^{-6} range (USEPA 1988a).

Noncarcinogenic effects are evaluated by calculating the ratio, otherwise known as the hazard quotient, of a site-specific exposure level for a specified time period to an RfD derived from a similar exposure period. Unlike cancer risk estimates, hazard quotients are not expressed as a probability. A hazard quotient of less than 1 indicates that exposures are not likely to be associated with adverse noncarcinogenic effects. As the hazard quotient approaches or exceeds 10, the likelihood of adverse effects is increased to the point where action to reduce human exposure should be considered (although the magnitude of the uncertainty factors used to derive the RfD should also be considered). Because of the uncertainties involved with these estimates, values between 1 and 10 may be of concern, particularly when additional significant risk factors are present. However, because RfDs do not have equal accuracy or precision and they are not based on the same severity of toxic effects, evaluation of hazard indices (the sum of two or more hazard quotient values for multiple substances and/or multiple exposure pathways) should take into account the uncertainties associated with specific RfDs.

The consumption of contaminated fish resulted in the greatest human health risk at the five priority AOCs examined in the ARCS Program: Saginaw River, Michigan (Crane 1992b); Sheboygan River, Wisconsin (Crane 1993a); Grand Calumet River, Indiana (Crane 1993b); Ashtabula River, Ohio (Crane 1992a); and Buffalo River, New York (Crane 1993c). Locally caught fish were assumed to accumulate contaminants primarily through the food chain, and in-place contaminated sediments were assumed to be the major source of contaminants to the food chain and water column. In most cases, PCB contamination contributed the greatest degree of carcinogenic risk. Noncarcinogenic risk levels were usually not of concern except for the subsistence exposure case; however, some chemicals (e.g., PCBs) lacked a verified RfD, and thus the noncarcinogenic effects of these contaminants could not be evaluated in the risk assessments. In addition, the consumption of bottom-feeding fish species, like carp, usually resulted in carcinogenic risks greater than 10^{-6}, whereas the consumption of water column-feeding fish species, like walleye, did not always result in significant carcinogenic risks.

Uncertainty Analysis

A number of assumptions and estimated values are used in baseline risk assessments that contribute to the level of uncertainty about possible human health risks. As with most environmental risk assessments, the uncertainty about the risk estimate is generally at least an order of magnitude or greater (USEPA 1989c). Thus, at a minimum, the risk assessment should include a qualitative uncertainty analysis that identifies the key site-related variables and assumptions that contribute most to the uncertainty inherent in the risk estimates.

Some of the major uncertainties in the ARCS risk assessments arose from the following factors:

- Use of contaminant burdens in fish based on uncooked fish, and in some instances, whole fish
- Exclusion of some complete exposure pathways (e.g., dermal exposure to water and sediments)
- Use of default exposure frequency and duration variables, body weight, life expectancy, and population characteristics
- Use of RfDs and CSFs that are usually based on animal studies and that may be based on only one form of a chemical (e.g., Aroclor® 1260 was used to derive the CSF for PCBs)
- Assuming additive health risks for both carcinogenic and noncarcinogenic effects
- Natural variability (e.g., small-scale spatial and temporal variability in sediment and hydrological conditions)
- Inherent approximations of physical, chemical, and biological processes in the models.

For each of these assumptions, the level of uncertainty associated with the final risk estimates was estimated as low, moderate, or high. Additional site-specific sources of uncertainty are likely to be important for risk assessments conducted at other contaminated sediment sites. Calibration and fine-tuning of model results after field testing can greatly reduce uncertainties associated with risk estimates at specific sites.

Applications

The results of the baseline risk assessment can be used by risk managers for several purposes. First, the baseline risk assessment provides a quantitative way to identify the exposure pathways and contaminants that contribute to carcinogenic and noncarcinogenic human health risks at a site. However, the calculated human health risks are not actual values, but are instead estimates that must be interpreted in the context of all the uncertainties associated with each step in the risk assessment process. Second, the baseline risk assessment can be used to identify sensitive subgroup populations (e.g., children, subsistence anglers) within the AOC. Third, the results of the baseline risk assessment can be used to compare the estimated risks of different sediment remedial alternatives with the impact of the no-action alternative during the comparative risk assessment. Additional applications of the baseline risk assessment are discussed in subsequent chapters.

ECOLOGICAL RISK ASSESSMENT

The wildlife and aquatic organisms of the Great Lakes region may be exposed to sediment contaminants through various mechanisms, resulting in ingestion, inhalation, or dermal uptake of potentially toxic chemicals. For the Great Lakes AOCs, ecological risk assessments may be conducted to evaluate the likelihood of acute and chronic adverse effects of sediment contaminants on wildlife, aquatic plants, benthic invertebrates, and fish that rely on lakes and streams for habitat, food, and drinking water. Under the ARCS Program, wildlife risk assessments were conducted for two of the priority AOCs, Buffalo River and Saginaw River.

Ecological assessments may involve empirical measurements of realized effects using the retrospective approach and theoretical modeling to estimate the probability of effects using the predictive approach. The balance of empirical and modeling approaches depends on the objectives of the assessment, the practicality of measurement methods for the receptors of concern, and data availability. For example, empirical approaches are commonly used to evaluate the effects of existing contamination on species populations and communities that are easily sampled, and on endpoints such as population abundances that can be easily quantified and interpreted. Theoretical models are used to estimate exposure of large-bodied wildlife species and rare species for which direct measurement of population or community endpoints is impractical, and to predict the effects of future conditions.

USEPA is currently developing guidelines for conducting ecological risk assessments (Norton et al. 1992) and recently issued their *Framework for Ecological Risk Assessment* (USEPA 1992b). This framework report describes the elements of an ecological risk assessment and provides the basis for conducting ecological risk assessments within USEPA. Development of specific guidelines for ecological risk assessment is in progress, but will require considerable time (Norton et al. 1992). USEPA Region V has issued its own framework document entitled *Regional Guidance for Conducting Ecological Assessments*. In addition, the State of Wisconsin recently issued their *Guidance for Assessing Ecological Impacts and Threats from Contaminated Sediments* (WDNR 1992a,b). In addition to the guidance offered herein, these other guidance documents should be consulted in planning an ecological risk assessment for contaminated sediments. Past ecological risk assessments that have been performed for contaminated sediment sites in Region V are available from USEPA Region V.

Because of the variety of habitats and species associated with sediments and interactions between biota and physical-chemical conditions, diverse techniques may be used in ecological risk assessments. The physical and chemical structure of an ecosystem influences the bioavailability and toxicity of contaminants to resident species. Biological interactions may determine the transport and fate of contaminants in the environment as well as species exposure patterns. Thus, the risk assessment process cannot be easily standardized to a "cookbook" approach.

The remainder of this chapter presents an overview of ecological risk assessment methods as they might be applied to contaminated sediments. The elements of the approach described below are consistent with USEPA's framework report and other USEPA reports (USEPA 1989d,e; Warren-Hicks et al. 1989). Other related activities in the development of ecological risk assessment guidance for aquatic habitats include: 1) formation of an Ecorisk Group within the USEPA Office of Water to develop a paradigm for ecological risk assessment, 2) development of sediment quality criteria by USEPA based on the equilibrium partitioning model (DiToro et al. 1991), 3) issuance of a review of aquatic risk assessment methods by Parkhurst et al. (1990) for the Water Environment Federation (formerly the Water Pollution Control Federation), and 4) USEPA's development of proposed methods for the derivation of ambient water quality criteria that would be protective of human health, wildlife, and aquatic organisms under the Great Lakes Initiative (40 CFR Parts 122, 123, 131, and 132; 58 Fed. Reg. 20802).

Because of the varied nature of contaminated sediment sites and the objectives of individual ecological risk assessments, only general guidance is offered here. Ecological risk assessors must still rely on their own judgment and expertise when evaluating potential risks to wildlife and/or aquatic organisms. Thus, any ecological risk assessment should include a clear statement of assumptions and an uncertainty analysis.

General Framework

The risk assessment process developed for estimating human health risks generally applies to determination of ecological risks. However, the complexity of ecological systems requires consideration of multiple species and other physical-chemical stressors in addition to toxic chemicals. Ecological endpoints may also differ from those used in human health risk assessment. For example, survival, growth, and reproduction may be emphasized as ecological endpoints, instead of cancer or more subtle sublethal effects. In ecological risk assessment, risks to populations, communities, and ecosystems are often considered more relevant than individual risk. Except in the case of rare, threatened, and endangered species, individual plants and animals are not highly valued because compensatory mechanisms in ecological systems may preclude higher-level effects even if individuals are eliminated from a population. In ecological risk assessment, the ability of the ecosystem to recover from the stress may also be considered.

USEPA (1992b) defines ecological risk assessment as "a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors." In general, both wildlife and aquatic risk assessments follow the basic framework shown in Figure 2-2 (USEPA 1992b):

- **Problem Formulation:** This planning and scoping step defines the objectives, approach, and data needs for the assessment. It includes: 1) a qualitative evaluation of contaminant release, transport, and fate; 2) identification of contaminants of concern, receptors, exposure pathways,

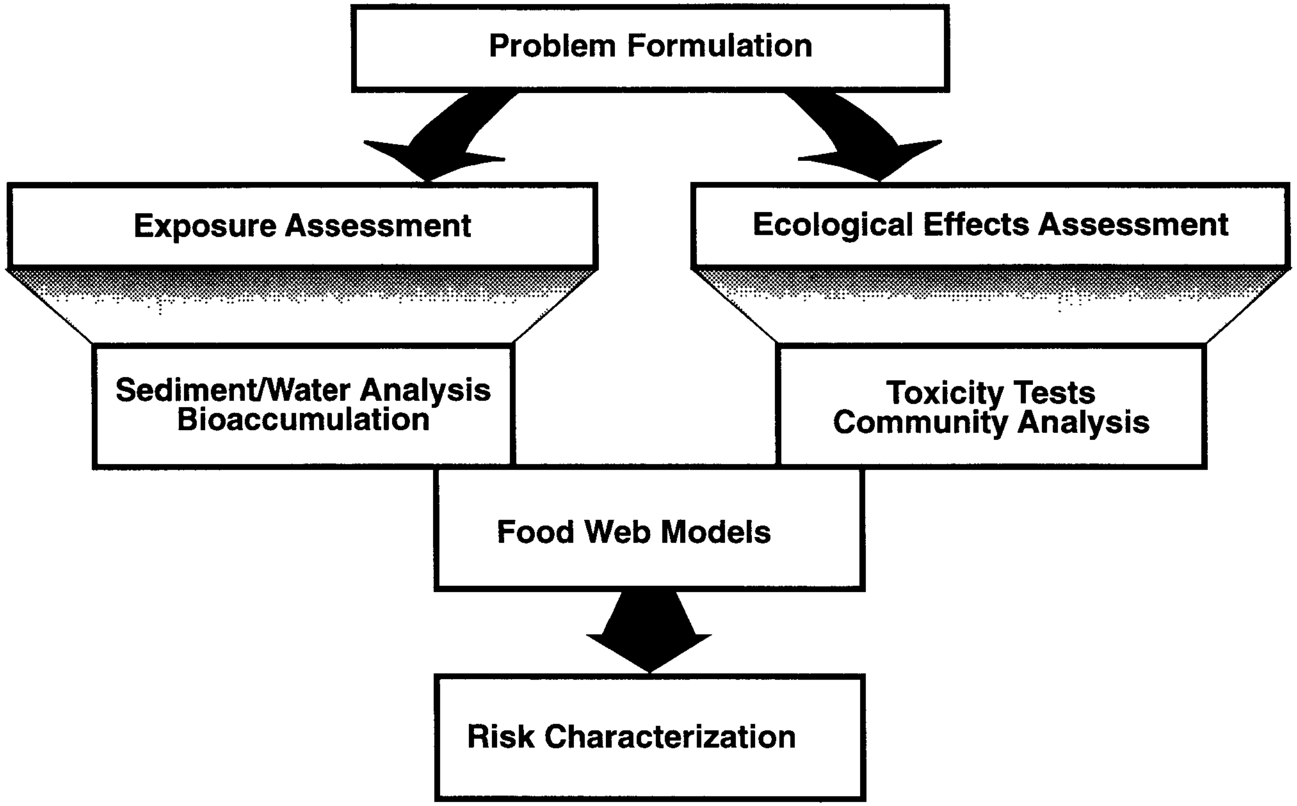

Figure 2-2. Ecological risk assessment framework.

and known ecological effects of the contaminants; 3) selection of endpoints for further study; and 4) integration of the preceding information into a conceptual model. Assessment endpoints should represent ecological "values" to be protected. Measurement endpoints are the observed or measured variables related to assessment endpoints. The lack of standardized ecological risk assessment procedures and the complexity of ecosystems make the initial planning of the assessment extremely important.

- **Exposure Assessment:** The exposure assessment uses chemical measurements and chemical transport and fate models to estimate the magnitude, duration, and frequency of exposure to the contaminants of concern. It involves the following steps: 1) quantification of contaminant release, transport, and fate including information on temporal and spatial variability; 2) characterization of exposure pathways and receptors; 3) measurement or estimation of exposure point concentrations (or chemical intake rates); and 4) evaluation of the quality of the data available for the exposure assessment.

- **Ecological Effects Assessment:** The ecological effects assessment determines the relationship between the levels of exposure and the levels and types of effects. It involves an evaluation of literature reviews, field studies, and toxicity tests that link contaminant concentrations to effects on ecological receptors. The effects assessment often uses models to extrapolate toxicity test data to different species, life stages, levels of biological organization, and exposure conditions.

- **Risk Characterization:** The risk characterization documents existing chemical effects and estimates the likelihood of adverse ecological effects by integrating the exposure and ecological effects assessments. It also provides narrative explanations of underlying assumptions, the nature and magnitude of uncertainties, and the quality of the data.

In ecological risk approaches, assessment endpoints are defined as environmental characteristics or values that are to be protected, such as wildlife population abundance, species diversity, or ecosystem productivity (Suter 1989). For example, maintenance or restoration of valuable natural resources is typically a goal of the remediation process. If protection of a valuable commercial fish stock is the goal, the assessment endpoint may be recruitment rate for the species population. Measurement endpoints are quantitative expressions of an observed or measured biological response related to the valued environmental characteristic chosen as the assessment endpoint. In some cases, the measurement endpoint is the same as the assessment endpoint. When these endpoints differ, a model must be used to express their relationship.

The process of estimating ecological risk based on chemical, toxicological, and ecological data is called a forward-mode assessment. Risk assessment procedures may also be used to back-calculate exposure guidelines from an allowable risk level or a no-observed-adverse-effect level (NOAEL). A reverse-mode assessment may be used to derive cleanup levels (e.g., maximum allowable concentrations of contaminants in sediment).

The level of detail required for a given risk assessment depends on remedial action objectives, the complexity of the site, and the difficulty in adequately describing exposure, toxicity, and other properties of the contaminants of concern. An ecological risk assessment can be conducted in tiers with the most basic analyses conducted first. For example, an initial screening-level risk assessment is conducted that uses available data and conservative assumptions about exposure and toxicity. From the results of this screening-level assessment, areas, contaminants, and species of concern are identified and decisions are made about additional data collection. In the next tier, more realistic models are used and additional data may be collected that will better define the relationship between chemical concentrations and adverse effects at the site.

Problem Formulation

The conceptual model developed during the planning phase of an ecological risk assessment illustrates how exposure to sediment contaminants may cause ecological effects. The results of the problem formulation stage clarify the scope of an ecological risk assessment and how the results will be used in developing RAPs for the AOCs. Based on the results of a screening-level assessment, chemicals, species, and endpoints are selected for a detailed assessment that may involve collection of additional field data and risk modeling.

Selection of Contaminants of Concern

Contaminants of concern are selected for the risk assessment based on available data and the preliminary evaluation of releases, transport, and fate of sediment contaminants relative to their potential toxicity. Sequential criteria for selection of contaminants of concern for an ecological risk assessment may include:

1. Detection in sediments within the AOC

2. Presence in sediments or tissues at concentrations significantly above reference concentrations

3. Relationship to human activities

4. Presence at concentrations above screening toxicity criteria.

The last step in the selection process is to compare measured or estimated environmental concentrations with threshold concentrations such as NOAELs. This analysis should be conservative by incorporating plausible worst-case assumptions regarding bioavailability, exposure, and sensitivity of ecological receptors.

The list of contaminants of concern for an ecological risk assessment may differ from that used for a human health risk assessment because of differences in exposure pathways, uptake, and sensitivities between humans and ecological receptors. Therefore, the initial list of contaminants considered for evaluation in the selection process described above should be comprehensive rather than simply the list selected for evaluation in the human health risk assessment.

Selection of Species

Because of the complexity of the food web in the Great Lakes basin, not all of the trophic levels and species can be evaluated. Thus, a few species or species groups may be selected as ecosystem indicators of environmental conditions. The Ecosystem Objectives Committee of the International Joint Commission developed the following criteria (as cited in Kubiak and Best [1991]) for selecting indicator species:

- Displays a broad distribution within the AOC
- Maintains itself through natural reproduction and is indigenous
- Interacts directly with many components of its ecosystem
- Maintains well-documented and quantifiable niche dimensions
- Exhibits a gradual response to a variety of human-induced stresses
- Responds to stresses in a manner that is both identifiable and quantifiable
- Represents an important species to humans.

These criteria for selection of indicator species were adopted for use in the ARCS Program. In addition, species selected for the assessment should be sensitive to effects of the contaminants of concern and, if possible, should be representative of a group of valuable species.

The U.S. Fish and Wildlife Service is evaluating the bald eagle, mink, otter, colonial waterbird group, and lake trout (salmonids) as possible indicator species for Great Lakes water quality (Kubiak and Best 1991). These species can also be used to evaluate the effects of contaminated sediments. In addition, other aquatic biota that should be considered in most risk assessments include benthic macroinvertebrates and bottom-feeding fish.

Any of the species or groups just discussed could be considered in a predictive assessment. For empirical assessments of existing conditions, the following ecosystem indicators are recommended:

- Bottom-feeding fish populations
- Higher tropic level fish (if AOC is very large)

- Benthic macroinvertebrate communities
- Locally important species of amphipods and chironomids.

For a specific risk assessment, the selection of species to be evaluated also depends on the contaminants of concern and the scale of resolution needed to define problem areas within AOCs. For example, assessment of bioconcentratable contaminants such as PCBs and dioxins over a wide portion of Lake Michigan might consider wide-ranging fish and wildlife species that feed at high trophic levels. Relatively fine-scale resolution of problem areas within a tributary might consider benthic macroinvertebrates and localized bottom-feeding fish species like bullhead. The selection of species and endpoints for an assessment should consider whether the contaminants biomagnify and whether they cause direct toxicity to receptors at lower trophic levels.

Practical methods for field and/or laboratory measurements must be available for retrospective assessments. Predictive and retrospective assessments both require adequate data on contaminant distributions or appropriate transport and fate models to estimate exposure. Limitations in data or models may influence the final selection of species for the assessment.

Selection of Endpoints

Both assessment endpoints and measurement endpoints are used as indicators of ecological risk. When the measurement endpoint differs from the assessment endpoint, a model must be used to express their interrelationship. The primary measurement endpoints for an ecological risk assessment should be related to the survival, growth, and reproduction of exposed organisms. These endpoints are used in most standardized toxicity tests and in the development of USEPA ambient water quality criteria, wildlife criteria, and sediment quality criteria. Moreover, such endpoints can be quantitatively related to changes in population numbers and structure. For example, PCBs are known to be accumulated in gull eggs in the Great Lakes region and have been linked to reproductive failure. Here, the measurement endpoint might be the proportion of nonviable gull eggs, as a predictor of effects at the population level. Although other endpoints, such as enzymatic responses and histological lesions in individual organisms, may indicate chemical exposure and response, they do not necessarily indicate adverse effects on populations, communities, or ecosystems.

Various endpoints may be used for predictive assessments, but their final selection is often affected by the availability of toxicity data in the literature and the quality of the data. Because prediction of community-level responses from survival, growth, and reproductive endpoints involves substantial uncertainties, wherever practical, effects on selected communities should be directly observed in the field. For example, population and community analyses of benthic macroinvertebrate communities may be used to evaluate toxic effects of sediment contamination.

Ecological Risk Assessment Objectives

The objectives of an ecological risk assessment are developed from conceptual models of chemical transport and fate, exposure pathways for selected receptors, and potential mechanisms for adverse ecological effects. The objectives should specify the selected contaminants, receptors, and endpoints to be included in the assessment. An example conceptual model of a site near a tributary might show the transport of contaminants of concern from soil to groundwater, which is then discharged to the river, where contaminants are absorbed or ingested by fish. Contaminants may be transferred to the terrestrial environment when a waterbird eats fish from an affected portion of the tributary.

Assessment techniques appropriate for the receptors and contaminants of concern and the level of complexity of the risk assessment are determined on a site-by-site basis. The selection of ecological assessment techniques to be applied at a site depends on the objectives of the risk assessment, site-specific receptor species and contaminants of concern, and the extent of available data. The primary techniques are:

- Chemical analysis of samples of sediment, surface water, and organism tissues from the site
- Toxicity testing of sediments
- Community analysis based on measurements of the types and number of benthic macroinvertebrates at the site
- Exposure models to predict chemical concentrations and bioavailability in environmental media and to estimate uptake by key receptors
- Ecological models to extrapolate from measurement endpoints to assessment endpoints in receptor groups for which community analysis is not a primary tool.

Each combination of tools selected for an AOC should provide adequate data for the assessment and facilitate risk predictions. Figure 2-3 summarizes some of the candidate tools for ecological risk assessments according to habitat, media, and receptors.

The problem formulation stage should include development of a strategy for integrating the results of individual assessment tools into the overall approach to risk characterization. Moreover, the risk assessment objectives should be clearly related to remedial action objectives and the decision-making process. The overall assessment strategy may involve both empirical and theoretical approaches.

Empirical approaches involve direct measurement of biological effects or derivation of relationships between chemical and biological variables from field data, or toxicity testing of field-collected samples. Empirical approaches rely heavily on observed relationships without attempting to describe theoretical cause-effect relationships. Warren-Hicks et al. (1989) describe empirical assessment approaches used to quantify the ecological effects

Tool (Habitat / Receptor)	Aquatic: Fish	Aquatic: Macroinvertebrates	Aquatic: Algae	Riparian and Upland: Plants	Riparian and Upland: Birds	Riparian and Upland: Small Animals	Riparian and Upland: Large Animals
Empirical Data							
Chemical Analyses							
Sediment/Water	●	●	●	●	●	●	●
Tissue	●	●	○	●	○	●	
Toxicity Tests	●	●	○	●	○	○	
Community Analysis	○	●	○	●	●	○	○
Models							
Exposure Models [a]	●			●	●	○	●
Ecological Models [b]	●					○	

a Includes transport and fate models and food web models to estimate exposure.

b Includes models to extrapolate measurement endpoints (e.g., organism-level effects) to assessment endpoints (e.g., population- or community-level effects).

● Primary tool – used at most sites

○ Secondary tool – used at selected sites based on specific conditions

Figure 2-3. Ecological assessment tools for contaminated sediments.

of contaminants at hazardous waste sites, including: 1) toxicity testing, 2) use of bio-markers, including analysis of tissue contaminants, and 3) community analysis based on field surveys.

Theoretical models are mainly derived from theoretical principles and include explicit mechanistic (cause-effect) relationships. Modeling may be used to support empirical risk assessments and to make risk predictions. Mathematical exposure models are used for dynamic systems (such as river water), for long-term predictions in more stable systems (such as sediment), and for transfer of chemicals through the food web to receptors higher up the food chain. Ecological models are used primarily to extrapolate from measurement endpoints to assessment endpoints in receptor groups such as amphibians, birds, and large mammals. Both exposure models and ecological models may vary from relatively simple extrapolation models with few data requirements to complex mechanistic models with substantial data requirements. Whenever practical, models should be based on site-specific data and validated.

Exposure Assessment

In the exposure assessment phase, measurements or estimates are made of the concentrations of contaminants of concern in the environment or the rate of chemical intake by organisms. Analysis of the magnitude, duration, and frequency of exposure is based on information or assumptions about:

- Chemical sources and pathways
- Chemical distributions in water, sediment, and organisms
- Spatial/temporal distributions of key receptors.

For empirical assessments, tissue concentrations of contaminants in key species may be measured as indicators of exposure. To develop estimates of exposure using models, exposure scenarios are developed from the conceptual site model to describe the pathways a chemical may take through various environmental media to reach an organism. For each site, the analysis of several exposure scenarios helps to identify data gaps for transport pathways and key exposure processes, such as chemical transformations or biological uptake. Data gaps for specific chemical forms or processes related to important pathways are filled through estimations from predictive chemical models or the collection of additional site-specific data.

The distributions and seasonal activity patterns of receptors are described relative to contaminant distributions in various habitats at a site. Habitats, concentrations of contaminants of concern, species distributions, and exposure variables related to species activities may be mapped and spatial patterns investigated using a mapping/database system such as a geographic information system.

Information from the transport and fate analysis for each exposure scenario is used to develop quantitative estimates of exposure that serve as inputs to the risk characterization. Summaries of data for the exposure assessment may include:

- Contaminant sources
  - Mapping of source locations
  - Contaminant release data for outfalls, landfills, combined sewer overflows (CSOs), and other sources
- Sediment
  - Mapping of contaminant distributions
  - Comparison of contaminant concentrations in sediments at the site with reference area values
  - Comparison of contaminant concentrations in sediments with levels associated with biological effects (based on available toxicological literature and field surveys)
  - Pattern analysis of contaminant data to evaluate potential sources of contamination
  - Evaluation of the suitability of the reference area
- Surface water
  - Comparison of concentrations of contaminants in water with USEPA ambient water quality criteria
  - Pattern analysis of contaminant data to evaluate potential sources of the contamination
  - Evaluation of the degree of chemical contamination in water collected from stations near the site relative to reference values
  - Evaluation of the suitability of the reference area
- Bioaccumulation
  - Evaluation of the degree of chemical contamination in fish tissue collected from the site relative to reference values
  - Evaluation of contaminant concentration gradients in tissue collected from the site
  - Evaluation of the suitability of the reference area.

Wildlife

The principal routes of chemical uptake for terrestrial wildlife are ingestion, inhalation, and dermal absorption. For riparian or terrestrial wildlife species, some routes of exposure, particularly ingestion, may involve many different media. Scientists from the U.S. Fish and Wildlife Service have conducted risk assessments for wildlife indicator species (e.g., bald eagle, other fish-eating birds, mink) for the ARCS Program at the Buffalo and Saginaw River AOCs. The approaches used by these researchers can generally be applied to other areas with sediments contaminated by bioconcentratable chemicals. Such chemicals are typically the primary contaminants of concern for assessment of sediment-associated risk to wildlife.

The exposure assessment for indicator species can be approached in two different ways (Kubiak and Best 1991). The first approach requires site-specific information or estimates of the types of forage items commonly eaten, contaminant concentrations in forage items, and grams of food eaten per kilogram of wildlife predator body weight. This information is used to calculate an ingested dose for contaminants of concern in the indicator species, similar to food web model approaches currently being used for sediments and other media at hazardous waste sites (e.g., Fordham and Reagan 1991; Menzie et al. 1992). The calculated ingested dose is then compared to a NOAEL developed from a model feeding study where known adverse toxicological endpoints were measured. There is considered to be a significant risk to the indicator species if the calculated ingested dose exceeds the NOAEL. Unfortunately, the database necessary for applying this approach is not sufficiently developed for Great Lakes species.

The second approach can be conducted for contaminants that bioaccumulate through the food chain (e.g., DDE, PCBs, dieldrin). In this approach, the concentration of a contaminant of interest is measured in a specific tissue of the indicator species. Because the exposure estimate is expressed as a contaminant concentration in tissue, the ecological effects assessment includes an estimate of the NOAEL expressed as a contaminant concentration in the tissue rather than as an ingested dose. The measured tissue contaminant concentration is compared with the NOAEL to evaluate existing risk. Based on the NOAEL and on biomagnification factors calculated from actual field data, Kubiak and Best (1991) also applied this approach to backcalculate the contaminant concentrations in forage fish that would be necessary to result in a contaminant concentration equal to the NOAEL in the tissue of the indicator species.

Either approach, appropriately used, could be applied to the calculation of "safe concentrations" or remedial action goals for sediments. The goal would be to determine the contaminant concentrations in sediments that would not result in exceedances of the NOAELs, expressed as an ingested dose to the indicator species in the first approach or as a tissue contaminant concentration in the second approach. For the ARCS Program, the second approach was applied for bald eagle, other fish-eating birds, and mink in the wildlife risk assessments conducted for the Buffalo River AOC (Mann-Klager, in prep.) and the Saginaw River AOC (Kubiak, in prep.).

Aquatic Life

The principal routes of chemical uptake for aquatic organisms are gill and body-surface absorption of chemicals in surface water or sediment pore water, and ingestion of water, sediments, and food. Exposure estimates for aquatic life may be based on one or more of the following:

- Contaminant concentration in bulk sediment
- Contaminant concentration in sediment pore water
- Contaminant concentration in the water column
- Contaminant concentration in organism tissue.

In most cases, the exposure estimate will be expressed as the contaminant concentration in the bulk sediment, pore water, or water column, which is obtained from empirical measurements. Few toxicity data are available to interpret exposure estimates expressed as contaminant concentrations in tissue (e.g., Dillon 1984).

Ecological Effects Assessment

Ecological effects assessment includes a hazard identification step to identify the potential effects of chemicals and an exposure-response assessment to characterize the relationship between each stressor and the biological or ecological endpoints. Confounding effects of physical stressors such as currents or sediment grain size must be addressed by using models, reference-area measurements, or experimental designs to separate the effects of physical factors from those of chemicals.

Techniques for ecological effects assessment may include the following:

- Laboratory/field toxicity tests
- Observational field studies
- Interspecies extrapolation of effects
- Interchemical extrapolation based on knowledge of their modes of action, such as quantitative structure-activity relationships
- Biological or ecological modeling to extrapolate from measurement endpoints to assessment endpoints.

Data from direct field measurements or from laboratory analyses of field samples should be used whenever possible to derive exposure-response relationships. Site-specific properties of sediment and water may modify the bioavailability of chemicals, and literature data may not be appropriate for ecological effects assessment.

Wildlife

Although site-specific toxicity data are preferred for exposure-response assessment, use of available data from the general literature may be appropriate in certain cases. For example, literature estimates of stress thresholds such as NOAELs for wildlife species might be used with conservative assumptions about bioavailability in a screening-level assessment.

An estimate of the NOAEL for each contaminant of concern can be derived from the literature or from a model feeding study on a surrogate laboratory species. NOAELs can be expressed as chemical intake rates, which typically result from ingestion of chemicals, or chemical concentrations in tissue. The units for the NOAEL must correspond to the units used for the exposure estimate. When developing NOAELs, the following guidelines developed by PTI (1992) are relevant:

- Data from the receptor species of concern or a representative surrogate species that is closely related to the receptor should be used to derive NOAELs

- Ecologically relevant endpoints should be selected or a quantitative uncertainty analysis, which delimits the probable range of the NOAELs, should be performed when endpoint extrapolations are required

- The mode of administration of chemicals in laboratory exposures must be evaluated, and inter-route extrapolations should be avoided if possible.

An endpoint for a relatively sensitive tissue, organ, or life stage should be determined to derive a conservative estimate of a NOAEL. Adverse effects on reproductive organs and early life stages are typically good endpoints for risk assessment because they are likely relevant to changes at the population level. For birds, the most sensitive stages are the egg and the developing embryo, at least for chlorinated organic compounds and methylmercury. In contrast, the liver is the most sensitive tissue known for mink (Kubiak and Best 1991).

Aquatic Life

Ecological effects assessments for aquatic life may be based on theoretical approaches such as the use of sediment criteria developed from equilibrium partitioning models (e.g., DiToro et al. 1991) and bioaccumulation models (e.g., Thomann et al. 1992). An ecological epidemiological approach is recommended to enhance sediment risk assessments by taking into account factors other than source information. In this empirical approach, summaries of data for the aquatic ecological effects assessment may include:

- Sediment toxicity

  - Estimation of mean and variance of percent response for each toxicity test endpoint at each station

    - Mapping of bioassay response data

    - Correlations of bioassay response with contaminant concentrations and ancillary variables to evaluate potential cause-effect relationships

    - Evaluation of the suitability of the reference area (if available) and negative controls

    - Pairwise statistical comparison of mean percent response at each potentially contaminated site with the reference area (if available) or negative control responses

- Benthic macroinvertebrate communities

    - Estimation of mean and variance of taxon abundances or community indices at each station

    - Mapping of benthic invertebrate data

    - Correlations of community parameters with contaminant concentrations and ancillary variables to evaluate potential cause-effect relationships

    - Evaluation of the suitability of the reference area (if available)

    - Pairwise statistical comparison of community parameters at each potentially contaminated site with the reference area value (if available)

- Fish histopathology

    - Correlations of the prevalence of tumors and other abnormalities with contaminant concentrations and ancillary variables to evaluate potential cause-effect relationships

    - Evaluation of the suitability of the reference area (if available)

    - Pairwise statistical comparison of the prevalence of tumors and other abnormalities at each potentially contaminated site with the reference area value (if available).

Guidance on the use of toxicity tests, benthic macroinvertebrate community surveys, and fish histopathology investigations in support of ecological risk assessments is provided in the *ARCS Assessment Guidance Document* (USEPA 1993).

Risk Characterization

In risk characterization, the exposure and ecological effects assessments are combined to estimate the probability of adverse ecological effects. Final risk estimates may be expressed in simple narrative terms or as quantitative values. The risk characterization should summarize:

- Results of the exposure and ecological effects assessments
- Risk estimates for aquatic and wildlife receptors of concern
- Potential ecological consequences
- Major sources of uncertainty.

Mitigating factors, such as reduced bioavailability of contaminants incorporated into sediment particles and mechanisms for possible wildlife avoidance of hot spots, should be discussed. The importance of mitigating factors should be confirmed by field measurements or laboratory experiments on samples from the site. Also, compensatory mechanisms that preclude population- or community-level effects should be acknowledged, even though effects on individuals may be predicted.

Approaches to develop quantitative risk estimates (or hazard indices) include the quotient method, joint probability analysis, model uncertainty analysis, and integrated analysis of site-specific empirical data (Barnthouse et al. 1986; Suter et al. 1992; Ginn and Pastorok 1992; Chapman et al. 1992). The quotient method uses a ratio of the value of an endpoint at the site to a toxicity reference value such as a NOAEL as an approximate risk index. The quotient method is useful mainly for screening-level analyses because it does not provide a complete characterization of the magnitude of risk and uncertainties (Bascietto et al. 1990). Joint probability analysis (Barnthouse et al. 1986) can be applied to estimate the risk that exposure exceeds toxicity thresholds or criteria where probability distributions are available for the variables being compared. Model uncertainty analysis (Barnthouse et al. 1986; Suter et al. 1992) may be used to develop risk estimates for a species based on statistical analysis of growth, survival, and reproduction of individuals. Approximate risk estimates may be derived for an ecological system associated with sediments by combining site-specific data for chemicals, toxicity tests, community indices, and possibly other risk indicators (Chapman et al. 1992).

Wildlife

The quotient method and joint probability analysis will likely continue to be the primary methods for expressing estimates of risk to wildlife receptors. An estimated chemical intake by a wildlife receptor may be compared with a NOAEL. In interpreting hazard quotients, Kubiak and Best (1991) propose that to be ecologically protective, the ratio of the exposure to the NOAEL should be less than 1, because this provides a reasonable level of assurance that adverse effects would not occur as a result of excessive contaminant exposures. However, hazard quotients must be interpreted relative to the assumptions on which the assessment is based, the assessment endpoints, and the degree of confidence in the relationship between the assessment endpoints and the measurement endpoint used in the hazard quotient. For example, for reasonable maximum exposure scenarios, hazard quotient values between 1 and 10 do not necessarily indicate a significant risk. For most exposure scenarios, hazard quotients of greater than 10 are generally considered to represent a significant ecological risk. Prior to interpreting any hazard quotient, agreement must be reached on the toxicity value analyzed (e.g., dose of a substance that results in 50-percent mortality in a population of test organisms [LD_{50}], lowest-observed-adverse-effect level [LOAEL], NOAEL, reference dose).

The joint action of contaminants should also be considered. Hazard quotients for individual contaminants with similar modes of action may be summed to yield a hazard index.

Aquatic Life

The quotient method, joint probability analysis, and site-specific integrated data analysis will likely be the approaches that are commonly applied to assess risks to aquatic life associated with contaminated sediments. Use of the quotient method may involve comparison of a measured assessment endpoint to a threshold value considered indicative of toxicity or to a value indicative of reference area conditions.

Uncertainty Analysis

Possible sources of uncertainty include natural variation, missing information, and errors associated with measurements, extrapolations of data, or models. Uncertainties may be related to selection of contaminants of concern, selection of species, estimates of exposure concentrations or doses, the quality of the toxicological data used for NOAELs or LOAELs, or differences in exposure-response relationships or bioaccumulation of chemicals among species. The most important sources of uncertainty identified in the exposure and ecological effects assessments should be evaluated and quantified to the extent possible. Model uncertainty analysis may include sensitivity analysis and Monte Carlo simulation.

The baseline aquatic risk assessments for the ARCS Program were designed to complement the baseline human health and wildlife risk assessments so that the exposure pathways leading from sediments to fish to humans and wildlife could be quantified. However, these aquatic risk assessments were difficult to perform for different trophic levels of the aquatic food chain because of data gaps. Conservative assumptions and published data from other studies were used to fill missing information. The problem of data gaps affects most aquatic and wildlife risk assessments and constitutes a major source of uncertainty in the risk estimates.

Uncertainty analysis should be conducted during the problem formulation stage to identify data gaps and plan the approach for the ecological risk assessment. When significant data gaps exist, the assessment should typically include several tiers of analysis, with use of available data in the early screening tier to help define critical data needs to be addressed in further field sampling or laboratory studies and subsequent analyses in the next tier.

3. MASS BALANCE MODELING APPROACH FOR ASSESSING REMEDIAL ALTERNATIVES AT CONTAMINATED SEDIMENT SITES

Mass balance modeling studies were conducted at the Buffalo and Saginaw River AOCs to demonstrate the use of this approach in evaluating remedial alternatives for contaminated sediments. Mass balance modeling studies were applied at these AOCs to estimate changes in contaminant concentrations in water, sediments, and biota that may occur following sediment remediation. The estimated concentrations can then be used to compute changes in human health and ecological risks and to aid in the selection of remedial alternatives. The mass balance modeling studies, as described below, are based on established models and methods and are considered applicable to other sites with contaminated sediments.

OVERVIEW

While there are many possible remedial alternatives for contaminated sediments, only a few may be feasible at a particular site. After the range of feasible remedial alternatives is identified for a site, the potential for reduction of contaminant concentrations in water, sediments, and biota must be considered when selecting a remedial alternative. This selection process requires some method to estimate changes in contaminant concentrations that would result from each remedial alternative. The preferred approach for estimating these changes, and the approach used in the ARCS RAM studies, is based on the application of mass balance models.

Mass balance models attempt to describe each of the underlying mechanisms causing change in the system, and are termed mechanistic. In the mass balance modeling approach, the law of conservation of mass is applied in the evaluation of the sources, transport, and fate of contaminants. This approach requires that the quantities of contaminants entering the system (i.e., contaminant loading) equal the quantities leaving the system, less the quantities stored, transformed, or degraded. Thus, the mass balance is simply a bookkeeping of all of the processes affecting the mass of contaminants in a system. After the mass balance has been established for each contaminant of concern, quantitative changes in contaminant concentrations can be estimated. For example, the mass balance can be used to estimate the change that may be expected following removal of some portion of the contaminant mass.

Mass balance models have been successfully applied to the Great Lakes and elsewhere in the regulation of toxic and conventional pollutants. Properly applied, a mass balance model can serve as a surrogate for the natural system that is easily manipulated to estimate the system's response to change.

Management questions that can be addressed with a mass balance model include:

1. What are the consequences of leaving the contaminated sediments in place (the no-action alternative) under present conditions or where contaminant loadings are reduced?

    a. Is the system in equilibrium with present loadings? For example, will sediment contaminant concentrations increase or decrease over time or remain the same under present loading conditions?

    b. What are the relative contributions of loadings from point and nonpoint sources? What are the major loss mechanisms (e.g., outflow, burial)?

    c. What is the effect of changes in loadings? For example, if loadings are reduced or eliminated, what is the effect on contaminant concentrations in the water, sediments, and biota?

    d. How long does it take for the system to respond to changes in loadings? For example, if the loads are reduced, how long would it take for the concentrations to reach acceptable levels in water, sediments, and biota?

    e. If left in place, will contaminated sediments pose a threat to downstream areas or will they become more widely dispersed?

2. What are the consequences of alternative remedial/mitigative actions, such as removing, immobilizing, or treating the contaminated sediments?

    a. What are the expected benefits of alternative remedial/mitigative actions in terms of contaminant reductions in water, sediment, and biota?

    b. What is the probability of recontamination following remedial/mitigative actions under present loading conditions? How long would recontamination take?

    c. What risks are associated with implementation of alternative remedial/mitigative actions?

3. What are the estimated loadings from the AOC to downstream water bodies? For example, the Buffalo River may serve as a source of contaminants to Lake Ontario, whereas contaminants from the Saginaw River may be transported to Saginaw Bay. What changes can be expected in these loadings following the implementation of selected remedial/mitigative actions?

COMPLEXITY OF THE MASS BALANCE MODELING STUDY

Mass balance modeling studies vary widely in their complexity. The modeling approach can vary from simple screening calculations to applications of complex computer programs. Modeling studies also vary in the complexity of the field studies used to support the mass balance calculations, from studies relying solely on historical data to large and expensive field efforts. The degree of complexity required depends on the physics of the system, factors affecting the transport and transformations specific to the contaminants of concern, and the management questions the mass balance modeling study will address. The degree of complexity used in particular studies is often dictated by the time and funding available.

Modeling studies of contaminants in the Great Lakes have typically been cataloged into groups or study levels depending on the level of effort and complexity in the modeling and supporting field studies. Studies have been categorized as either "screening" or "detailed" studies, as well as "Level 0" through "Level III" studies. The study levels can generally be described as:

- **Level 0**—Application of simple manual or graphical methods based on statistical or deterministic equations to obtain rough estimates of contaminant concentrations over extensive areas or to identify trouble spots for more detailed analyses. These analyses rely solely on available data to obtain a preliminary assessment of management options and to identify deficiencies in the database when planning more detailed evaluations.
- **Level I**—Application of simple computerized models to obtain rough estimates of contaminant concentrations over extended periods of time. Model equations are generally mechanistic in nature but only approximate the basic processes. As a result, model projections used to address the management questions involve considerable uncertainty. Data collection is usually limited to one preliminary data collection study. Qualitative estimates are usually based on experience in interpreting the results. A formal uncertainty analysis is generally not included.
- **Level II**—Application of a computerized model of intermediate complexity as a planning model, or as a rough engineering design or resource management model. Extensive areas and periods of time can be simulated but at significant cost in data collection and preparation. Data collection involves acquisition of at least two independent data sets to bracket important

environmental conditions (e.g., high-flow vs. low-flow). All contaminant loadings to the system must be well characterized. Some simplifications and approximations limit the applicability of the model for remedial design and management. Thus, uncertainty analyses are generally included as part of the model application.

- Level III—Application of advanced mechanistic computerized models for detailed remedial design and management. The modeling approach would typically include descriptions of the hydrodynamics and sediment transport in the system, as well as detailed computerized models of water quality processes. Data collection involves at least two surveys to provide input for both model calibration and model evaluation. The surveys may be coupled with data collection over longer periods to establish trends and the range of environmental conditions.

Generally, the level of uncertainty is reduced as the studies increase in complexity from Level 0 to Level III, while the time and costs associated with each study level increase.

Increases in modeling complexity need not always correspond to increases in data collection, so various combinations of the levels mentioned above are possible. For example, simple models usually predict average conditions, so sufficient data must be collected to provide an accurate estimate of the average condition for comparison with predictions. However, more complex models may predict conditions at a specific point (in time and space), thus reducing the need for averaging.

In the ARCS RAM studies, both near-field and far-field modeling studies were conducted. The near-field modeling studies concentrated on the lower portions of the Buffalo and Saginaw Rivers where sediment contamination was the greatest and where remedial actions will likely be implemented. The far-field modeling studies were used to estimate the impact of remediation of the Buffalo and Saginaw Rivers on their receiving waters—Lake Ontario and Saginaw Bay, respectively.

While the general mass balance modeling approaches used in the near-field and far-field studies were similar, they differed in the resolution of the models applied and in the level of supporting field studies. The ARCS near-field studies were categorized as Level I or a "Mini-Mass Balance" modeling effort. The primary criterion restricting the modeling studies to Level I was the limited level of supporting field studies.

The far-field modeling was more typical of Level 0, in terms of data collection, and Level I, in terms of modeling. These studies were designed to estimate the long-term impact of changes in loadings from the rivers on their respective receiving water bodies. As such, hydrodynamics and sediment transport were described rather than predicted, and the resolution was not as precise as in the near-field studies. The far-field modeling studies of Lake Ontario and Saginaw Bay relied exclusively on historical data.

COMPONENTS OF THE MASS BALANCE MODELING STUDY

The application of the mass balance modeling approach involves the quantification of the sources, transport, and fate of contaminants. The components of the modeling approach are illustrated in Figure 3-1. The typical steps in a mass balance modeling study are: 1) predict water and sediment transport, 2) use the predicted water and sediment transport, along with estimates of contaminant loadings from point and nonpoint sources, to estimate the changes in chemical concentrations in water and sediments, and 3) use the predicted contaminant concentrations in water and sediments to estimate the transfer of contaminants through the food chain and their accumulation in fish. The models used for each step of this process are described below.

Water Transport Models

The first step in characterizing the transport of dissolved contaminants is to characterize the transport of the water, or its motion. Often, much of the variability of contaminant concentrations in the water column can be explained by water transport alone. Water transport models may be descriptive (i.e., based on a balance of the water's mass) or hydrodynamic (i.e., based on a balance of the water's momentum).

Characterization of water transport may be qualitative or quantitative, depending on the study level. In a qualitative approach, flow patterns are either measured directly or inferred from measurements of related parameters. The qualitative approach is often adequate where the system is very simple (hydraulically) or where only long-term, relatively rough estimates of water transport are required. A qualitative approach is most often used in Level 0 and Level I studies and is occasionally used in Level II studies.

Hydrodynamic models are used to quantitatively predict changes in volumes, depths, and velocities in response to changes in flow or water surface elevations. Hydrodynamic models require data on boundary conditions (i.e., flows or water surface elevations, wind speed and direction), which are applied to predict the resulting flows within the modeled system. Flows are often measured at gaging stations on many Great Lakes tributaries, and water surface elevations are routinely measured at many locations within the Great Lakes. Where such direct measurements of flows in the tributaries of the Great Lakes are problematical because of the complex interactions and relationships between upstream inflows and lake effects, hydrodynamic models can be used to quantitatively predict flows. Hydrodynamic models can also be used to estimate changes in flows that may occur under future conditions, such as in evaluating the effects of changes in dredging patterns.

The hydrodynamic models used in the ARCS RAM modeling studies included HYDRO-3D, a 3-dimensional hydrodynamic and transport model, and RIVMOD, a 1-dimensional model. Both models are maintained and distributed by the Center for Exposure Assessment Modeling at Athens, Georgia. Multidimensional models were required where it was necessary to resolve variations in water transport and chemical

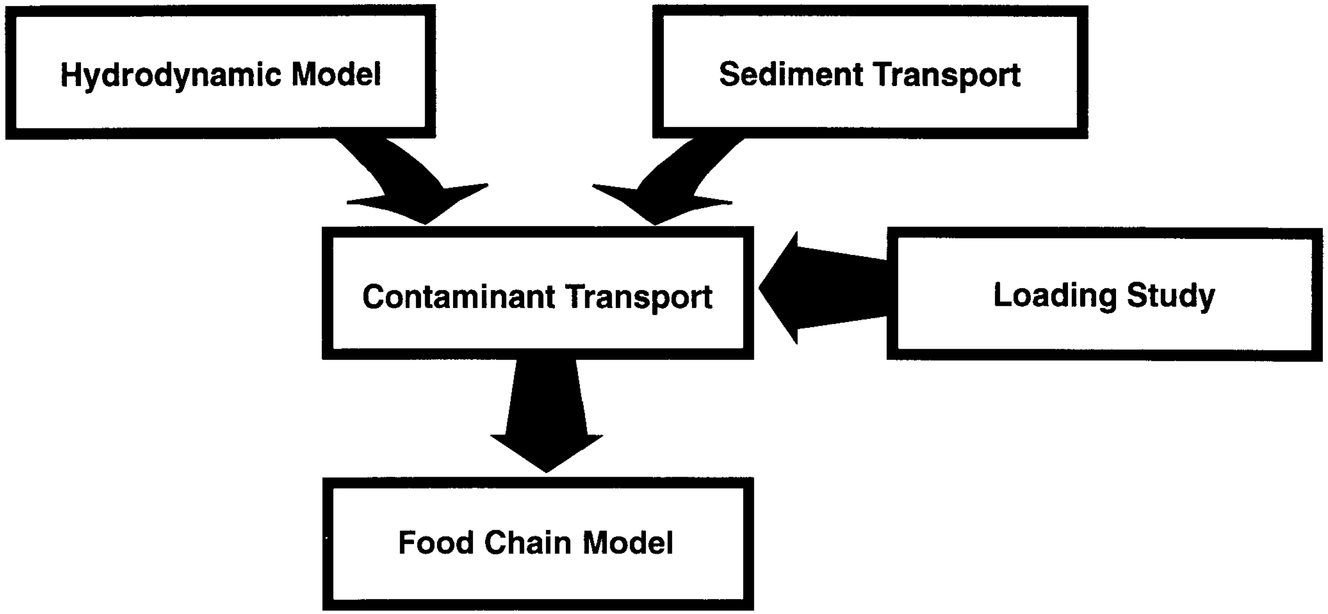

Figure 3-1. Components of the mass balance modeling study.

characteristics along the longitudinal, vertical, and lateral axes of the rivers. For example, a model that was capable of estimating variations in transport with depth was required in the Buffalo River because of water column stratification during low-flow periods.

A qualitative approach was taken for the far-field studies. For Lake Ontario, historical measurements of inflows and outflows were used to compute water transport, while for Saginaw Bay, water transport was computed from measured flows and chemical data.

Sediment Transport Models

Because contaminant concentrations in surface sediments are a primary concern in the AOCs, adequate characterization of the movement of surface sediments is a critical step in the mass balance modeling process. There are two primary goals of the sediment transport model: 1) to predict the movement of the sediments to estimate changes that may occur in patterns of erosion, deposition, and transport, and 2) to estimate the transport of the particulate contaminant mass.

Sediment transport models are based on a sediment mass balance. As with water transport, sediment transport may be described either qualitatively or quantitatively in mass balance modeling studies.

Qualitative approaches have commonly been used in Level 0 and Level I studies. In studies of this type, measured or estimated settling, resuspension, and sedimentation rates are used to compute the transport of sediments and their associated contaminants. Although it is generally assumed that there is no net sedimentation or erosion, the different contaminant concentrations on suspended and settled particulate matter make the quantification of these processes important. The qualitative approach has proven useful in providing rough estimates of the effects of sediment transport on contaminant distributions. However, sediment transport is a very dynamic process and the assumption of no net sedimentation or erosion is a gross simplification.

In more quantitative sediment transport models, resuspension and transport are computed using the output of a hydrodynamic model and the measured characteristics of the sediments. The sediment types of primary importance are silts and clays, rather than sands. Silts and clays are classified as cohesive sediments, while sand is classified as a noncohesive sediment. The sediment transport model is used to predict changes in sediment resuspension, deposition, and transport and the effect of these processes on particulate concentrations in the water column. The sediment transport model is also used to estimate changes in the structure of the sediment bed, such as the impact of erosion or deposition on the channel shape and sediment composition.

As with hydrodynamic models, sediment transport models can be used to interpolate between existing measurements or estimate sediment transport for conditions for which data are not available. The majority of sediment transport occurs during extreme (rare) events, such as storms on lakes and large runoff events in rivers. Because data are rarely available for these events, sediment transport models must be used to estimate transport under these conditions. For example, these models may be used to estimate whether contaminated sediments may be buried or exposed by a 10-year or 100-year flood. This information can be used in the evaluation of remedial alternatives as well as the no-action alternative. Sediment transport models may also be used to evaluate the impact of removing or immobilizing sediments on subsequent erosion and deposition patterns. For example, if sediments are removed from a particular area, sediment transport models may be used to estimate how long it may take for the area to fill in, as well as changes that may occur in deposition and erosion areas.

In the ARCS RAM modeling studies, a model of the transport of cohesive sediments was applied to both of the near-field study sites (i.e., Buffalo and Saginaw Rivers) to predict the interactions between the transport, deposition, and resuspension of sediments under various meteorological and hydrological conditions. The model applied was a 2-dimensional (longitudinal and lateral), time-variable, hydrodynamic and sediment transport model developed by Wilbert Lick at the University of California, Santa Barbara. This model has been applied at various locations around the Great Lakes, including the Detroit River, Fox River, Green Bay, Lake Erie, and elsewhere. The sediment transport model was coupled with a 3-dimensional, time-variable model of the sediment bed and its properties, the Water Quality Analysis Program, WASP4 (Ambrose et al. 1990). WASP4 integrates predictions from the hydrodynamic and sediment transport model to estimate contaminant concentrations in the water and sediment. The WASP4 model, maintained and distributed by the Center for Exposure Assessment Modeling at Athens, Georgia, has been widely distributed. WASP4 is currently the framework used for modeling studies in Green Bay, Lake Michigan, Lake Ontario, and elsewhere around the Great Lakes. It is generally assumed that the water column is completely mixed vertically because sediment resuspension typically occurs at flows where this would be the case. This model was used to predict variations in suspended solids concentrations, resuspension and deposition rates, and variations in the sediment bed as a function of flows and loadings. The model provided predictions for use in determining the transport of sorbed contaminants and resuspension of toxic sediments. Application of the sediment transport model was of particular importance in these studies because of the lack of historical sediment data. Therefore, these studies relied heavily on the sediment transport model to supply estimates of sediment transport, resuspension, and deposition.

Contaminant Exposure Model

The contaminant exposure model is the mass balance model for contaminants. The contaminant exposure model used as a framework for the ARCS RAM mass balance modeling studies was WASP4. In the application of the contaminant exposure model, the rate of change in contaminant mass (accumulation) is a function of the transport of a contaminant into, out of, and within the system via water transport or sediment transport for those materials that sorb to sediments; the mass added to the system via point and nonpoint loadings minus the outputs; and the quantities transformed and degraded within the system via processes such as volatilization, biodegradation, and photodegradation. The output expected from the contaminant exposure model includes estimated contaminant concentrations of both particulate and dissolved forms in water and sediments, as well as estimated changes in mass fluxes due to inflows and loadings, outflows, and degradation and transformation processes. Depending on the level of complexity, the transport via water and sediments may be described or predicted using hydrodynamic and sediment transport models, which are then coupled with the contaminant exposure model.

In the ARCS RAM near-field studies, WASP4 was applied to predict the effects of water transport, sediment transport, sorption, and transformation processes on the concentrations of selected contaminants of concern. The WASP4 model was linked to the output of the hydrodynamic and sediment transport models, which together provided the necessary transport information, using data collected during ARCS field studies. The output of the contaminant exposure model included temporally and spatially varying estimates of contaminant concentrations in water (both particulate and dissolved) and sediments for comparison to field data and for projections of the effectiveness of various remedial alternatives. In addition, the output included estimates of the magnitude of all processes that result in gains or losses of contaminants, so that their relative importance could be evaluated. The contaminant exposure model also provided information that could be incorporated into the food chain model to estimate the contaminant body burdens in fish due to varying contaminant concentrations in water and sediment.

The same contaminant exposure model used in the near-field studies (WASP4) was applied in the far-field studies. However, in the far-field studies the resolution was not as precise as in the near-field studies. Therefore, there was no need to apply the hydrodynamic and sediment transport models in the far-field studies. WASP4 was applied to predict steady-state (long-term average) conditions over large spatial scales. The output of the model included load-response relationships. For example, given a particular contaminant loading, the model predicted the average contaminant concentrations in water [both particulate and dissolved] and sediments. This relationship was then used to evaluate the impact of changes in loadings from the two AOCs on Lake Ontario and Saginaw Bay.

Food Chain Model

The food chain model is a mass balance model for contaminants where the rate of change in contaminant mass in each component of the food chain is a function of the transport of a contaminant into and out of that component (e.g., via ingestion, gill exchange, excretion), as well as internal changes that may occur due to growth. The food chain model supports evaluation of the impact of various remedial alternatives including the no-action alternative on contaminant concentrations within the food chain, given variations in contaminant concentrations in water and sediments derived from the contaminant exposure model. Outputs from the food chain model include time-variable estimates of contaminant concentrations in each component of the food chain.

The food chain model applied to both the Buffalo and Saginaw Rivers as part of the ARCS RAM studies was the Water Quality Analysis Simulation for Toxics, WASTOX (USEPA 1985), a predecessor of WASP4 that includes a food chain component. In the Buffalo River, the only fish species simulated was carp, because it is a bottom-feeder with a high fat content and would be expected to have an appreciable contaminant body burden. In the Saginaw River, the food chain model included forage fish and walleye, the latter due to its high importance to recreational anglers. Data collected as part of the ARCS RAM studies were used to construct a simple food chain model.

REQUIRED FIELD DATA

Field data are required to apply any mass balance model. For model application, field data are required to define the characteristics of the site and the mass fluxes of water, sediment, and the associated contaminants into and out of the site. In addition, data are often required to estimate the site-specific values of model parameters and to assess the uncertainty associated with model projections. The confidence that can be placed on those projections is dependent upon both the integrity of the model and how well the model is calibrated to that particular water body.

While models can be run with minimal data, the resulting predictions are subject to large uncertainty. Models are best used to interpolate between existing conditions but may be used to extrapolate from existing to future conditions, such as in the evaluation of the effects of remedial or mitigative alternatives.

The types of data required and the necessary frequency of measurement vary with the level of model complexity, the characteristics of the system, and the contaminants of concern. Generally, three kinds of data are required: 1) boundary conditions, 2) initial conditions, and 3) data for calibration/evaluation. Boundary conditions are external to the model (i.e., the model does not predict them). Instead, they are used to "drive" the model. Initial conditions are used to aid in the design of the model application (e.g., to determine segmentation) and to provide a starting point for model predictions. For example, the initial, or existing, sediment contaminant concentrations are required to provide a starting point for model predictions. Temperature data may be used to determine the need for predicting water-column stratification and its effects on contaminant transport. Model calibration/evaluation data include the parameters that the model is designed to predict (e.g., contaminant concentrations and fluxes). These data are compared to the model's predictions to aid in determining the values of site-specific parameters and in estimating the uncertainty associated with those predictions. The types of data required for each component of a mass balance modeling study are described below.

Water Transport Data

Boundary condition data for the water transport models include the system's bathymetry, point source and nonpoint source flows and loadings, upstream and downstream flows and/or water surface elevations, and weather data (e.g., wind speed and direction, temperature). These data can often be obtained from the National Oceanic and Atmospheric Administration, U.S. Army Corps of Engineers (Corps), USEPA, USGS, and local government agencies. Both historical data and data collected during the course of field studies of the AOC are required. Model calibration/evaluation data may include measurements of flows, velocities, or water surface elevations within the system under investigation for comparison to model predictions.

The frequency of measurement of hydrodynamic parameters varies with the complexity of the system under investigation. However, for most Great Lakes rivers and harbors, their dynamic nature requires that data be available on an hourly basis for major tributary flows, water surface elevations, and weather data. Water surface elevation and weather data are usually readily available at this frequency.

In the ARCS RAM modeling studies, historical data were available on flows, water surface elevations at the mouths of the Buffalo and Saginaw Rivers, meteorological conditions, and the values of some conventional parameters, such as temperature and conductivity. Additional measurements of water surface elevations, water velocities and discharges, and wind velocities and directions were also obtained concurrently with the ARCS field studies.

Sediment Transport Data

Because sediment transport models are driven by hydrodynamic models, their application requires the same data described above. The data for the system's bathymetry should be the same as that used for the hydrodynamic model. However, because a sediment transport model may be used to predict changes in the bathymetry in response to storm or flow events, additional bathymetric information collected periodically (e.g., every 3 months or after large storm events) is highly desirable for evaluating the model's performance.

Data on the initial conditions of sediment properties include particle size distributions, bulk densities, porosity (water content), and resuspension potential. Because sediment properties may vary spatially, characterization or mapping of these properties over the study area is usually necessary. This mapping may be quantitative or qualitative, depending on the objectives of the particular study.

Suspended solids concentrations at the boundaries over time are required to drive the model, and suspended solids data within the modeled system are required for model calibration/evaluation. Because sediment transport is nonlinear and highly dynamic, these data are required as frequently as possible (i.e., continuously if feasible) at least during the course of high-flow events. Collection of these data may often be readily accomplished using automated sampling techniques or using some surrogate measurement (e.g., water transparency) to estimate suspended solids concentrations where a relationship can be established between the surrogate measurement and the suspended solids concentration.

Data used in the ARCS RAM studies of the Buffalo and Saginaw Rivers included historical data, such as Corps dredging records. In addition: 1) data on sediment characteristics (e.g., grain size, water content) were collected during the ARCS sediment surveys, 2) periodic bathymetric surveys were conducted to estimate changes in the system's morphometry, 3) suspended solids concentrations were measured concurrently with the river sampling, 4) suspended solids concentrations were measured either during high-flow events (Buffalo River) or hourly during periods of the year (Saginaw River) to support the sediment transport model, and 5) studies were conducted to estimate the resuspension characteristics of the sediments.

Contaminant Exposure Data

The computation of contaminant exposure also requires information on water and sediment transport from the hydrodynamic and sediment transport models. When not available from models, these data must be derived from field measurements, similar sites, or general guidelines.

Data on boundary conditions include measured concentrations of the contaminant(s) of concern or loadings (mass per unit time) from all significant sources (e.g., tributaries, point and nonpoint sources). Data on initial conditions include existing contaminant concentrations in the water and surface sediments. Contaminant concentration profiles of sediments are required if the impact of exposing deep sediments is to be evaluated. For example, if dredging to a certain depth is proposed, then measurements of contaminant concentrations in the sediment layer expected to be exposed are required. Data should be collected during both high- and low-flow events because the relative importance of processes affecting contaminant transport and fate vary under different flow conditions.

The analytical approach used to support mass balance modeling may differ from that used to support other studies. For example, for regulatory purposes it may be sufficient to know that contaminant concentrations are below some criterion. However, to compute contaminant mass the contaminant concentrations must be accurately known. Appreciable numbers of concentrations below analytical detection limits, "non-detects," may make the data unsuitable for use in mass balance modeling studies. In addition, excessive "noise" in the data (random high or low detections due to sampling or analytical variability) may make it difficult to distinguish trends. Specialized sampling and analytical methods for trace metal and organic analyses are often required to reduce analytical noise and obtain sufficiently low analytical detection limits.

Data are also required for materials that may affect the transport and transformation of contaminants. For example, sorption onto solids affects the transport and transformation of some organic and inorganic chemicals. The prediction of sorption requires that data be available on the fraction of organic carbon in sediments and suspended solids, dissolved organic carbon concentrations, and phytoplankton concentrations. Data on pH are required to predict the speciation of metals and ionization of some organic chemicals. Data on sulfides are required to predict metal precipitation. If data are not available for constituents that affect contaminant transport and transformations, large uncertainties may be introduced in the modeling study. Knowledge of the geochemistry of the contaminant(s) of concern and of the site to be modeled is essential.

In the ARCS RAM near-field studies, both historical and field data collected as part of the ARCS Program for ambient water, sediment, contaminant loading, and food chain relationships were used for the calibration of the contaminant exposure model. For the far-field studies, only historical data were used.

The ARCS RAM near-field studies concentrated on the lower Buffalo and Saginaw Rivers. Although there were some differences, the field sampling plans for the two sites were similar in design. Synoptic surveys were conducted at the Buffalo and Saginaw Rivers to identify spatial variability in the systems during certain low- and high-flow periods. Synoptic surveys provide a "snapshot" of the system, or data at a particular point in time. Six sampling stations were selected to allow estimates to be made of contaminant fluxes into, and out of, the AOCs. Samples were integrated over the width of the system and, in some cases, over depth. Where significant stratification was encountered, samples were collected at several depths. Ambient data for particulate and dissolved contaminants, as well as conventional parameters, were obtained over six sampling days during 1990 and 1991. Selected conventional parameters were measured at a greater frequency to aid in calibration of the hydrodynamic and sediment transport models and to aid in estimating yearly loadings. Examples of the parameters measured for the ARCS RAM mass balance modeling studies are listed in Table 3-1. Sediment contaminant concentrations were measured during separate field sampling studies.

ARCS studies were also conducted to identify contaminant loadings and concentrations in water, sediments, and biota. Contaminant loadings were estimated and/or measured from both point and nonpoint sources. Historical data were used to estimate loadings from point sources. Loading measurements were also acquired concurrently with the ambient water quality studies. Loadings from CSOs in the Buffalo River were estimated based on a limited field sampling program (24 samples at 10 CSOs) and storm water modeling. For the Saginaw River study, CSOs were not identified as a significant source of contaminant loadings and were, therefore, not sampled. Loadings of contaminants and suspended solids from upstream tributaries were based on six daily measurements during the fall of 1990 and the spring of 1991. Historical contaminant, suspended solids, and flow data, as well as data from a suspended solids survey, were collected to extrapolate these measurements to annual loading rates.

TABLE 3-1. EXAMPLES OF PARAMETERS MEASURED FOR THE ARCS RAM MASS BALANCE MODELING STUDIES

Parameter	River	Tributaries, Point Sources	Biota
Dissolved PCBs	X		
Particulate PCBs	X		
Total PCBs	S	X	X
Dissolved PAHs	X	X	
Particulate PAHs	X	X	
Total PAHs (benz[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, chrysene)	S	S	
Dissolved metals	X	X	
Particulate metals	X	X	
Total metals (lead, copper, zinc[a])	S	S	X
Particulate iron	X		
Sulfides[a]	X	X	
Dissolved oxygen	X	X	
Conductivity	X	X	
Temperature	X	X	
pH	X	X	
Alkalinity	X	X	
Suspended solids	X	X	
Dissolved organic carbon	X	X	
Particulate organic carbon	X	X	X
Chlorophyll *a* and phaeophytin	X		
Hardness	X		
Total incident radiation	X		
Light extinction	X		
Wind velocity and direction	X		
Water surface and elevation	X		
Flow	X		
Lipid content		X	X

Note: PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
S - total computed from sum of particulate and dissolved concentrations

[a] Saginaw River study only.

Food Chain Data

The predictions of the food chain model are based on the contaminant concentration data obtained from the contaminant exposure model. That is, the food chain model is the final link in the chain of models relating transport and contaminant transformations to concentrations of contaminants in biota. The exposure concentrations of chemicals are used along with information on the bioenergetics of organisms to estimate concentrations in the biota.

Generally, the species that are of interest in food chain modeling are the higher predators, such as walleye. However, to estimate the variation of contaminant concentrations in these higher predators, and the effects of changing conditions on them, it is necessary to estimate the contaminant concentrations in the components of the food chain leading up to the higher predators. Therefore, one of the initial steps in the food chain modeling is the determination of the components of the food chain for the species of interest. The food chain may not only include different species of fish and forage but may be further subdivided into different life stages because of variations in food or feeding patterns. In addition, many higher predators may migrate into and out of the study area. The preference for a particular forage species may also change seasonally. It is often desirable to select target organisms that have limited and well-defined food chains and migration patterns to minimize the complexity of the food chain model and the extent of the supporting field data that will be required.

In addition to food chain relationships, it is necessary to have information on the bioenergetics of the components of the food chain. This information may include growth rates, reproduction rates, gill characteristics, ingestion rates and uptake efficiencies, swimming rates, and excretion rates. Although site-specific information is preferred, much of these data can be obtained from available literature and databases.

Contaminant concentrations in the various components of the food chain are required to provide a starting point for the model (the initial conditions) as well as to provide for model calibration/evaluation. In addition to contaminant concentrations, data should be collected on other characteristics affecting uptake. For example, data on lipid content are required because it affects the uptake of many hydrophobic contaminants. One difficulty with obtaining data for calibration/evaluation of food chain models, as with contaminant exposure models, is that changes in contaminant concentrations may occur slowly. Particularly in higher predators, changes may occur over seasons and over years so that, ideally, data should be available on those time scales. Therefore, a complete characterization requires either an adequate historical database or long-term field studies. The lack of these data may result in greater uncertainty in model predictions.

Data were collected in ARCS studies to support food chain modeling studies. In the Buffalo River, contaminant concentrations in carp and in their stomach contents were analyzed to establish a relationship between contaminant concentrations in carp tissue and contaminant concentrations in their benthic forage. Carp were selected because there are currently advisories in effect against consumption of carp from the Buffalo River. Data were collected for nine carp composited into three age classes for analysis. Sampling in the Saginaw River concentrated on walleye and its food chain, because of the importance of the walleye fishery in this area.

MODEL APPLICATION

A typical model application involves characterization of the system followed by an evaluation of model predictions. Upon completion of the evaluation, the model is then used to address management questions.

The characterization step involves establishing the boundary conditions for the model (those things outside of the model that affect its predictions, such as inflows and bathymetry), determining the initial conditions of the system (such as initial sediment contaminant concentrations), and determining site-specific parameter values. Some site-specific parameters may be known from previous studies or may be easily measured. Some parameters may be unknown or difficult to measure. Unknown parameters can be estimated by calibrating the model to field data, by applying values that have been established as representative of similar sites, or by using general guidelines to establish values.

Model calibration using site-specific field data is likely to yield the most accurate estimates for unknown parameters, but can only be applied when existing data (e.g., particle distributions, contaminant distributions in water and solids, or contaminant distributions in species) are available. In addition, as the number of unknown parameters increases, so does the difficulty of estimating them by calibrating the model to a single data set. With even two unknown parameters, it is possible that values could be selected so that the model prediction matches a data set used for calibration, but these values may not be intrinsically accurate. Therefore, the values may lead to biased predictions under other conditions, such as a change in contaminant loading.

Parameter values that have been established as representative of a site similar to the one being modeled may be used if those values are the result of direct observation, calibration of the model at that other site, or the prediction of a different model that in turn has been calibrated and verified. Two sites need not be similar in all respects for some parameters to apply equally well to both. For example, similarity of particle settling rates might be established on the basis of bottom type, particle type, particle supply rate, and current velocities; other site characteristics such as size and boundary configuration could be very different.

General guidelines for establishing parameter values may consist of published tables of values, statistical distributions or regressions, or rule-of-thumb calculations. A literature search may be needed to locate such general guidelines, if they exist. One potentially useful compendium of guidelines is Bowie et al. (1985). General guidelines usually suggest a range of parameter values that may be appropriate for a given situation. Such ranges may be used to estimate the potential error of the model's prediction, or they can be used to select an environmentally protective value.

The evaluation step involves comparisons of model predictions to field data. These data are separate from those used in the calibration step. The comparison of predictions to field data in the evaluation step allows an estimation of model uncertainty.

The final step of model application is the use of the model for its intended purpose, that is to address management questions concerning an AOC. Some of the management questions a model may be used to address were listed at the beginning of this chapter.

The model applications at the Buffalo and Saginaw River AOCs resulted in estimated concentrations of selected contaminants in sediment, water, and biota as a function of contaminant loadings. In addition, these models provided a means of estimating the consequences of various remedial or mitigative alternatives. Specific model outputs for each AOC included:

- Loading/response curves for each selected contaminant, relating external loadings to contaminant concentrations in water, sediment, and specific fish species by age group, for each river reach. Uncertainty estimates were provided for loading/response relationships.
- For each selected contaminant, estimated loadings from the AOC to the receiving water body for a variety of flow conditions and as affected by selected remedial or mitigative alternatives.
- Estimated time to recovery in order to assess the no-action alternative as well as the relative benefits of selected remedial or mitigative alternatives.
- Estimates of the relative importance of various processes affecting the transport and transformations of contaminants, such as losses due to volatilization, burial, and outflows.

4. COMPARATIVE RISK ASSESSMENT

The comparative risk assessment is the final step in the comprehensive risk management process (Figure 1-1) that was addressed in the ARCS RAM studies. The comparative risk assessment integrates information from all previous steps in the process to estimate changes in risk, relative to the baseline risk, that would result from implementation of the various sediment remedial alternatives evaluated. Thus, the comparative risk assessment provides a prognostic framework that can aid in addressing management questions. Management questions that may be addressed by the comparative risk assessment, such as the potential impacts of proposed remedial alternatives, generally coincide with those addressed by mass balance models. However, in the comparative risk assessment, contaminant concentration estimates generated by mass balance modeling (Step 7 of Figure 1-1) are used to derive risk estimates for various remedial alternatives being considered within an AOC. The remedial action objectives that had initially been developed during risk assessment planning (Step 2 of Figure 1-1) are then refined during the comparative risk assessment.

The approach used in the comparative risk assessment integrates the results from the baseline risk assessment, field studies, and mass balance modeling studies to provide estimates of the potential impact of remedial actions on human health, wildlife, and aquatic organisms. The output of the baseline risk assessment and modeling studies serves as input to the comparative risk assessment (Figure 4-1). For example, output from the baseline risk assessment includes algorithms, exposure parameters, and toxicity values used for deriving conservative estimates of risks based on current site conditions. Output from the field studies includes measured contaminant concentrations in water, sediments, and biota. Output from the mass balance modeling studies includes estimated contaminant concentrations in water, sediments, and selected fish species following the implementation of proposed remedial alternatives. The comparative risk assessment integrates these outputs to produce estimates of risks for all remedial alternatives under consideration. Thus, the risks associated with each remedial alternative can be compared with the risks associated with the other remedial alternatives, as well as with the baseline risks.

Comparative risk assessment studies were conducted for the Buffalo and Saginaw River AOCs to demonstrate methods for estimating potential changes in risks to humans, wildlife, and aquatic organisms exposed to sediment-derived contaminants under selected remedial alternatives. The comparative risk assessments resulted in estimates of potential risks that may be used in the selection of the most appropriate remedial alternatives.

In the ARCS RAM studies, potential remedial alternatives considered included no action and several dredging and capping scenarios. No action was defined as no change in existing sediment management practices (e.g., continued maintenance dredging).

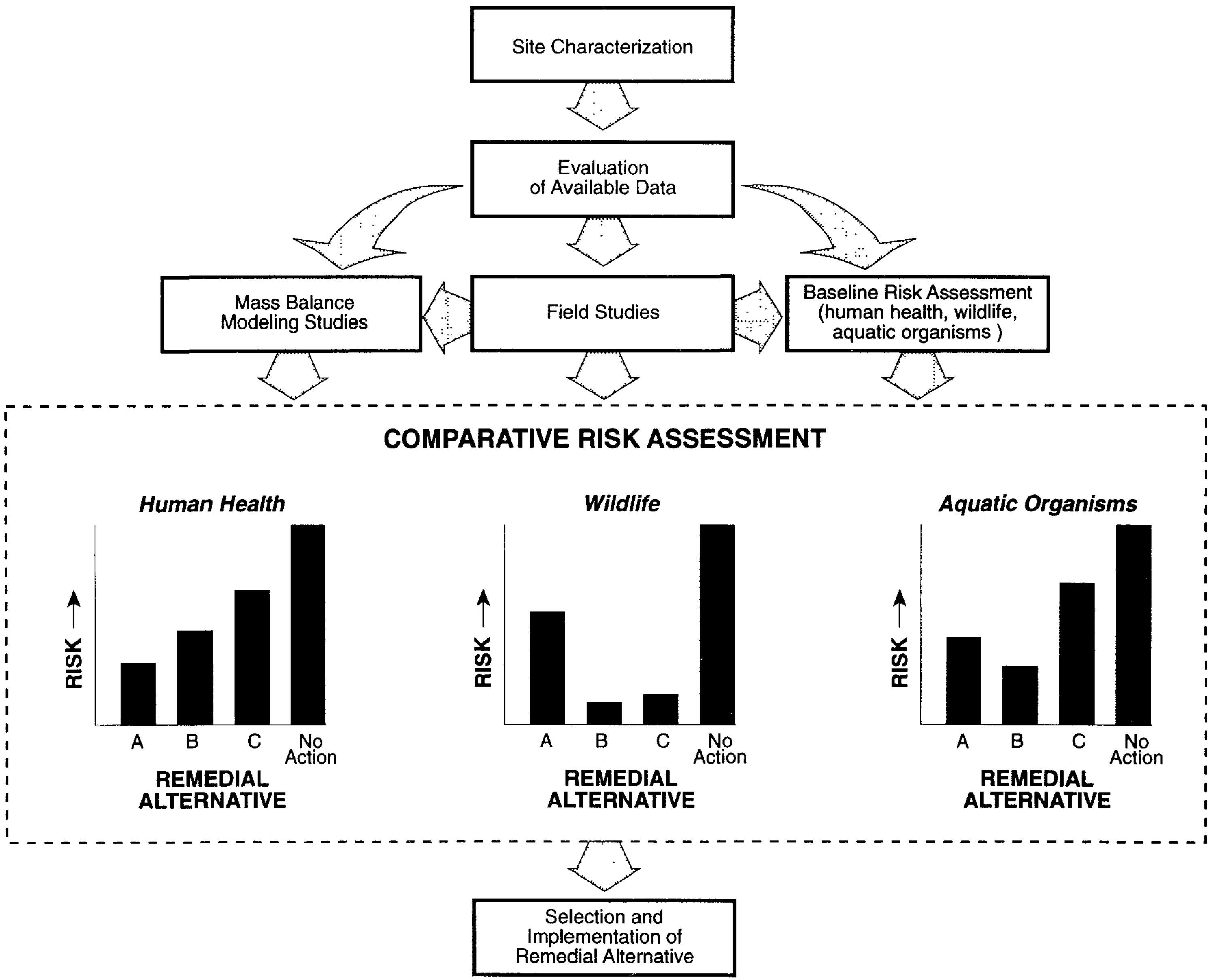

Figure 4-1. Comparative risk assessment in the risk management process.

Dredging options that were considered included complete dredging of contaminated sediments, dredging of hot spots, dredging to a selected depth, and cessation of dredging in part or all of a channel. The comparative risk assessment produced estimates of the potential changes in risk that may result from each of these remedial alternatives. These estimates may then be compared to the existing (baseline) risk to evaluate the relative benefits of remediation and aid in the selection of the most appropriate remedial alternative.

5. SUMMARY

There are currently large numbers of sites in the Great Lakes and elsewhere where contaminant concentrations in sediments are at levels that are of concern to environmental scientists, regulatory agencies, and the general public. Many of these contaminants, such as heavy metals and chlorinated organic compounds, are resistant to degradation by physical, chemical, or biological processes. Therefore, the impact of contaminated sediments may continue well into the future.

Among the primary concerns associated with contaminated sediments are the risks they pose to humans, wildlife, and aquatic organisms. To protect human health, local authorities in many areas have posted fishing and swimming bans, issued fisheries advisories, or ceased drawing drinking water from areas with contaminated sediments. Sediment contamination has made many areas uninhabitable for benthic organisms or has resulted in bioaccumulation through the food chain, adversely affecting both fish and wildlife.

The risks posed by contaminated sediments result from the exposure of organisms to the chemicals through a number of pathways, including direct adsorption from water or sediments or through feeding. However, myriad processes are known to affect that exposure. For example, factors affecting the degree of contamination and exposure may include the degree of ongoing contaminant loading, hydraulic and sedimentation patterns, the physical characteristics of the sediments, the degree of sediment/water interaction, and the characteristics of the chemicals.

As part of the ARCS Program, studies were conducted by the RAM Work Group to demonstrate an integrated approach to evaluating the processes that affect exposure and risks resulting from sediment contamination. This approach is potentially applicable to other areas with contaminated sediments, both in the Great Lakes region and elsewhere. The first step in the approach is the identification of the degree of existing sediment contamination, potential exposure pathways, chemical toxicity, and the resulting potential risks posed to humans, wildlife, and aquatic organisms. The second step, if required, is to conduct field studies to further characterize the distributions and concentrations of contaminants of concern and to aid in identifying factors affecting their transport and transformations. Based on those field studies and risk assessments, priorities can be established for areas that need remediation and potential remedial alternatives can be identified. Next, mass balance models can be used to evaluate the processes affecting exposures and risks and to predict potential changes in conditions that may occur following implementation of the selected remedial alternative(s). Finally, these predicted changes in conditions can be used to estimate potential changes in risks that may occur following remediation.

The integrated approach used in the ARCS RAM studies and described in this report provides a means of evaluating the potential consequences of remediation, in terms of both exposure and risks. The approach can be used to aid in addressing management questions, such as "How long will it take for the problem to go away through natural processes in the absence of active remediation?," "What happens if the sediments are dredged to a particular depth?," or "Will the sediments become recontaminated following remediation?" Like all such approaches, there are various limitations that result from deficiencies in the current understanding of factors affecting the transport and transformations of contaminants, chemical characteristics, or chemical toxicity. However, properly used, the approach may provide a viable means of assessing the nature and extent of sediment contamination and aid in the evaluation and selection of appropriate remedial alternatives.

6. GLOSSARY

Acute—Characterized by a time period that is relatively short in comparison to the life span of an organism. Acute toxicity is the characteristic of a chemical to cause a toxic response in organisms immediately or shortly after exposure to the chemical.

Adverse effect—An impairment of biological functions or description of ecological processes that results in unfavorable changes in an ecological system.

Ambient water quality criterion (plural: criteria)—An estimate of how much of a chemical could be present in the water without harming human health or aquatic life.

Aquatic—Living or growing in water.

Area of Concern (AOC)—A waterbody (e.g., river, harbor, bay) within the Great Lakes basin that has been identified as having impairment of beneficial uses attributable to chemical contamination. In most of the Great Lakes AOCs, sediment contamination is a significant contributor to the impairment of beneficial uses.

Assessment endpoint—An ecological value to be protected (e.g., trout population abundance or community structure that indicates a "healthy" biological community).

Baseline risk assessment—An assessment that estimates risks associated with existing environmental conditions.

Benthic—Pertaining to, or associated with, the bottom of a body of water.

Bioaccumulation—Net uptake of a chemical into the tissues of an organism as a result of direct contact with a medium, such as water or soil, or through the diet.

Biodegradation—The decomposition of a chemical substance by natural biological processes.

Biomagnification factor—A measure of the degree of increase in the tissue concentration of a chemical with each trophic step in a food chain. For example, a biomagnification factor of 5.0 indicates that the concentration of a given chemical in the tissues of a predator is 5 times the concentration of that chemical in the tissues of its primary prey species.

Bioavailability—The degree to which a chemical can be taken into the tissues of an exposed organism.

Biomass—The total weight of live organisms in a sampled population or community.

Cancer slope factor (CSF)—Plausible upper-bound estimate of the probability of a response per unit of intake of a chemical over a lifetime. The cancer slope factor is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a particular carcinogen.

Carcinogenic—Capable of causing cancer in an organism.

Chronic—Characterized by a time period that represents a substantial portion of the life span of an organism (e.g., chronic toxicity is the characteristic of a chemical to produce a toxic response when an organism is exposed over a long period of time).

Chronic intake level—Exposure expressed as the mass of a substance contacted per unit body weight over a long-term exposure period, often expressed as mg/kg-day over a lifetime.

Community—Interacting populations of species (plants or animals) living in the same habitat.

Comparative risk assessment—An evaluation of the changes in human health and/or ecological risks resulting from a range of candidate remedial alternatives. Ideally, the comparative risk assessment should include evaluation of the risks associated with all components (e.g., removal, pretreatment, treatment, disposal) of each remedial alternative under consideration.

Compensatory mechanism—A biological process that offsets or counteracts an adverse effect (e.g., increased survival of young fish related to reduced competition because egg hatching success was reduced).

Concentration—The amount of a chemical substance expressed relative to the amount of environmental medium (e.g., μg/L [micrograms of chemical per liter of water] or μg/g [micrograms of chemical per gram of soil]).

Conceptual model—A simplified description of important functional or structural relationships in an ecosystem, including working hypotheses of how chemicals might affect populations or communities.

Contaminant exposure model—A mass balance model for contaminants, in which the rate of change in contaminant mass (accumulation) is a function of the transport of a contaminant into, out of, and within the system (via water transport or sediment transport for those materials that sorb to sediments); the mass added to the system (via point and nonpoint loadings) minus the outputs; and the quantities transformed and degraded within the system (via processes such as volatilization, biodegradation, and photodegradation).

Contaminant of concern—A chemical or specific form of a chemical suspected of being present in concentrations in the environment that may cause adverse effects to humans or ecological receptors.

Dose—The amount of chemical taken into an organism per unit of time.

Dose-response relationship—The relationship between the dose of a contaminant administered or received and the incidence of adverse effects in the exposed population. From the quantitative dose-response relationship, toxicity values are derived that are used in the risk characterization step to estimate the likelihood of adverse effects occurring in humans at different exposure levels.

Ecological effects assessment—An assessment conducted to determine the relationship between the levels of contaminant exposure (or other stressors) and the levels and types of ecological effects.

Ecological epidemiological approach—An empirical assessment approach based on an evaluation of existing ecological effects, especially the establishment of the causes of reduced population abundances and alterations of community structure.

Ecological risk assessment—Evaluation of the likelihood of adverse effects on organisms, populations, and communities from chemicals present in the environment.

Ecosystem—An ecological community of plants and animals together with its physical environment, regarded as a unit.

Endpoint—The biological or ecological unit or variable being measured or assessed (see measurement endpoint and assessment endpoint).

Equilibrium partitioning model—A mathematical expression that describes the distribution of a chemical between sedimentary organic carbon and interstitial water based on the assumption of thermodynamic equilibrium.

Exposure—Contact between a human or ecological receptor and a chemical in the environment.

Exposure assessment—The portion of the risk assessment that describes the frequency, magnitude, and duration of exposure of human or ecological receptors to contaminants of concern.

Exposure duration—In human health risk assessment, the estimated number of years over which exposure to contaminated media may occur.

Exposure frequency—In human health risk assessment, the number of days per year that a person may contact contaminated media.

Exposure parameters—Values used to estimate exposure in a risk assessment, such as the number of days per week that exposure is expected to occur, or the amount of contaminated media that a person might incidentally ingest per day.

Exposure pathway—The path a chemical takes or could take from a source to exposed organisms. Exposure pathways include the source, the mechanism of release and transport, a point of contact, and the means of contact (e.g., ingestion or inhalation).

Exposure point concentration—The concentration of a chemical at the point of exposure to an organism. For example, the concentration in the soil in which a plant is growing.

Exposure-response assessment—A description of the relationship between the concentration (or dose) of the chemical that causes adverse effects and the magnitude of the response of the receptor.

Exposure route—The means of contact between an organism and a toxic chemical (e.g., eating [ingestion], breathing [inhalation], or touching [dermal contact]).

Exposure scenario—A conceptual model of how exposure takes place, including specific combinations of exposure media, pathways, and receptors and organism activities that may lead to exposure.

Food chain—A sequence of species at different trophic (feeding) levels that represent a single path of energy within a food web. For example, grasses and seeds are eaten by a mouse which is then eaten by an owl. The owl is higher up the food chain (at a higher trophic level) than the mouse.

Food web—Interconnected food chains that describe the pathways of energy and matter flow in nature.

Forward-mode assessment—The use of ecological risk assessment techniques to estimate risk (see reverse-mode assessment).

Fractional intake—The fraction of total contaminant intake that occurs at an AOC. For example, an exposure assessment might estimate that an individual ingests 56 g of fish per day from all sources, but that the fractional intake of fish consumed from the specific AOC is only 0.1, indicating that only one-tenth of the total fish consumption (or 5.6 g per day) is from the AOC in question.

Habitat—The place where animals and plants normally live, often characterized by a dominant plant form or physical characteristic.

Hazard—The ability of a physical, chemical, or biological agent to harm plants, animals, or humans under a particular set of circumstances.

Hazard identification—The stage of the toxicity assessment that defines the qualitative relationship between chemicals and adverse effects to receptors.

Hazard index (plural: indices)—The sum of more than one hazard quotient for multiple substances and/or multiple pathways. The hazard index is calculated separately for chronic, subchronic, and shorter duration exposures.

Hazard quotient—The ratio of a single substance exposure level over a specified time period (e.g., subchronic) to a reference dose for that substance derived from a similar exposure period.

Human health risk assessment—Prediction of the likelihood of adverse effects in human populations through calculations combining quantitative estimates of the toxicity of chemical contaminants in the environment with quantitative estimates of the potential for human populations to be exposed to those contaminants.

Hydrodynamic model—A description of the transport of water, or its motion, based on a balance of the water's momentum.

Intake—A measure of exposure expressed as the mass of a substance in contact with the exchange boundary per unit body weight per unit time (e.g., mg chemical/kg body weight-day)

Joint probability analysis—A statistical technique used to estimate the likelihood of chemical concentrations exceeding toxicity criteria.

LD_{50}—Dose of a substance that results in 50-percent mortality in a population of test organisms.

Life stage—A developmental stage of an organism (e.g., juvenile, adult, egg, pupa, larva).

LOAEL (lowest-observed-adverse-effect level)—The lowest concentration or dose at which significant adverse effects were observed in experimental trials.

Macroinvertebrate—An invertebrate organism visible to the naked eye. Often refers to animals such as insects, worms, and snails.

Mass balance model—A quantitative description of the sources, transport, and fate of the mass of a substance (e.g., water, sediment, or chemical contaminants), such that the mass entering the system equals the mass leaving the system, less the mass stored, transformed, or degraded.

Measurement endpoint—An ecological variable that is measured to quantify the response of an organism, population, community, or ecosystem to chemicals. Each measurement endpoint is related to an assessment endpoint.

Medium (plural: media)—The substance in which a chemical may exist, such as air, soil, sediments, and water.

No-action alternative—The alternative in which no remedial action is taken.

Noncarcinogenic—Capable of causing chronic or subchronic effects other than cancer in an organism.

NOAEL (no-observed-adverse-effect level)—The highest concentration or dose at which no significant adverse effects were observed in experimental trials.

Organism—An individual plant or animal.

Photodegradation—The decomposition of a chemical substance by radiant energy, generally natural sunlight.

Population—A group of individuals of the same species interacting within a given habitat.

Predictive approach—Any assessment approach that estimates risks based on assumed scenarios (e.g., future conditions), extrapolation models, or theory rather than direct measurement.

Probability—The likelihood of an event occurring, expressed as a numerical ratio, frequency, or percent.

Quotient method—The process of comparing a concentration or dose (estimated or measured) with a concentration or dose known to have adverse effects on organisms.

Receptor—The organism, population, or community that might be affected by exposure to a contaminant of concern.

Reference area—An area that has similar physical characteristics to a site being evaluated, but is unaffected by contaminants of concern. The reference area is compared to the site to assess the effects of contaminants of concern.

Reference concentration (RfC)—For assessment of inhalation exposures, the contaminant concentrations in the air at which no adverse effects are expected to occur.

Reference dose (RfD)—For an individual chemical, an estimate of an exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of noncarcinogenic effects. There are chronic, subchronic, and developmental reference doses, but when used without a modifier, reference dose is generally understood to mean the chronic reference dose, or the acceptable daily exposure level that is likely to be without an appreciable risk of noncarcinogenic effects during a lifetime.

Remedial action objectives—The general descriptions of what remedial actions should accomplish (e.g., to reduce risks to particular species of plants and animals at a site).

Remedial action goals—A subset of remedial action objectives consisting of medium-specific chemical concentrations that are protective of human health and the environment.

Remedial Action Plan—A detailed description of the activities selected for the remediation of contamination (especially sediments) within a given AOC.

Remedial action alternative (or remedial alternative)—A combination of technologies used in series and/or parallel to isolate contaminated sediments or to alter the concentrations of sediment contaminants in order to achieve specific project objectives. In the simplest case, a remedial alternative may employ a single technology, such as *in situ* capping. In more complex cases, a remedial alternative may involve several technologies, such as dredging, pretreatment, treatment, and confined disposal.

Remediation—Action taken to control the sources of contamination and/or to clean up contaminated media (e.g., sediments).

Retrospective approach—Any empirical assessment approach based on evaluation of existing ecological effects and stressors.

Reverse-mode assessment—The use of ecological risk assessment techniques to derive criteria or cleanup levels corresponding to a specified risk level (e.g., acceptable risk level set by policy) (see forward-mode assessment).

Riparian—The land and habitat along the bank of a stream, river, or lake. The riparian area of a river or stream includes the active flood plain (contrasted with upland).

Risk assessment planning—A step in an ecological risk assessment that evaluates physical, chemical, and biological characteristics of a site to provide a preliminary risk characterization, to determine whether an ecological risk assessment is warranted, and, if so, to develop risk assessment objectives.

Risk characterization—The step in an ecological risk assessment in which information on exposure and toxicity are combined to estimate the probability of adverse effects on organisms, populations, or communities.

Risk index—An expression of the potential for adverse effects to the biological community derived from endpoints. For example, the quotient of exposure concentrations to species toxicity values.

Risk management—The process of integrating findings from a risk assessment with engineering, policy, and nontechnical concerns to make decisions about the need for remediation at a specific site or to set remediation priorities among sites.

Screening level—A process or criterion that separates sites that pose no apparent risk from those for which further analysis is necessary.

Sediment transport model—A description of the physical transport of sediments in natural systems, in either bed-load or suspended forms.

Speciation—Refers to the various forms in which metals occur.

Stressor—A physical, chemical, or biological agent that can induce an adverse response in organisms or other components of ecosystems.

Subchronic intake level—Exposure expressed as the mass of a substance contacted per unit body weight over an exposure period of less than a lifetime, often expressed as mg/kg-day over 1-10 years.

Terrestrial—Living or growing on land.

Threatened or endangered species—Species that are at risk of becoming extinct.

Threshold—The chemical concentration (or dose) at which physical or biological effects begin to be produced.

Toxicity—The property of a chemical substance manifested as its ability to cause a harmful effect (e.g., death, disease, reduced growth, modified behavior) on an organism.

Toxicity assessment—The stage of a risk assessment that describes the potential effects of a chemical on organisms and the quantitative exposure-response relationship.

Toxicity test—A test in which organisms are exposed to chemicals in a test medium (e.g., waste, sediment, soil) to determine the effects of exposure.

Transport and fate—A description of how a chemical is carried through the environment. This may include transport through biological as well as physical parts of the environment.

Uncertainty analysis—An evaluation, qualitative or quantitative, of parameters or assumptions used in a risk assessment that are not completely known or cannot be precisely estimated, which is used to help place quantitative risk estimates in perspective.

Upland—Land usually above the floodplain of a river or stream (contrasted with riparian).

Volatilization—The conversion of a chemical substance from a liquid or solid state to a gaseous or vapor state by the application of heat, by reducing pressure, or by a combination of these processes.

Water transport model—A description of the transport of water, or its motion, in natural systems that may be descriptive (i.e., based on a balance of the water's mass) or hydrodynamic (i.e., based on a balance of the water's momentum).

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